EUROPEAN COMMISSION
Community Research

ERICA

(Contract Number: FI6R-CT-2003-508847)

DELIVERABLE D9:
Application of FASSET framework at case study sites

Editor(s): **N.A. Beresford & B.J. Howard**

Reporting period: 01/03/04 – 31/04/05

Date of issue of this report : **05/04/05**

Start date of project : **01/03/04** Duration : 36 Months

Project co-funded by the European Commission under the Euratom Research and Training Programme on Nuclear Energy within the Sixth Framework Programme (2002-2006)		
Dissemination Level		
PU	Public	PU
RE	Restricted to a group specified by the partners of the [ERICA] project	-
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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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Contract No:	**FI6R-CT-2003-508847**
Project Coordinator:	**Swedish Radiation Protection Authority**

Contractors:

Swedish Radiation Protection Authority	SSI
Swedish Nuclear Fuel and Waste Management Company	SKB
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Agricultural University of Norway	NLH
Electricité de France	EDF

List of contributors

Editors: N.A. Beresford & B.J. Howard
(NERC Centre for Ecology & Hydrology, Lancaster, UK).

Case study chapter authors:

Chapter 3 *Sellafield* – C. Johnson, W.A. Marshall & S.R. Jones (WSC)

Chapter 4 *Loire River* - K. Beaugelin-Seiller (IRSN) & J. Charton (EDF)

Chapter 5 *Oil and gas platforms* – J. Brown & T. Gäfvert (NRPA)

Chapter 6 *Chernobyl exclusion zone* – N.A. Beresford, C.L. Barnett & S.M. Wright (NERC – terrestrial); A. Ulanovsky & G. Pröhl (GSF – freshwater)

Chapter 7 *Komi* – J. Brown & H. Thørring (NRPA)

Other contributors:

D. Copplestone (EA); L. Moberg (SSI); J. Vives i Batlle (WSC); M. Wood (UNILIV)

Executive summary

The assessment tools which will be the outputs of the ERICA project will be based on the FASSET framework for assessing the environmental impact of ionising radiation. This deliverable describes the application of the FASSET framework to five different case study sites. The case study sites were selected to test all of the components of the FASSET framework and included: (i) sites contaminated by anthropogenic releases of radioactivity and technologically enhanced natural radionuclides; (ii) regulated sites; (iii) contaminated areas where potential radiation induced effects had been observed; (iv) marine, freshwater and terrestrial ecosystems. The case study sites were: Sellafield, Loire River, North Sea oil and gas platforms, the Chernobyl exclusion zone and areas of enhanced natural radionuclides in the Komi Republic. The objectives of the case study applications were:

- to assess of the applicability of the FASSET framework methodology;
- to compare predicted and observed activity concentrations in biota (and water/sediments for aquatic ecosystems);
- to identify data gaps;
- to compare, where possible, observed radiation induced effects with estimated doses and predicted effects;
- to make recommendations to the ERICA project to guide developments of the ERICA assessment tools.

The process of applying the FASSET framework to different case studies has been valuable in highlighting areas of improvement for consideration during the ERICA project resulting in the following recommendations:

- ERICA should consider the scenarios it expects its tools to address. ERICA should be clear in its output when the methodology will and will not be applicable, considering: equilibrium, site specific factors and historic discharges.
- The guidance produced by ERICA must be user friendly and concise, it needs to clearly guide the assessor through the conduct and interpretation of all stages of the assessment, providing: interpretation of results at the various stages, guidance on how to proceed if required data or parameters are missing, guidance on how to take background exposure into account and guidance on chemical toxicity
- The ERICA tool and other outputs presenting guidance must be consistent, and their purpose and status clear. A consistent terminology must be used. Consideration should be given to providing guidance on how to present the assessment process and results to an interested but non-technical audience.
- The ecosystems and reference organisms considered by ERICA should be rationalised and consideration given to interface between different ecosystems. The reference organism list should encompass protected species, for instance, terrestrial birds and amphibians. The additional radionuclides identified in the case study assessments need to be prioritised for inclusion within ERICA.

Many aspects of the FASSET framework which could be improved during the development of the ERICA tools were identified. It will not be possible to address all of these within the resources and timescale of the ERICA project. We therefore need to agree and prioritise the recommendations. A fundamental question to ask during this prioritisation is: *how (and where) do we envisage the ERICA tools will be applied by end-users and what will they expect of it?* Interaction with end-users within the ERICA project may help in addressing this important question.

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1 Introduction

The assessment tools which will be the outputs of the ERICA project are based on the framework for assessing the environmental impact of ionising radiation previously developed by the FASSET project [Larsson *et al*. 2004; Williams 2004]. A critical analysis of the components of the FASSET framework is required to help identify and prioritise areas needing improvement.

This deliverable describes the application of the FASSET framework to five different case study sites. The case study sites were selected to test many of the components of the FASSET framework. They included: (i) sites contaminated by anthropogenic releases of radioactivity and technologically enhanced natural radionuclides; (ii) regulated sites; (iii) contaminated areas where potential radiation induced effects had been observed; (iv) marine, freshwater and terrestrial ecosystems. The case study sites and brief justification for their selection were:

Sellafield, UK (terrestrial ecosystems) – Regulated site, comparatively high actinide levels previously measured on coastal saltmarshes, a number of local sites protected under international and national environmental legislation.

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

Oil and gas platforms, Norwegian sector of North Sea (marine ecosystem) – Potential significant source of technologically enhanced alpha-emitting radionuclides.

Chernobyl exclusion zone, Ukraine (terrestrial and freshwater ecosystems) – Comparatively good datasets available on Cs and Sr activity concentrations in biota, radiation induced effects reported in biota.

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

The objectives of the case study applications were:

- to assess the applicability of the FASSET framework methodology;
- to compare predicted and observed activity concentrations in biota (and water/sediments for aquatic ecosystems);
- to identify data gaps;
- where possible to compare observed radiation induced effects with estimated doses and predicted effects;
- to make recommendations to the ERICA project to guide developments of the ERICA assessment tools.

In the following section the FASSET framework is outlined. The application of the framework is then described in a separate section for each case study site. The final section of this report summarises the case study findings and presents recommendations for improvement of the framework for consideration by the ERICA project. Whilst the case study reports have been edited into a format suitable for this deliverable, the views of each assessing group have been retained in the case study assessment. In some of the case study reports, editorial comment can be found as footnotes on specific aspects of the interpretation/implementation of the FASSET framework; those attributable to the editors are marked '(eds)'.There is editorial comment (discussed and agreed with the assessors) on some of these views and on the methods of application in the final section of this deliverable.

2 The FASSET framework – an overview

Larsson *et al*. [2004] provide a step by step overview of the elements of the FASSET framework and its implementation from source characterisation through to interpretation of the assessment. The information and tools provided for each of these steps are described below.

2.1 Source characterisation

The need to characterise both the nature of the radionuclides released into the environment and the nature of the receiving environment are discussed. The application of screening or guidance levels within this stage of the assessment is also discussed. However, FASSET did not make any recommendations on screening levels (i.e. estimated dose rates or media activity concentrations) for use in environmental impact assessments.

The selection/screening criteria listed as potentially being required during the first stage of an assessment include: source term description (total releases and relative contribution of each radionuclide; temporal changes in releases; pathways between release and receiving ecosystem; speciation of released radionuclides), physical parameters (half-lives; radiation type and energy), environmental fate (solubility; sorption behaviour; isotopic dilution), biological activity (degree of hydrolysis; reaction with biological ligands), and potential chemical toxicity.

The FASSET tools consider 37 radioisotopes of 20 elements. These were selected to include a range of environmental motilities and biological uptake rates, anthropogenic and natural radionuclides likely to arise in assessments of different sites, and α-, β- and γ-emitters.

2.2 Ecosystems and selection of reference organisms

The FASSET framework considers a number of different European ecosystem types (described in detail in the appendices to Strand *et al*. [2001]):

Forests - Communities dominated by trees.

Semi-natural pastures and heathlands - A broad range of ecosystems including mountain .and upland grasslands, heath and shrub lands, saltmarshes and some Arctic ecosystems. These ecosystems are termed 'semi-natural' since, whilst they comprise natural species not introduced by man, they have been influenced by human use.

Agricultural ecosystems - Arable land, intensively managed pastures and areas used for fruit production. Within FASSET wildlife have not been considered as part of the agricultural ecosystem.

Wetlands - Areas of marsh, fen, peatland, etc., whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt.

Freshwaters - All freshwater systems including rivers and lakes.

Marine - European marine ecosystem (North-Eastern section of the Atlantic Ocean and its marginal seas including the Mediterranean Sea, Greenland Sea, the Irish Sea, North Sea, Norwegian Sea, Skagerrak, Kattegat and Barents Sea).

Brackish waters - Within the scope of FASSET this only includes the non-tidal, shallow Baltic Sea; organisms are immigrants from either marine or freshwater systems.

To overcome problem of the vast number of species which may require consideration within an environmental impact assessment, FASSET adopted the 'reference organism' concept as proposed by Pentreath & Woodhead [2001] amongst others. FASSET's definition of the reference organism was: "*a series of entities that provide a basis for the estimation of radiation dose rate to a range of organisms which are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects*". The reference organisms were thought of as a set which, taken together, are likely to cover the range of both radiation exposures and radiosensitivities which may arise within contaminated ecosystems. *Circa* 30 reference organisms were selected for each of the ecosystem types [Strand *et al*. 2001] based on a consideration of:

- whether the habitat or feeding habits of the organism are likely to maximise its potential exposure to radionuclides, based on an understanding of the distribution of the different radionuclides within the ecosystem;
- whether the organism exhibits radionuclide-specific bioconcentration which is likely to maximise internal radionuclide exposures in particular circumstances;
- whether the position of the organism within the foodchain is such that biomagnification of radionuclides up the foodchain may lead to enhanced accumulation;
- comparative radiosensitivity (including different life-stages);
- ecological function.

2.3 Exposure assessment

The methodologies within the FASSET framework used to estimate exposure of organisms are detailed in Brown *et al*. [2003a]. The framework recommends the use of measured whole-body activity concentrations in biota to determine internal dose. However, for use when this is not possible, look-up tables presenting mean 'equilibrium' ratios of the activity concentrations in biota to the activity concentration in media (water, air or soil) (CRs) are provided[1]. There are many missing concentration ratio values in the look-up tables. The CR values are based upon a mixture of empirical data compilation and model prediction. Building upon recommendations of Copplestone *et al*. [2003] the FASSET framework includes some advice on how to address these data gaps. Sediment – water distribution coefficients (k_d) are also presented for aquatic ecosystems to enable the prediction of water activity concentrations from those of sediments (or *vice versa*).

Absorbed dose is used as the quantity for assessing exposures to ionising radiations. Unweighted dose conversion coefficients (DCCs; see Pröhl [2003] for derivation of DCCs) are presented to allow the estimation of: (i) internal dose rate from measured or modelled whole-body activity concentrations; and (ii) external dose rate on the basis of media activity concentrations[1]. Dose conversion coefficients are presented for a range of geometries and sizes selected to be representative of the reference organisms (e.g. for terrestrial carnivorous mammals DCCs are presented for geometries and shapes approximating to a fox and a

[1] Note DCC and CR values are also provided for terrestrial ecosystems to enable assessments under conditions of chronic deposition relating absorbed dose and whole-body activity concentrations to the annual deposition of a given radionuclide. These have not been applied in this report.

weasel). For a given geometry, external exposure DCCs are presented for different media (e.g. to enable dose to be estimated for biota in the water column, in soil, on soil etc.). For terrestrial plants, DCCs are presented for external doses to bud and meristem only. The total absorbed dose is estimated as the sum of the internal and the external doses; external dose rates being adjusted for the amount of time a organism spends in a specific habitats.

Radiation weighting factors and appropriate values for the relative biological effectiveness (RBE) are discussed in a number of the FASSET deliverables. To illustrate the potential impact of the value of RBE on nuclide specific DCCs, Pröhl [2003] used values of 10 for internally incorporated α-radiations and 3 for low energy (<10keV) β-radiations. Woodhead & Zinger [2003] suggested that to demonstrate the influence of radiation quality, estimations of weighted dose should use RBE values for α-radiations of 5, 10 and 50.

The potential contribution of background exposure (to natural radionuclides) is discussed within the FASSET documentation [Pröhl 2003] although no guidance on the consideration of background exposure within the assessment process is given.

Brown *et al.* [2003a] recommend that uncertainty analyses are always conducted as an integral part of the assessment (page 50 of Brown *et al.* [2003a]). However, whilst some general guidelines on how to conduct uncertainty analyses are provided, the FASSET framework does not provide the ability to conduct uncertainty analyses (e.g. only mean CR values are presented).

To assist in assessments, 'life-history' information is presented for species representative of each reference organism group [Brown *et al.* 2003a]; these are usually aligned to the species selected to represent the geometries and sizes used to derive DCC's. This contains information which may be useful in an assessment such as occupancy factors for different habits/media (e.g. amount of time spent in soil compared to on soil by representative burrowing mammals).

2.3.1 Application within ERICA case study assessments

For the purposes of implementation within the ERICA case studies, the FASSET CR and DCC values were incorporated into a spreadsheet tool based on that developed by Copplestone *et al.* [2001].

For marine and freshwater ecosystems the implementation of DCCs within the spreadsheet tool is as suggested by Pröhl [2003] with the exception that values presented as negatives (in Brown *et al.* [2003a]) were assumed to be zero. The FASSET framework contains separate DCCs for terrestrial ecosystems for animals on the soil surface and animals in soil (i.e. for burrowing mammals, soil invertebrates etc.). However, a number of discrepancies were noted in the external DCCs for animals in soils (e.g. external DCCs for ^{90}Sr were lower for animals in soil than animals on the soil surface). Furthermore, external doses from β-emitters were often zero whereas external DCC values were given for the same radionuclides in aquatic systems. In the lack of resolution of these discrepancies a decision was made to use the FASSET external on soil DCC values multiplying these by 2 when animals were in soil (taking into account the 4 π geometry for submerged organisms; see Vives i Batlle *et al.* [2004]).

The FASSET DCCs for a given radionuclide are the sum of contributions from α-, β- and γ-emissions (including their daughter products with half-lives of <10 days). To enable the application of a radiation weighting factor, the DCC values have to be broke down into their components parts.

2.4 Effects analysis and database

A review of available radiation effects data was performed within the FASSET project [Woodhead & Zinger 2003] and the information organised into a database (*FRED* – the FASSET Radiation Effects Database). The database includes approximately 25000 data entries from more than 1000 references. It is structured to allow the user to search for information based on wildlife group (generally the same as reference organism) and umbrella endpoint (i.e. morbidity, mortality, reduced reproductive success and mutation). Tabulated output summaries from the database are presented for the different wildlife groups within the FASSET framework [Larsson *et al*. 2004]. The fragmentary nature of the available, and relevant, chronic exposure information made it difficult to develop (the desired) dose rate - response relationships in any detail. However, some very broad and general conclusions were drawn. Although minor effects may be seen at lower dose rates in sensitive species and systems, (e.g. haematological cell counts in mammals, immune response in fish, growth in pines, and chromosome aberrations in many organisms) the threshold for statistically significant effects in most studies was about 0.1 mGy h^{-1} (i.e. 2.4 mGy d^{-1}); responses then increased progressively with increasing dose rate usually becoming very clear at dose rates >1mGy h^{-1} over a large fraction of a lifespan. However, it was advised against using this information for establishing 'environmentally .safe'. levels of radiation as the database contained large information gaps for environmentally relevant dose rates and ecologically important wildlife groups.

2.5 The FASSET deliverables

Within the following sections we will refer to a number of the FASSET deliverables (reports) these are listed in Table 2.1 (all the deliverables are available from http://www.erica-project.org/).

Table 2.1: The deliverables of the FASSET project.

Output	Title	Editors
Deliverable 1 (main report and two appendices)	Identification of reference organisms from a radiation exposure pathways perspective (48pp) Appendix 1: Ecological characteristics of European terrestrial ecosystems. Overview of radiation exposure pathways relevant for the identification of candidate reference organisms (115 pp) Appendix 2: Ecological characteristics of European aquatic ecosystems. Overview of radiation exposure pathways relevant for the identification of candidate reference organisms (79 pp)	Strand *et al.* [2001]
Deliverable 2 (two reports)	Part 1: Formulating the FASSET Assessment Context (77 pp) Part 2: Overview of programmes for the assessment of risks to the environment from ionising radiation and hazardous chemicals (84 pp)	Larsson *et al.* [2002a;b]
Deliverable 3 (report)	Dosimetric models and data for assessing radiation exposures to biota (103 pp)	Pröhl [2003]
Deliverable 4 (report and database)	Radiation effects on plants and animals (196 pp) FASSET radiation effects database, FRED (separate deliverable under D4)	Woodhead & Zinger [2003]
Deliverable 5 (main report and two appendices)	Handbook for assessment of the exposure of biota to ionising radiation from radionuclides in the environment (101 pp) Appendix 1: Transfer factors and dose conversion coefficient look-up tables (111 pp) Appendix 2: Underpinning scientific information (life history sheets, empirical data and models) (183 pp)	Brown *et al.* [2003a]
Deliverable 6 (report)	Framework for assessment of environmental impact of ionising radiation in major European ecosystems (70 pp)	Larsson *et al.* [2004]

3 Sellafield case study report

The Sellafield case study was envisaged to be an assessment of environmental risks from ionising radiation on both agricultural and semi-natural ecosystems in the vicinity of the BNFL Sellafield site (including the sand dune and saltmarsh areas of the Drigg Coast Natura 2000 designated conservation site). Uniquely among the case studies, the Sellafield case study has sought input from local stakeholders to identify needs or concerns specific to the study area and to attempt to integrate these into this assessment. Two stakeholder meetings were held to elicit views both before the assessment and when results were available. The stakeholders included representatives of: local conservation groups, local (parish) government, farming and tourist industries, the Lake District National Park, local interest groups, the Sellafield site operator and government agencies responsible for environmental protection and nature conservation in the area. The stakeholder groups at the two meeting differed (with the exception of a governmental environmental agency representative). At the pre-assessment meeting, the stakeholders expressed the view that artificial boundaries were being placed between sites. Therefore, it was decided to undertake both agricultural and semi-natural assessments on both study sites fully integrating semi-natural species found on and around farmland and agricultural species found on the Drigg saltmarshes into the assessment. As a result, the case study has focused on species found at the Low Church Moss Site of Special Scientific Interest (SSSI) to represent semi-natural species associated with farmland near Sellafield and sheep and the vegetation they graze at the Drigg coast site.

3.1 Exposure assessment

3.1.1 Drigg coast

Drigg Coast is an estuarine system, designated as both a Site of Special Scientific Interest (SSSI) and a candidate Special Area of Conservation (cSAC), that extends for almost 11 km along the West Cumbrian coast from Seascale, south towards Bootle. Drigg Coast is one of the best examples in the UK of a small bar-built estuary complex, which is one of the most natural and least developed estuaries in the UK, with little industry and virtually no artificial coastal defences. The estuary is fed by the Rivers Irt, Mite and Esk, which discharge through a mouth that has been narrowed by large sand and shingle spits on which the Drigg and Eskmeals dune systems have developed. Within the site, there is an excellent sequence of saltmarsh habitats from pioneer through to upper marsh with relatively undisturbed transitions to terrestrial habitats, particularly to sand dune, shingle and freshwater swamp. These are some of the best examples of transitions from saltmarsh to freshwater and sand dune habitats of any estuary in the UK; such successions are absent in most other British estuaries [JNCC, 2004].

The site is designated as a cSAC due to the presence of the following Annex I habitats: estuaries, Atlantic decalcified fixed dunes and dunes with *Salix repens* ssp. *argentea* (*Salicion arenariae*). Other Annex I habitats are also present but are not a primary reason for cSAC selection (further details of the site and a species list can be found in Johnson & Marshall [2005]).

Much of the Drigg coast is grazed and the intensity of grazing has a large effect on the species compositions of the communities in these areas. Grazing is an important factor in the management of the site and goes a long way to maintaining habitat and halting or slowing down the ecological succession to mature woodland communities.

3.1.2 Agricultural land around Sellafield

The agricultural assessment includes not only the agricultural species grown on farms in the vicinity of Sellafield but also the semi-natural species present in the same area. Within the 3 km area to be examined there is one SSSI, Low Church Moss and it is the species present on this site that have provided the basis for the semi-natural assessment.

Agricultural area

Little published information is available regarding the agricultural practices around Sellafield. However, during 1988 a detailed survey was carried out to determine the habits of people living within a 3 km radius of the Sellafield nuclear reprocessing plant including details of types of livestock and crops produced [Stewart *et al*. 1990]. From this information a broad outline of farming practices in the late 1980's could be obtained. Seventeen farms were visited within the area – 12 were milk producers, 8 produced finished beef cattle, 14 finished sheep, 1 kept pigs and 6 had poultry. The pig and poultry producers were small scale operations, the output being mainly used for home or family consumption. Several farms also produced beef cattle of various ages, which were sold outside the area for further finishing before slaughter. These were not included in the figure for beef producers. Several arable crops were grown, the majority being for animal feeding on the premises where they were produced. Eight farms grew potatoes. Other root crops were grown on 8 farms and vegetables by three farms, mainly for home use. Barley was grown on 11 farms and oats on 1 for animal consumption on the farm.

During the pre-assessment stakeholder consultation, discussions indicated that farming practises have changed slightly since the 1980's. The number of farms that produce potatoes has reduced and the number of farms within the area has also reduced. This is due in part to the expansion of the Sellafield site and also to farms increasing in size. With the consolidation of farms, farmers tend to concentrate on one crop rather than mixed farming. Fewer farms are also producing milk compared to the 1980's. The stakeholders stated that root crops are produced as fodder for livestock and include swede and general brassicas. From the information provided by Stewart *et al*. (1990) and the stakeholders a species list was drawn up [Johnson & Marshall 2005].

Low Church Moss

Low Church Moss has been designated as a SSSI. The site is located in a shallow depression to the east of the River Ehen. Low Church Moss is a wetland site, which supports a variety of habitats including wet heath, acidic marshy grassland, tall fen and swamp, willow scrub, and a transition between poor fen and open water, which are very scarce on the West coast of Cumbria. The site is home to a diverse range of plant species and contains a rich invertebrate fauna including, in particular, Chrysomelid and Curculionid beetles, several species of which are scarce within Cumbria (English Nature, 2004). The species list that was been derived for the current study is included within Johnson & Marshall [2005].

3.2 Method

The soil activity concentrations were used along with CR values to determine how accurately the FASSET methodology can predict biota concentrations. The second part of the assessment involved the calculation of doses. The methodology states that if measured biota concentration data are available this should be used in preference to predicted concentrations calculated from the soil concentrations.

3.2.1 Drigg coast input data

There have been a number of studies conducted on semi-natural ecosystems along the coast of West Cumbria including the Drigg coast. Each study has addressed a different suite of radionuclides and a different group of organisms. The Drigg coast has two important habitat types: estuaries/saltmarsh and sand dunes. As these are very different in nature, it was important to obtain data specific to each habitat type. In the current assessment, soil and sediment data were selected using the most recent information available for each radionuclide in each habitat type. In the absence of data on the exact location being assessed, data from another location was used where it was thought to be comparable (e.g. given the lack of specific data for Drigg dunes themselves data from a dune system closer to Sellafield were used). The radionuclides for which data was available in each habitat were:

- ^{40}K, ^{106}Ru, ^{134}Cs, ^{137}Cs, ^{226}Ra, ^{228}Th, ^{230}Th, ^{238}Pu, $^{239+240}$Pu, ^{241}Am and ^{237}Np for saltmarshes;
- ^{40}K, ^{106}Ru, ^{134}Cs, ^{137}Cs, ^{210}Pb, ^{226}Ra, ^{228}Th, ^{230}Th, ^{235}U, ^{238}Pu, $^{239+240}$Pu and ^{241}Am for sand dunes.

The potassium, radium, lead and thorium radionuclides measured are not emitted in any significant quantity from Sellafield and may be considered to be natural in origin.

The concentrations for each radionuclide in the soil/sediment of each habitat are presented in Table 3.1. Similarly for each organism in the species list, data on the internal concentrations of radionuclides were selected using the most recent information available for each radionuclide in each organism where available. The organisms for which data were available in each habitat type were:

- field mouse (*Apodemus sylvaticus*), field vole (*Microtus agrestis*), common shrew (*Sorex araneus*), vegetation (many species), composite herb, lamb, ewe, black headed gull (egg), oystercatcher (egg), greylag geese (*Anser anser*), shelduck (*Tadorna tadorna*), wigeon (*Anas penelope*), mallard (*Anas platyrhynchos*), black-headed gull (*Larus ridibundus*), great black-backed gull (*Larus marinus*), lesser black-backed gull (*Larus fuscus*), curlew (*Numenius arquata*), bar-tailed godwit (*Limosa lapponica*) and oystercatchers (*Haematopus ostralegus*) for the saltmarsh;
- field mouse (*Apodemus sylvaticus*), field vole (*Microtus agrestis*), common shrew (*Sorex araneus*), red fescue (*Festuca rubra*), composite herb, marram grass (*Ammophila arenaria*), millipede, snail and worm for the sand dunes.

The concentrations of each radionuclide in the organisms used in the current assessment can be found in Johnson & Marshall [2005]. All of the data relating to birds (including eggs) was from a study conducted at Ravenglass (Drigg coast) between 1980 and 1984 [Lowe, 1991]. Soil/sediment data for ^{106}Ru and general vegetation data comes from a study of grazed and ungrazed marsh at Drigg coast [Horrill, 1984]. Soil/sediment data for ^{237}Np comes from samples from the Esk estuary [Hursthouse *et al*., 1991]. The remainder of the data relating to the saltmarshes and all of the data relating to the sand dunes comes from samples collected between 1993 and 1995 from sand dunes at Sellafield (to the north of Drigg coast) and saltmarsh in the Esk estuary at Drigg coast [Copplestone, 1996]. Use of the Sellafield sand dune data to represent the Drigg coast sand dune habitat is justified on the basis that its greater proximity to the Sellafield site means that radionuclide concentrations here are likely to be higher than those which would be found on the dunes at the Drigg coast site.

These data are clearly not ideal for the purpose of assessing the current status of the study sites, since the measurements were made some time ago. However, discharges of radioactivity from Sellafield reached a peak in the mid-1970s and have declined substantially and progressively through the 1980s and 1990s [Gray *et al.* 1995]. It is therefore reasonable to assume that an assessment based on data from the 1980s and 1990s is likely to over-estimate the present day radiation dose rates to organisms, and hence the likelihood of any detrimental effects.

Table 3.1: Radionuclide activity concentrations (Bq kg^{-1} fw) in soils/sediments used in the assessment of the effects of ionising radiation on biota at Drigg coast.

Radionuclide	Soil/sediment concentration	
	Saltmarsh	Sand dunes
^{40}K	539	384
^{106}Ru	1150	5.0
^{134}Cs	18.4	2.5
^{137}Cs	6590	341
^{210}Pb	-	25.4
^{226}Ra	172	22.3
^{227}Th	91500	9.4
^{228}Th	354	38.0
^{230}Th	1040	101
^{234}Th	20400	21.7
^{235}U	-	1.2
^{238}Pu	352	34.0
$^{239+240}Pu$	1860	165
^{241}Am	1600	177
^{237}Np	3.0	-

The data are also not ideal for the purpose of testing the radionuclide CR values provided by FASSET. An ideal comparison would require samples of environmental media and potentially impacted organisms taken from the same location during the same period of time; this is not always the case for the data we have been able to assemble for the sites.

A further difficulty with these sites, in relation to the FASSET radionuclide CR values, is that the underlying assumption of equilibrium almost certainly does not apply. The saltmarsh habitats are characterised by continuing deposition (and, in places, erosion) of fine sediments containing radionuclides; the coastal dunes are subject to continuing deposition of radionuclides largely in wind-borne seaspray.

3.2.2 Drigg coast dose assessment

The Drigg coast dose assessment has been carried out in accordance with Brown *et al.* [2003a][2]. The reference organism list from Brown *et al.*[3] is: woodlouse, earthworm, mouse, mole, weasel, snake, rabbit, red fox, roe deer, cattle, bird egg, herbivorous bird, carnivorous bird, herb, shrub and tree. This list does not represent the organisms found at Drigg coast and, as a result, a number of assumptions have been made so that an assessment could be carried out. The assumptions made are:

- Vole and field mouse are represented by the mouse reference organism;
- Shrew is represented by the weasel reference organism;
- All vegetation types assessed are represented by the herb reference organism;

[2]Note the assessors used Brown *et al*. [2003] as the definitive FASSET methodology and not Larsson *et al*. [2004] (eds).

[3]These are the geometries proposed as being representative of the reference organism list for dosimetry calculations, not all of those presented here have corresponding reference organisms listed for semi-natural ecosystems (i.e. birds). The allocation of some species to the geometries as listed here is not in agreement with the guidance as presented by Brown *et al*. [2003] (eds).

- Sheep are represented by the roe deer reference organism;
- Greylag goose, widgeon and mallard are represented by the herbivorous bird reference organism;
- Shell duck, black headed gull, great and lesser black backed gulls, curlew, bar-tailed godwit and curlew are represented by the carnivorous bird reference organism.
- These assumptions were applied throughout the assessment, to both the ecology through CR values applied and the dose to biota. No adequate reference organisms could be found to represent the reptiles, amphibians, butterflies, moths, and the lichen and bryophyte flora found at Drigg coast.

3.2.3 Agricultural area input data

Agricultural species

There is a good record of radionuclide concentrations in farm produce around the BNFL Sellafield site as a result of environmental monitoring programmes conducted by BNFL and regulatory agencies. Data has been collated from the British Nuclear Fuels Limited annual reports and the Radioactivity in Food and the Environment (RIFE) reports for 2000 to 2002 [BNFL, 2001; 2002; 2003; Food Standards Agency and Scottish Environment Protection Agency, 2001; 2002 & 2003]. Data have been averaged for the years and reported as average, minimum and maximum values in Bq kg^{-1} fw. However, these data have been recorded primarily as a means of assessing doses to the public as a result of food consumption pathways and therefore data has been derived only for edible fractions of food groups. The exact location at which samples have been taken is not available and, therefore monitoring data cannot be tied directly to individual farms.

Only the principal radionuclides released from the site have been monitored. Therefore, there are many radionuclides considered by FASSET for which there are no data available, for example concentrations of natural radionuclides. However, data are available for three radionuclides not considered by FASSET; ^{35}S, ^{60}Co, ^{125}Sb. The radionuclides for which observed data was available and are consistent with the FASSET methodology are:

- ^{3}H, ^{14}C, ^{90}Sr, ^{99}Tc, ^{106}Ru, ^{129}I, ^{137}Cs, ^{238}Pu, ^{241}Pu, ^{241}Am and very limited data for uranium (^{234}U, ^{235}U and ^{238}U).
- The biota for which data were available and hence examined in the current assessment were:
- cattle, sheep, pears, apples, elderberries, damsons, blackberries, potatoes, carrots, swedes, cabbages, beans and cereal crops.

The concentrations of each radionuclide in the organisms used in the current assessment can be found in Johnson & Marshall [2005]. As for the Drigg coast site, the data are far from ideal for testing the FASSET data on radionuclide CR values. In this case the data are from a reasonably consistent (and current) time period, but in many cases, reported concentrations are below the limit of detection. An assessment made using the limit of detection values will over-estimate the actual radionuclide concentration within the organism. As the inland agricultural and semi-natural ecosystems near Sellafield receive a continuing input of radionuclides from the ongoing emissions to atmosphere from the Sellafield site, they are not a simple system in which radionuclide concentrations in organisms are in equilibrium with those in soil.

Low Church Moss species

No radionuclide concentration data were found for the particular species present at Low Church Moss SSSI. Data were available however, for rabbits, deer, birds eggs (hens and ducks), various fruit trees, and blackberries within the 3km assessment zone around Sellafield and therefore the semi-natural assessment has been carried out on these species for the radionuclides reported above. Data has been collated from the BNFL annual reports and the RIFE reports for the years 2000 to 2002 [BNFL, 2001; 2002; 2003; Food Standards Agency and Scottish Environment Protection Agency, 2001; 2002 & 2003]. The concentrations of each radionuclide in the organisms used in the current assessment can be found in Johnson & Marshall [2005].

3.2.4 Agricultural dose assessment

As above, Brown *et al*. [2003a] was used for the assessment. Reference organisms as listed in Brown *et al*. are: potato, carrot, onion, lettuce, tomato, wheat, grapevine, orange, apple, olive, cow, sheep and pig[4]. The list clearly does not represent the agricultural practices around Sellafield. A number of assumptions have therefore been made so that an assessment of the crops and livestock farmed within the local area could be carried out. The assumptions made are:

- All fruit trees are represented by the apple reference organism;
- All root crops are represented by the potato and carrot reference organisms;
- All leafy green vegetables are represented by the lettuce reference organism;
- All fruit vegetables are represented by the tomato reference organism;
- All types of cereal are represented by the wheat reference organism.

The soil data used in the assessment was primarily taken from the 2000 RIFE report [Food Standards Agency and Scottish Environment Protection Agency, 2001] and supplemented, if no RIFE data was available, by soil core data taken in the 1990s [Jones *et al.*, 1996]. The soil data available for the area is limited as only the key radionuclides discharged by Sellafield are monitored routinely (Table 3.2).

Table 3.2: Radionuclide activity concentrations (Bq kg^{-1} fw) in soils used in the assessment of the effects of ionising radiation on biota on agricultural land around Sellafield.

Radionuclide	Soil concentration
^{90}Sr	3.7
^{106}Ru	2.70
^{137}Cs	74.0
^{234}U	10.7
^{235}U	0.39
^{238}U	10.1
^{238}Pu	0.47
$^{239+240}$Pu	9.6
^{241}Am	5.8

3.3 Results

The results for each of the two stages of the assessment are presented in Johnson & Marshall [2005]. Tables 3.3-3.5 present selected organisms as examples, the tables list the predicted organism concentrations for each radionuclide for which soil/sediment concentration data was available. For comparative purposes, observed organism concentrations are given where

[4]These are listed as 'representative species' in Brown *et al.* (eds).

available as the mean as well as the maximum and minimum values. For radionuclides where both observed and predicted values are available the ratio of the two values is given.

In some cases, the FASSET methodology does not provide any CR value from which the concentrations of radionuclides in a particular organism can be estimated. In these cases predictions were not made. The calculated radiation dose rates in these cases reflect only the external radiation dose delivered to the organism from radionuclides in the surrounding environment (see Tables 3.3-3.5).

The results of the dose rate calculation have been included as both unweighted and weighted doses. The weighted dose rates take account of the differing biological effectiveness of different types of ionising radiation. Following the recommendations of the FASSET project [Pröhl 2003] weighting factors of 3 are applied to low energy beta particle radiation, unity (i.e. no adjustment) to other beta particle radiation and gamma radiation, and 10 to alpha particle radiation[5]. Figures are provided for the total radiation dose rate and for the separate contributions from these radiation types.

The calculations demonstrate a gap in the FASSET data in that methods are provided to calculate radionuclide uptake from soil by plants, but no dose coefficients are provided which allow the contribution to radiation dose rates from internally incorporated radionuclides to be calculated. All of the radiation dose rates to plants therefore reflect only the external radiation dose rates from radionuclides in the surrounding environment.

3.3.1 Drigg coast

Ratios of predicted to calculated radionuclide concentrations in organisms for the saltmarshes

For small mammals on the saltmarshes (Table 3.3) the measured radionuclide concentrations in the organisms are largely considerably lower than those predicted from the underlying sediment concentrations. The explanation for this is almost certainly that these animals spend only a portion of their time on the contaminated areas which are subject to inundation, the majority of their time being spent in areas where contamination levels in soil are much lower. In terms of radionuclide uptake, they are clearly not in equilibrium with the contaminated sediments. Furthermore, the bioavailability of radionuclides deposited on the saltmarshes is comparatively low [Howard *et al*. 1996] (i.e. the FASSET CR values may be inappropriate for this ecosystem). For the (omnivorous) shrew, measured concentrations of plutonium and americium are similar to those for mice and voles whereas the predicted concentrations are much lower; this results from the use of lower CR values suggested by FASSET for carnivorous mammals.

For saltmarsh vegetation, the situation is reversed with the measured radionuclide concentrations in many instances being substantially higher than those predicted from the underlying sediment concentrations. Here, the explanation is that tidal inundations result in substantial amounts of contaminated sediment being directly deposited onto the vegetation surfaces [e.g. Howard *et al*. 1996]; this mechanism of soil to plant transfer is particular to saltmarshes and is not reflected in the FASSET CR values for either semi-natural or agricultural ecosystems.

For the lambs and ewes which graze the saltmarshes (e.g. Table 3.4), where comparison is possible measured concentrations of radionuclides are substantially lower than those predicted from the soil concentrations. Two factors are relevant here: firstly, the animals will graze saltmarsh pasture only for a portion of the time and, secondly, radionuclides bound to sediment (which represent the bulk of the contamination on saltmarsh vegetation) are not readily absorbed across the gut wall of grazing animals [Beresford *et al*. 2000].

[5] The weighting value of 10 is the example value Pröhl [2003] used; the recommendation was to use values 5, 10 and 50 to demonstrate the impact of RBE (eds).

For organisms which were assessed using both the semi-natural and agricultural assessment tools (lamb, ewe and vegetation), predicted organism concentrations differed (e.g. Tables 3.4). For sheep, the agricultural assessment predictions were closer to observed predictions than the semi-natural one which tended to over-predict organism concentrations. For vegetation the semi-natural assessment gave better predictions, whilst the agricultural assessment under-predicted organism concentrations.

Table 3.3: Predicted and observed activity concentrations of radionuclides and the resulting radiation dose rates in the Common Shrew (*Sorex araneus*) at Drigg coast salt marshes.

Nuclide	**Organism concentrations (Bq kg^{-1} wet weight)**				**Observed /predicted ratio**	**Dose rate (µGy h^{-1})**	
	Predicted	**Observed**				**weighted**	**unweighted**
		mean	**min**	**max**			
^{40}K		20	2.9	179		2.2x10^{-2}	2.2x10^{-2}
^{106}Ru	1380	2.1	8.3x10^{-2}	35	1.5x10^{-3}	0.47	0.47
^{134}Cs	91	0.37	3.3x10^{-3}	6.3	4.0x10^{-3}	5.6x10^{-3}	5.6x10^{-3}
^{137}Cs	32700	37	14	87	1.1x10^{-3}	0.72	0.72
^{226}Ra	6.1	7.9	0.12	124	1.3	1.2	0.17
^{228}Th	2	17	0.95	264	8.4	3.1	0.41
^{230}Th	5.7	28	0.41	466	4.8	0.75	7.6x10^{-2}
^{238}Pu	5.6x10^{-5}	0.42	0.19	0.83	7470	1.4x10^{-2}	1.4x10^{-3}
$^{239\&240}$Pu	3.0x10^{-4}	0.79	0.43	1.1	2640	2.4x10^{-2}	2.4x10^{-3}
^{241}Am	6.4x10^{-4}	1.0	0.39	1.5	1550	3.6x10^{-2}	7.8x10^{-3}
^{237}Np	2.8x10^{-4}					1.1x10^{-5}	1.1x10^{-5}
Total						**6.3**	**1.9**
Total low β						**3.13x10^{-4}**	**1.04x10^{-4}**
Total βγ						**1.4**	**1.4**
Total α						**4.9**	**0.49**

Table 3.4: Predicted and observed concentrations of radionuclides and the resulting radiation dose rates in lambs at Drigg coast salt marshes using both a semi-natural (S-n) and agricultural (Ag) assessment.

Nuclide	**Organism concentrations (Bq kg^{-1} wet weight)**			**Observed /predicted ratio**		**Dose rate (µGy h^{-1})**	
	Predicted		**Observed (mean)**				
	Ag	**S-n**		**Ag**	**S-n**	**weighted**	**unweighted**
^{40}K						3.6x10^{-3}	3.6x10^{-3}
^{106}Ru	12	265				8.9x10^{-2}	8.9x10^{-2}
^{134}Cs	0.13	34				1.1x10^{-3}	1.1x10^{-3}
^{137}Cs	138	12100	69	0.50	5.7x10^{-3}	0.17	0.17
^{226}Ra	0.18	7.1				1.2x10^{-2}	1.2x10^{-2}
^{228}Th	4.1x10^{-4}	2.7				2.4x10^{-2}	2.4x10^{-2}
^{230}Th	4.7x10^{-3}	8.0				8.7x10^{-6}	8.7x10^{-6}
^{238}Pu	8.8x10^{-3}	0.64				2.7x10^{-6}	2.7x10^{-6}
$^{239\&240}$Pu	4.8x10^{-2}	3.4				9.9x10^{-3}	9.9x10^{-4}
^{241}Am	7.4x10^{-3}	6.5	0.57	77	8.8x10^{-2}	1.9x10^{-2}	2.2x10^{-3}
^{237}Np	7.4x10^{-3}	4.5x10^{-3}				1.1x10^{-6}	1.1x10^{-6}
Total						**0.33**	**0.30**
Tot low β						**6.3x10^{-5}**	**2.1x10^{-5}**
Total βγ						**0.30**	**0.30**
Total α						**2.8x10^{-2}**	**2.8x10^{-3}**

For birds (e.g. Tables 3.5), the FASSET methodology provides no data on radionuclide transfer from soils or sediments[6] and no real comparison is possible between predicted and observed concentrations. FASSET does provide some indicative values on transfer from seawater to seabirds, although we have not attempted to use this data in the assessment.

From these comparisons it is clear that the saltmarsh ecosystems do not conform to the simple assumption of organisms in equilibrium with the surrounding environment, an implicit assumption in the FASSET CR values. Furthermore, whist saltmarshes are explicitly listed within the FASSET definition of semi-natural ecosystems they have distinctive radioecological characteristics, typified by low bioavailability; the data used to compile CR values within FASSET generally came from studies in ecosystems which could be expected to have higher transfers [Brown *et al*. 2003a]. In these circumstances, the importance of making measurements of actual concentrations is apparent.

Ratios of predicted to calculated radionuclide concentrations in organisms for the sand dunes

For small mammals in the sand dune site, ratios of observed to predicted radionuclide concentrations show little consistency with some lower than predicted and some higher; however most are within the same order of magnitude. The results for shrews bear the same relation to mice and voles as for the saltmarshes, with measured concentrations of plutonium and americium being similar but predicted concentrations very much lower due to the use of the CR value for carnivorous mammals.

For vegetation the majority of observed concentrations are substantially higher than the predicted values. This can be explained by the continuing deposition of radionuclides, largely carried in seaspray, directly onto the vegetation at the site. As for the saltmarshes, radionuclide concentrations in the vegetation are not in simple equilibrium with those in the underlying soil. For soil organisms, where comparison is possible, observed concentrations are quite close to predicted concentrations.

Calculated radiation dose rates

Radiation dose rates to small mammals in the saltmarsh system ranged from 1.7 to 6.1 µGy h^{-1} (unweighted) and 2.3 to 46 µGy h^{-1} (weighted). The majority of the dose originates from isotopes of thorium, which are of natural origin.[7] Radiation dose rates to vegetation were about 2 µGy h^{-1}, to lambs and ewes about 0.3 µGy h^{-1} and to birds between 0.4 and 1 µGy $^{-1}$.

Radiation dose rates to small mammals in the sand dunes were about 0.1 µGy h^{-1} (unweighted) and 0.5 µGy h^{-1} (weighted). Naturally occurring thorium isotopes again made a substantial contribution. Radiation dose rates to vegetation were about 0.1 µGy h^{-1}; those to soil fauna ranged from 0.2 to 1 µGy h^{-1} (unweighted) and 1 to 10 µGy h^{-1} (weighted), with gastropods (snails) receiving the highest doses.

When compared to the FRED database, the doses received by all organism groups were below the levels at which significant effects would begin to be seen (typically, about 100 µGy h^{-1}). Moreover, with the exception of voles on the saltmarsh, for which natural thorium isotopes (based on potential suspect measurements) make a large contribution to dose, there is a considerable margin between calculated radiation dose rates and those at which significant effects have been reported.

[6] This is because birds are not reference organisms in FASSET terrestrial ecosystems (eds).

[7] This result appears unusual and the assessors are currently checking back to the original source of the thorium measurements.

Table 3.5: Estimated radiation dose rates in greylag geese (*Anser anser*), shelduck (*Tadorna tadorna*), wigeon (*Anas penelope*), mallard (*Anas platyrhynchos*) and oystercatchers (*Haematopus ostralegus*) at Drigg coast salt marshes.+

Nuclide	Dose rate (μGy h^{-1})									
	Greylag goose		Shell duck		Wigeon		Mallard		Oyster catcher	
	weighted	unweighted	weighted	unweighted	weighted	unweighted	weighted	unweighted	weighted	unweighted
^{40}K	6.2x10^{-3}	6.2x10^{-3}	4.9x10^{-3}	4.9x10^{-3}	6.2x10^{-3}	6.2x10^{-3}	6.2x10^{-3}	6.2x10^{-3}	4.9x10^{-3}	4.9x10^{-3}
^{106}Ru	0.18	0.18	0.14	0.14	0.18	0.18	0.18	0.18	0.14	0.14
^{134}Cs	2.2x10^{-3}	2.2x10^{-3}	1.7x10^{-3}	1.7x10^{-3}	2.2x10^{-3}	2.2x10^{-3}	2.2x10^{-3}	2.2x10^{-3}	1.7x10^{-3}	1.7x10^{-3}
^{137}Cs	0.29	0.29	0.22	0.22	0.29	0.29	0.29	0.29	0.22	0.22
^{226}Ra	2.2x10^{-2}	2.2x10^{-2}	1.8x10^{-2}	1.8x10^{-2}	2.2x10^{-2}	2.2x10^{-2}	2.2x10^{-2}	2.2x10^{-2}	1.8x10^{-2}	1.8x10^{-2}
^{228}Th	3.8x10^{-2}	3.8x10^{-2}	3.1x10^{-2}	3.1x10^{-2}	3.8x10^{-2}	3.8x10^{-2}	3.8x10^{-2}	3.8x10^{-2}	3.1x10^{-2}	3.1x10^{-2}
^{230}Th	2.9x10^{-5}	2.9x10^{-5}	1.7x10^{-5}	1.7x10^{-5}	2.9x10^{-5}	2.9x10^{-5}	2.9x10^{-5}	2.9x10^{-5}	1.7x10^{-5}	1.7x10^{-5}
^{238}Pu	5.2x10^{-2}	5.2x10^{-3}	4.6x10^{-2}	4.6x10^{-3}	4.2x10^{-2}	4.2x10^{-3}	3.6x10^{-2}	3.6x10^{-3}	3.2x10^{-2}	3.2x10^{-3}
$^{239\&240}$Pu	2.0x10^{-1}	2.0x10^{-2}	0.20	2.0x10^{-2}	0.13	1.3x10^{-2}	0.10	1.0x10^{-2}	0.14	1.4x10^{-2}
^{241}Am	1.5x10^{-3}	1.5x10^{-3}	1.1x10^{-3}	1.1x10^{-3}	1.5x10^{-3}	1.5x10^{-3}	1.5x10^{-3}	1.5x10^{-3}	1.1x10^{-3}	1.1x10^{-3}
^{237}Np	3.9x10^{-6}	3.9x10^{-6}	3.0x10^{-6}	3.0x10^{-6}	3.9x10^{-6}	3.9x10^{-6}	3.9x10^{-6}	3.9x10^{-6}	3.0x10^{-6}	3.0x10^{-6}
Total	**0.78**	**0.56**	**0.66**	**0.44**	**0.70**	**0.55**	**0.67**	**0.55**	**0.59**	**0.44**
Total low β	**6.0x10^{-5}**	**2.0x10^{-5}**	**2.2x10^{-4}**	**7.3x10^{-5}**	**1.2x10^{-4}**	**3.9x10^{-5}**	**1.3x10^{-4}**	**4.4x10^{-5}**	**4.2x10^{-4}**	**1.4x10^{-4}**
Total βγ	**0.53**	**0.53**	**0.42**	**0.42**	**0.53**	**0.53**	**0.53**	**0.53**	**0.42**	**0.42**
Total α	**2.5x10^{-1}**	**2.5x10^{-2}**	**0.24**	**2.4x10^{-2}**	**0.17**	**1.7x10^{-2}**	**0.14**	**1.4x10^{-2}**	**0.17**	**1.7x10^{-2}**

+Where measured activity concentrations in a species were not available, dose rates are external only (no CR values being given for birds in Brown *et al*. [2003a]).

3.3.2 Agricultural land around Sellafield

Ratios of predicted to calculated radionuclide concentrations in agricultural produce

Comparison of observed and predicted concentrations is severely hampered by the low radionuclide concentrations; most of the measured values are below limits of detection.

For cattle and sheep, comparison can only be made for caesium and americium; predicted concentrations of caesium match observations quite well, whilst predicted concentrations for americium represent a substantial under-estimate of the observed values.

For fruit trees, comparison is possible for strontium, caesium, plutonium and americium; in all cases the predicted concentrations substantially under-estimate the observed values.

For root vegetables, comparison can be made for strontium, caesium, americium and plutonium; in all cases the predicted concentrations represent an under-estimate of the observed values, albeit to a somewhat lesser extent than was observed for fruit.

The remaining produce – leafy vegetables, beans, and cereals show a similar pattern with predicted concentrations for strontium, caesium, plutonium and americium under-estimating the measured values where comparisons can be made.

As for the sand dune system, these differences may reflect the effects of ongoing deposition from the atmosphere as a result of continuing emissions to the atmosphere from Sellafield.

Ratios of predicted to calculated radionuclide concentrations in semi-natural species

For mammals (deer, rabbits and hares) comparison can be made for concentrations of caesium, plutonium and americium[8]. In all cases, the predicted concentrations are an over-estimate of those measured.

Blackberries and elderberries have been assessed both as potential agricultural produce and as semi-natural species (assuming that the 'shrub' reference organism is appropriate). When assessed using semi-natural CR values, it is only possible to compare predicted and observed concentrations of strontium and caesium; predicted concentrations are now an over-estimate of the measured values.

The results for the semi-natural assessment draw attention to the higher transfer of radionuclides from soil to biota observed in semi-natural environments, and reflected in the FASSET methodology.

Calculated radiation dose rates

Calculated radiation dose rates to livestock are very low, at 0.003 to 0.005 μGy h^{-1}, as are those to mammals and bird's eggs in the semi-natural environment, at 0.007 to 0.01 μGy h^{-1}.

Calculation of radiation doses to crops and hedgerow fruit is hampered by the lack of dose coefficients in the FASSET reports which would allow the calculation of doses from internally incorporated radioactivity. Dose rates from radioactivity in the surrounding environment are assessed as about 0.01 μGy h^{-1}.

For all these organisms, the calculated radiation dose rates represent only a small addition to the dose rate which will arise from natural sources of radiation. The dose rates are well below those which, based on the FASSET database, might be expected to lead to any effects on the organisms.

[8]The reported data are for 'edible parts' (i.e. as likely to be eaten by humans). The FASSET CR-values estimate whole body activity concentrations and this discrepancy will contribute to the lack of agreement between observed and measured values for some radionuclides (eds).

3.4 Discussion

3.4.1 Differences in the observed and predicted concentrations

The data on radionuclide concentrations in soils and biota at our study sites are far from ideal for testing the radionuclide CR values recommended in the FASSET methodology. Nonetheless, there are clearly significant differences between concentrations in biota predicted by the FASSET CR values and those measured. Given some knowledge of the behaviour of radionuclides in the environment, it is possible to explain in most cases why the predicted and observed concentrations differ, and in what direction they differ.

It is clear that the FASSET CR values are not appropriate when a site is experiencing continuing inputs of radionuclides through deposition processes, as opposed to a situation where a site has a largely static burden of radionuclide contamination in soil. Even in the latter situation, wide variations in radionuclide transfer will occur according to factors such as soil type.

Although these points are well understood by those involved in the study of environmental radioactivity, the limitations of the FASSET CR values need to be clearly spelt out in future reports.

3.4.2 Dose rates to biota and the FRED database

The FRED database was found to be useful for looking up the range of effects of high doses found in laboratory studies and controlled field research. However, it is apparent that the availability of data is deficient in many respects, particularly in terms of information relevant to the radiation dose rates likely to occur as a consequence of radioactive waste management activities (less than 100 $\mu Gy\ h^{-1}$) and for many of the wildlife groups of interest.

3.4.3 End users perspective on the FASSET methodology

As first time end users of the FASSET assessment methodology, the authors of this report found Brown *et al*. [2003a] (the handbook of assessment of the exposure of biota to ionising radiation from radionuclides in the environment) to be unwieldy as a guide through the assessment methodology. Brown *et al*. [2003a] was useful as a reference work. Although Larsson *et al*. [2004] does provide a step by step guidance to the assessment this could be made more straightforward and comprehensive.

Particular issues which we note in terms of the structure of the data provided are:

- The lack of dose coefficients which would permit the calculation of doses to plants from internally incorporated radioactivity is an important omission;
- Attention should be paid to achieving consistency between the definition of reference organisms for tabulating radionuclide CR values and those for tabulating dose coefficients - this is a problem for the terrestrial ecosystems generally;
- Guidance on the selection of CR values generally (including the limitations discussed above) and on combining generic ecosystems such as agricultural, semi-natural and freshwater into a single assessment would be helpful.

3.4.4 Stakeholder input

Pre-assessment meeting

At a pre-assessment stakeholder meeting the principal concern of the stakeholders, with regard to how the assessment was to be tackled, was found to be what was perceived as an artificial division between agricultural and semi-natural habitats. Following these concerns, it was considered important to undertake both agricultural and semi-natural assessments on both study sites. In this way semi-natural species found on and around farmland and agricultural species found at Drigg coast were fully integrated into the assessment. Stakeholders also helped draw up a list of important species to be included in the assessment.

Concern was also expressed at the possibility that adverse findings from studies of this kind may lead to a form of 'blight' to the disadvantage of the landowner.

Post-assessment meeting

The stakeholders thought that the reference organisms chosen were inappropriate and wanted some explanation as to why the species were chosen. It was thought that it would be interesting to include indicator species for particular sites, for e.g. *Sabellaria* spp. (reef forming worm) for the Drigg Coast as this makes its home out of what is surrounding it and then lives there. Therefore if it was absent from a site or numbers were decreasing it would imply that there was something wrong that required further investigation. Also not being able to carry out an assessment for amphibians was seen as a massive gap in the current study as the Natterjack toad (*Bufo calamitais*) is a key species within the area and also a European protected species. The stakeholders found it difficult to comprehend how the reference organism species were chosen as they bared no resemblance to what species are considered as being endangered or needing protecting. Stakeholders thought that a good starting point for considering which species were important were the local Biodiversity Action Plans (see http://www.ukbap.org.uk/) for the areas in question.

In the report they received[9] stakeholders would have liked to have seen: (i) a distinction between the contribution of Sellafield to the overall dose compared with the contribution of background sources and the Chernobyl deposit; (ii) comparison with doses to same organisms in different parts of the UK or Europe; (iii) CR value used ('technical' stakeholders); (iv) clear information as to if doses were estimated using measured or predicted values; (v) an assessment by an independent body (in this case English Nature) on the general state of the environment in the area being assessed. Issues of how to address the uncertainty of FASSET predictions were raised, stakeholders pointing out that some predicted and observed activity concentrations in biota varied by 4-orders of magnitude. This gives a lack of credibility to conclusions made with regard to estimated dose rates and the potential for radiation to impact on the environment.

The question was raised as to whether the FASSET framework could have been used to address environmental protection problems associated with the Sellafield site in the past. Specifically the reduction in numbers of black-headed gulls (*Larus ridibundus*) in the 1980's [Lowe 1991]. Sellafield was suggested as the cause although there was no evidence to suggest that this was the case. However, the lack of CR values for birds was seen as a major problem in the FASSET methodology and something that should be addressed within ERICA.

The stakeholders felt that the number of data gaps meant the report was not comprehensive and that they therefore could not have confidence in it. However, overall, they felt that the progression of radiological protection to include the environment was very positive. To some extent the developmental nature of the field helped alleviate concerns about 'not being able to do everything'.

3.4.5 Recommendations

The principal recommendations from this case study for further development of the FASSET methodology are within ERICA:

- Dose coefficients should be provided to enable the calculation of dose rates to plants from internally incorporated radioactivity;

[9] Stakeholders received an expanded version of the case study report as presented here with basics of the methodology, radiological units etc. explained [Johnson & Marshall 2005].

- Attention should be paid to achieving consistency between the definition of reference organisms for tabulating radionuclide CR values and those for tabulating dose coefficients;
- Attention should be paid to filling, where possible, the gaps in availability of CR values. This assessment draws particular attention to the lack of data for amphibians and terrestrial birds;
- There should be clear advice on the selection and use of CR values, including advice on their limitations;
- Consideration should be given as to how actual ecosystems, in which particular species are of special interest, can best be represented by the limited number of 'reference organisms';
- Although we recognise a need to simplify the methodology on the basis of a limited number of 'generic' ecosystems, guidance on how to combine data to assess a real site would be useful – as in the example here of agricultural land with co-existing semi-natural components;
- The coverage of radionuclides provided by the FASSET project should be extended as necessary, including the addition of ^{35}S, ^{60}Co and ^{125}Sb;
- A concise, user friendly, guide through the assessment methodology would make the methodology easier to apply and would aid uptake of the methodology by a wider cross section of end users;
- Stakeholder inputs into assessments of this nature are beneficial in ensuring that all important components of an ecosystem are included within an assessment. However the methodology as a whole is difficult to explain to interested stakeholders who lack a specialist background in radioecology, and an 'intelligent lay persons guide' to the methodology, including its key assumptions and limitations, would be very helpful.

4 Loire River case study report

Between 1963 and 1999, 14 nuclear reactors were commissioned at 5 different sites on the Loire River and its tributaries. The Loire, with its estuary into the Atlantic Ocean, provides an opportunity to test the application of the FASSET framework to both fresh and brackish[10] waters.

4.1 The Loire River

The Loire River is considered one of the last 'wild' rivers in Europe. Several initiatives including local to international actions [Rivernet, 2004; MEDD, 2004a; 2004b] recognise and establish the important status of the Loire River and its valley, partially 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. 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 (Figure 4.1).

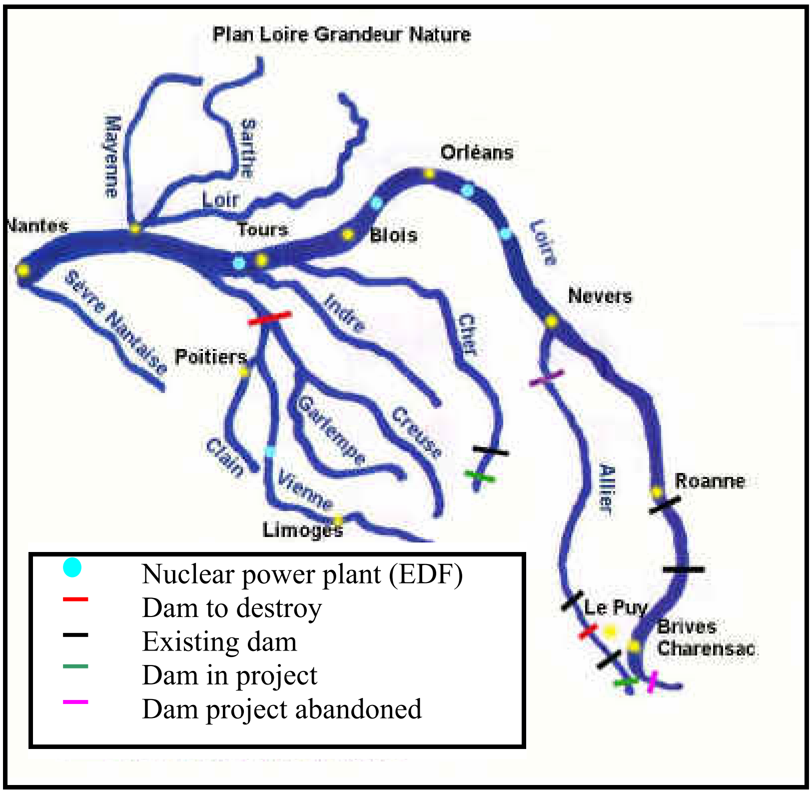


Figure 4.1: Location of the nuclear power plants on the Loire River (section of river shown is *circa* 350 km from estuary).

[10] Note in the context of the FASSET framework the definition of brackish waters is restricted to the Baltic Sea (eds).

4.1.1 The river

The following overview of the physical and biological characteristic of the Loire River is taken from [Beaugelin-Seiller 2004], and is for the section of the Loire 350 km downstream from the first power plant at Belleville and 350 km upstream from the estuary.

The annual average flow rates over this section of the river are from 335 $m^3 s^{-1}$ (upstream) to 850 $m^3 s^{-1}$ (downstream). The ranges of flow rate for the same locations are 50 and 140 $m^3 s^{-1}$ during low water and 2200 to 4400 $m^3 s^{-1}$ during high water respectively. The average slope of the stream bed is about $4x10^{-4}$ $m.m^{-1}$. The channel is normally from 270 m to 680 m wide in the study area (average of about 410 m) and 6 m deep. The average load of suspended matter is around 30 mg l^{-1}, although this value can double when the river is in flood.

This section of the river is comprised entirely of Natura 2000 sites. As such detailed information on habitats and species is available. In summary these show:

- High variability in flow creates many micro-habitats which often change (shores, meanders, islands, islets etc.);
- The fluvial dynamics determine the type of vegetation communities near the river bed, generating a high diversity; alluvial forests are present;
- The river is important for migratory fish (including salmon and lamprey);
- The river is important for more than 190 bird species during migration and for over-wintering;
- A number of nationally important species are found in the river;
- Important fish species are salmon (*Salmo salar*), shad (*Alosa alosa, Alosa fallax*), lamprey (*Lampetra planeri, Lampetra fluviatilis, Petromyzon marinus*), Cottus gobio (bullhead) and bitterling (*Rhodeus serviceus amarus*).

4.1.2 The estuary

The Loire estuary can be considered to comprise the last 120 km of the river. The following information comes largely from the Loire estuary group (http://www.loire-estuaire.org).

The slope of the river bed is now on average $7x10^{-5}$ $m\ m^{-1}$ and corresponds with a broadening of the channel width (390 to 485 m). The suspended matter load increases from 30 mg l^{-1} to 50 mg l^{-1} close to the sea under normal flow conditions. During flood events this can reach 200 mg l^{-1} or more. This area constitutes a sediment trap, which generates two specific formations, the 'silt plug' (turbidity from to 2 to 20 g l^{-1}) and the 'silt cream' (turbidity around 100 to 150 g l^{-1}, locally from 300-400 g l^{-1}). The nature and the position of these formations depend on the respective flows of the river and the sea. The change in water chemical composition is essentially marked by a reduction of carbonates and an increase in the following anions: Cl^-, SO_4^{2-}, NO_2^-, PO_4^{2-}.

The estuarine area is of high botanical interest, due to the diversity of the vegetation habitats including: mud flats and reed fields; "Loire" fields (old islands and shorelines); wetlands; swamps. Among the 700 plant species recorded, 14 are protected at local to European level (Loire Estuaire, 2002).

More than 250 bird species frequent the estuary during the year, and about 100 breed there. Water birds are the most numerous, with 30 000 to 40 000 individuals present during the winter. About 50 species are of a natural heritage interest. The other protected wildlife groups present in the estuary are mainly mammals (otter, bat), insects and amphibians. The same fish species as in the river itself are found in the estuary, especially shad and lamprey species.

4.1.3 Low-level liquid release of radioactivity

As the nuclear power plants are not allowed to release α emitters into the environment, the data available from Electricité de France (EDF) are for β and γ emitters (^{54}Mn, ^{58}Co, ^{60}Co, ^{110m}Ag, ^{63}Ni, ^{123m}Te, ^{124}Sb, ^{125}Sb, ^{131}I, ^{134}Cs, ^{137}Cs, ^{3}H and ^{14}C), for which the released activity is known by measurement, except for the ^{14}C which is estimated. Over the last decade, the annual released activity of gamma emitters has decreased year to year, stabilising in recent years (Figure 4.2).

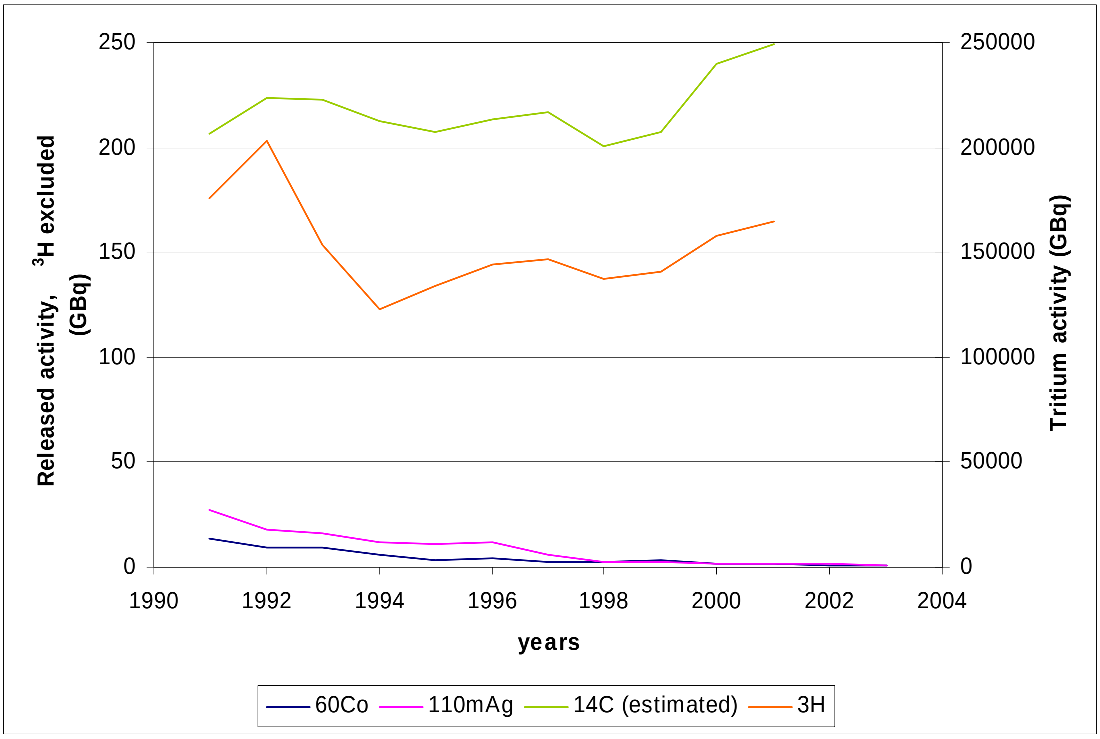


Figure 4.2: Total annual releases of selected radionuclides from all five nuclear power sites in the Loire basin.

Available radionuclide measurement data are issued from two main sources: monitoring programs around the power plants and studies at specific locations. The monitoring data is limited to the stretches of river close to the nuclear sites, and excludes the estuary. Figure 4.3 summarises the availability of monitoring data in biota and media for the area around the Chinon nuclear power plant. The data available from specific studies comes from the "Loire River radioecology program", initiated by EDF in 1998 which considered both the river and the estuary. In the river ^{3}H, ^{14}C, ^{58}Co, ^{60}Co, ^{110m}Ag, ^{134}Cs, ^{137}Cs, ^{54}Mn, ^{124}Sb and ^{131}I were measured in dissolved, particulate and sedimentary forms. In the estuary over 1000 measurements were made to determine ^{234}Th, ^{228}Th, ^{238}U, ^{226}Ra, ^{228}Ra, ^{137}Cs, ^{134}Cs, ^{60}Co, ^{58}Co, ^{110}Ag, ^{54}Mn, ^{7}Be, ^{131}I, ^{124}Sb, ^{210}Pb, ^{40}K, ^{241}Am, ^{14}C, ^{3}H, ^{90}Sr in water, bottom sediments and sediment cores. Results from other older studies are also available [Siclet *et al*. 2002; Ciffroy *et al*. 2003a; 2003b].

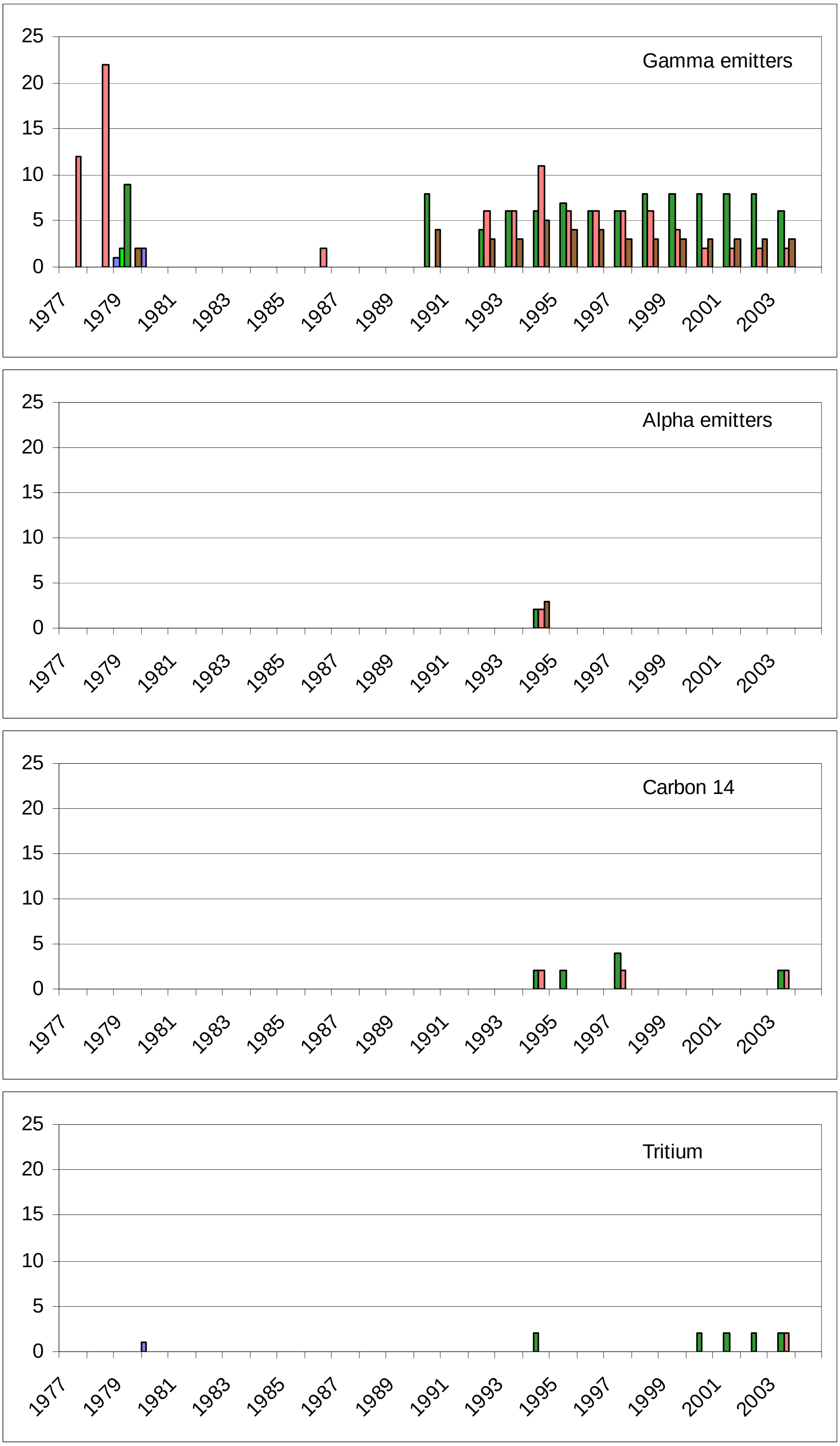


Figure 4.3: An example of the availability of monitoring data for biota and media (number of measurements per year for the area of Chinon power plant are plotted on the y-axis).

4.1.4 Selection of case study site

All the river sites are more or less equally documented in terms of available data. But two of them are of particular interest. The Belleville power plant is located upstream from the other nuclear installations releasing into the Loire River and the available data upstream of it may constitute a reference set for evaluating the radiological impact. The Chinon power plant is the last installation on the river and the river downstream of this site is therefore subjected to the cumulative discharges of all the nuclear industry in the Loire region. To meet the requirements of testing the applicability of the FASSET framework the Loire River downstream of the Chinon nuclear power plant and the estuary have been selected. The assessment has been carried out for 1999. The full detailed presentation of the case study is presented by Beaugelin-Seiller & Charton [2004].

4.2 Testing the FASSET framework

In applying the FASSET framework to the Loire River the assessors have followed the guidance within Larsson *et al.* [2004] in a step by step manner. Where gaps in the methodology are identified these are listed at the end of each assessment step.

4.2.1 Source characterisation

Identification of radionuclides

The radionuclides identified as being released by the nuclear power plants are the 13 for which routine monitoring data exist (see Table 4.1). Of the radionuclides released into the Loire River only seven are covered by the FASSET framework. As the framework cannot be tested for radionuclides it does not include the following assessment is restricted to these seven (Table 4.1) Although ^{3}H, ^{14}C, ^{63}Ni and ^{60}Co are within the FASSET framework[11], the level of information (in terms of k_d and CR values) varies considerably between the three FASSET aquatic ecosystems (Table 4.2). The FASSET framework does give some guidance for conducting assessments when there are data gaps [Brown *et al.* 2003a; Larsson *et al.* 2004]. For this assessment this guidance was followed such that it was assumed:

- k_d and CR values recommended for freshwater ecosystems (lakes)[12] may be appropriately applied to river ecosystems;
- k_d and CR values determined for marine waters may be appropriately applied to brackish waters;
- For aquatic systems, the highest available concentration factor for a specified radionuclide considering all reference organisms types should be compared with the k_d for that radionuclide; the larger number can be selected for the assessment when no CR value is available for the organism being assessed.

Applying these rules it was possible to cope with data gaps identified in Table 4.2 except for Co in all three aquatic ecosystems and ^{3}H and Ni in freshwater ecosystems. Therefore the radionuclides appropriate to the case study site which can be considered in the assessment are therefore following: ^{3}H, ^{14}C, ^{131}I and ^{134,137}Cs. Nickel-63 is excluded because it was not measured in releases before 2002.

[11] Note the list of radionuclides selected for FASSET does not include ^{60}Co (see Table 2-1 of Larsson *et al.* [2004], however, this radionuclide appears in some DCC look-up tables [Pröhl 2003] (eds).

[12] Freshwater ecosystem CR values presented in the FASSET framework were collated for lakes only (eds).

Gaps identified:

- Six radionuclides released into the Loire are not in the FASSET framework (^{54}Mn, ^{58}Co, ^{110m}Ag, ^{123m}Te, ^{124}Sb, ^{125}Sb);
- For two radionuclides (^{60}Co and ^{63}Ni) no CR or k_d values are presented in the FASSET look-up tables;
- Tritium and carbon-14 are only partially documented in terms of transfer parameters within the FASSET framework.

Table 4.1: Radionuclides released from all the nuclear power plants (NPPs) on the Loire River.

Radionuclide (Element group)	Radioisotope (half-life)	Radiation type	Release from Loire NPPs in 1999 (Bq)	Considered by FASSET
H (Ia)	^{3}H (12 y)	β^-	1.4×10^{14}	Yes
C (IVb)	^{14}C (5600 y)	β^-	2.1×10^{11} (estimate)	Yes
Ni (VIIIc)	^{63}Ni (100 y)	β^-	No measure (5.8×10^{8} in 2002)	Yes
Mn (VIIa)	^{54}Mn (0.9 y)	β^+,γ	3.4×10^{8}	No
Co (VIIIb)	^{58}Co (0.2 y)	β^+,γ	3.5×10^{9}	No
	^{60}Co (5.3 y)	β^-,γ	3.4×10^{9}	Aquatic DCCs only
Ag (Ib)	^{110m}Ag (0.7 y)	β^-, e^-,γ	2.8×10^{9}	No
Te (VIb)	^{123m}Te (0.3 y)	e^-,γ	No measurement (2.9×10^{8} in 2001)	No
Sb (Vb)	^{124}Sb (0.2 y)	β^-,γ	7.6×10^{9}	No
	^{125}Sb (2.7 y)	β^-,γ	No measurement (3.3×10^{8} in 2001)	No
I (VIIb, halogen)	^{131}I (0.02 y)	β^-,β^+,γ	1.13×10^{9}	Yes
Cs (Ia)	^{134}Cs (2.1 y)	β^-	1.77×10^{9}	Yes
	^{137}Cs (30 y)		4.86×10^{9}	Yes

4.2.2 Assessing potential risk

The source term

Of the radionuclides for which an assessment could be conducted (^{3}H, ^{14}C, ^{131}I, ^{134}Cs and ^{137}Cs), two groups were identified. Tritium and ^{14}C releases are 2 to 6 orders of magnitude higher than those of gamma-emitting radionuclides. These obviously have to be considered in the assessment. The others represent 1 to 5 % of the total α-activity released by the five nuclear power plants in 1999.

Under normal operating procedures the operation of the nuclear power plants can be considered to represent a situation of chronic exposure (i.e. releases are regular). In addition to liquid releases from the nuclear power plants atmospheric weapons testing, the Chernobyl accident and atmospheric releases from the nuclear power plants constitute (in decreasing order of importance) secondary contamination sources of the Loire watershed. These deposits will contribute to the contamination (largely by radiocaesium) of the river through run-off and soil erosion.

There is no knowledge available concerning the chemical speciation of radionuclides released into the Loire River.

Table 4.2: Available k_d and CR values for radionuclides of interest for aquatic ecosystems in the FASSET framework.

Radionuclide	Freshwaters		Brackish waters		Marine waters	
	k_d	CR	k_d	CR	k_d	CR
H	No data	No data	No data	No data	Yes	No data
C	Yes	No data	No data	Plankton, plants, fish, mammals and birds	Yes	Phytoplankton, vascular plants, zooplankton, molluscs, worms, crustaceans, fish, birds, mammals
Ni	No data	No data	No data	No data	Yes	Phytoplankton, macroalgae, vascular plants, zooplankton, molluscs, worms, crustaceans, fish, birds, mammals
Co	No data	No data	No data	No data	No data	No data
I	Yes	Amphibians, molluscs, crustaceans, fish, insect larvae, macrophytes, phytoplankton, zooplankton	No data	No data	Yes	Phytoplankton, macroalgae, vascular plants, zooplankton, molluscs, worms, crustaceans, fish, birds, mammals
Cs	Yes	Birds, fish, insect, macrophytes, molluscs, plankton	No data	Phytoplankton, zooplankton, macroalgae, vascular plants, molluscs, worms, crustaceans, insect larvae, fish, birds, mammals	Yes	Phytoplankton, macroalgae, vascular plants, zooplankton, molluscs, worms, crustaceans, fish, birds, mammals

Physical data

Radionuclide half-lives are presented in Table 4.1, only ^{131}I has a half-life of <1 y. Considering its 8 days half-life with regard to the lifespan of all the reference organisms, a yearly assessment of chronic exposure need not include ^{131}I (see Table 4-2 in Larsson *et al.* 2002a).[13]

The radionuclides for which the assessment can be conducted are γ- and/or β- emitters for which external contamination will be the most relevant exposure route.[14]

[13] Larsson *et al.* [2002a] give the following example – '*For chronic releases: The half life of the radionuclide or of its daughter radionuclides should be significant with respect to the lifespan of the reference organism, e.g. radionuclides with a $T_{1/2}$ > 30 days would be relevant to the phytoplankton bloom in a lake*'.

[14] This may appear to be a prejudgement by the assessors, they are however, following criteria within Table 2-2 of Larsson *et al.* [2004] (eds).

Environmental fate

In aquatic ecosystems, the solubility of each element can be characterised with regard to three of its salts: hydroxides, carbonates and chlorides. Calculations tend to indicate that all the existing compounds of radionuclides released would be very soluble, especially those of caesium.

The order of magnitude of k_d is known for each element in freshwater as well as in seawater (for which k_d values are higher)[15]. The lowest values of k_d are reported for ^{3}H in seawater and ^{14}C in fresh water (1 l kg^{-1} and 5 l kg^{-1} respectively); the k_d value proposed for ^{14}C in seawater is 10^3 l kg^{-1}. For iodine, values range from 100 l kg^{-1} in freshwater to 700 l kg^{-1} in seawater. Caesium has k_d values in the order of 10^3 to $4x10^3$ l kg^{-1}.

The assessment of the isotopic dilution (released radionuclide/(released radionuclide+stable isotope+known analogues)) requires data that are not directly accessible. However, their evaluation is possible based on literature values (Table 4.3).

Biological activity and potential chemical toxicity

Iodine is an essential element required for thyroid metabolism. Hydrogen and carbon, together with oxygen and nitrogen, are major constituents of all living matter. Whilst caesium has no known biological function its chemical analogue, potassium, is an essential element. None of the five radionuclides being assessed present a potential chemical toxicity risk at the level at which they are released.

Table 4.3: Information required under environmental fate criterion of FASSET source characterisation screening (see Larsson *et al*. [2004]).

Radioisotope			**Stable isotope**			**Chemical analogue**			**Isotopic dilution ratio**
Radio-nuclide	Specific activity[a] (Bq g^{-1})	Released mass (g)	Element	Concent-ration (g m^{-3})	Mass (g)	Analogue	Concent-ration (g m^{-3})	Mass	
^{3}H	$3.56x10^{14}$	0.12	H	$1x10^{5}$	$1.7x10^{15}$	None	0	0	$7.1x10^{-17}$
^{14}C	$1.65x10^{11}$	408.5	C	not known	not known	None	0	0	not determined
^{131}I	$4.59x10^{15}$	$1.3x10^{-8}$	I[b]	$5x10^{-3}$	$7.1x10^{7}$	Cl[d]?	$1.7x10^{1}$	$2.4x10^{11}$	$5.4x10^{-20}$
$^{134,137}Cs$	$3.22x10^{12}$	$2.5x10^{-5}$	Cs[c]	$5x10^{-5}$	$7.1x10^{5}$	K[d]	$3.8 x10^{0}$	$5.3x10^{10}$	$4.7x10^{-16}$

[a][JRC 1999];[b][Fuge & Johnson 1986]; [c][Coughtrey & Thorne 1983]; [d][CASTEUR database in Beaugelin Seiller 2004].

Conclusions – source term characterisation

Taking into account all of the data compiled in this section there is no reason to exclude any of the five radionuclides from the assessment.

The full source term characterisation and associated selection/screening criteria (Chapter 2 of Larsson *et al*. [2004]) requires a considerable amount of data not all of which are readily available. Due to the lack of required data assumptions had to be made, although all points could not be addressed. The conclusions reached from this source term characterisation step of the assessment are not decisive and result in all radionuclides being considered. Provision of default values for some of these parameters should be considered.

[15] Note the FASSET framework does not contain k_d values for all of these elements and ranges quoted here are not all consistent with Brown *et al*. [2003] (eds).

Gaps identified:

- physical parameters – radiation energies could have been included in the FASSET framework;
- environmental fate – provide default values for solubility constants, not all k_d values are provided in FASSET, provide list of chemical analogues[16], default values for environmental concentrations of stable/analogue elements;
- biological activity – provide default values for FASSET radionuclides;
- assessing the potential risk – guidance on interpretation and use of the collated data is needed.

4.2.3 Ecosystems and selection of reference organisms

The Loire River and its estuary represent the final receiving ecosystem (before the ocean) and are respectively analogous to the freshwater and brackish water ecosystems of the FASSET framework.

The reference organism categories should be sufficient to represent organisms at the case study sites based on the flora and fauna present in the river and estuary (see 4.1.1 and 4.1.2). However, concentration measurements are only available for two reference organisms: plants and fish (see 4.1.3). The compartments to be considered (for comparison of predicted and observed data in the next section) are therefore, water, sediment, plants and fish.

4.2.4 Transfer modelling

To assess transfers in the Loire River lake transfer parameters available in FASSET have been applied for ^{14}C, ^{131}I, ^{134}Cs and ^{137}Cs. No transfer values are available for ^{3}H in the FASSET framework.

The estuary can be described as a brackish environment. However, in FASSET, this category corresponds only to the Baltic Sea. Moreover, due to the lack of specific transfer data for brackish waters, it is recommended to apply marine transfer values. The estuary has also been assessed using freshwater transfer values. In this case, ^{3}H, ^{14}C, ^{131}I, ^{134}Cs and ^{137}Cs have been considered.

Loire River

No measurements of water concentration around the Chinon NPP are available for 1999. It is then necessary to estimate water concentrations; a number of methods being available to do this (Table 4.4). The yearly dilution approach assesses the mean radionuclide concentrations in water as the ratio between the release of radionuclide to the annual water volume ($1.5x10^{10}$ m^3 y^{-1}). The FASSET approach (suggested in Appendix 2 of Brown *et al*. [2003a]) is based on a dilution of the releases by a flow rate equal to a third of the average annual flow rate (472 m^3 s^{-1}). The CRESCENDO model was used by EDF for the Loire River radioecology programme.

Table 4.4: Release of radionuclides from the Loire River NPPs in 1999 and estimated water concentrations based on different approaches.

Radioisotope	Release (Bq)	Mean water concentration (Bq m^{-3})		
		Yearly dilution	FASSET approach	CRESCENDO
^{14}C	$2.07x10^{+11}$	14	42	7.80
^{131}I	$1.13x10^{+9}$	$7.50x10^{-2}$	0.23	$2.40x10^{-2}$
^{134}Cs	$1.77x10^{+9}$	0.12	0.36	$4.20x10^{-2}$
^{137}Cs	$4.96x10^{+9}$	0.32	0.98	$7.80x10^{-2}$

[16] Chemical analogues are tabulated within Strand *et al*. [2001] (eds).

Using the estimated water activity concentrations and the FASSET k_d values the corresponding sediment concentrations can be estimated (Table 4.5). For caesium, the assessment underestimates sediment concentrations by a factor 4 to 50 compared with the single measurement available for sediment. Several explanations for this can be suggested[17]:

1. The sediment is an accumulation compartment the concentration of which is largely influenced by the previous years water activity concentrations (including sources other than the Loire River NPPs).
2. The assessment does not include any background concentration, giving just the "added" concentration. Both are integrated in the measurements.
3. Available k_d values show a large variation, FASSET tends to apply the lowest ones (Table 4.6).
4. The simple assumption of equilibrium, corresponding to the use of k_d, has been much debated for rivers.

Whilst activity concentrations have been estimated for all freshwater reference organisms (and these are used in subsequent dose estimates) only those for which observed data are available are presented in this section.

Table 4.5: Estimated and measured sediment activity concentrations (Bq kg^{-1} dw) near the Chinon NPP.

Radioisotope	Modelled concentration from modelled water concentration by:			Measurement	
	Yearly dilution	FASSET approach	CRESCENDO	Upstream	Downstream
^{14}C	7.00×10^{-2}	0.21	3.90×10^{-2}	No data	No data
^{131}I	7.50×10^{-4}	2.30×10^{-3}	2.40×10^{-4}	Not detected	Not detected
^{134}Cs	0.12	0.36	4.20×10^{-2}	< 0.30	<0.29
^{137}Cs	0.32	0.98	7.80×10^{-2}	4.04±0.29	3.78±0.28

Table 4.6: A comparison of available k_d values for freshwater systems.

Radioisotope	k_d (m^3 kg^{-1})				
	FASSET	IAEA [1994; 2001]	IRSN compilation[a]	EDF data[b] Summer	Winter
^{14}C	2	5×10^{-3}	2	Not considered	
^{131}I	10^{-2}	10^{-2}	0 - 10	Not considered	
^{134}Cs	10	1	0.2 – 150	6.4	63
^{137}Cs	10	10	0.2 – 150	6.4	63

[a]the IRSN compilation consists in a set of sheets, each of them being devoted to a radionuclide and its behaviour in the environment [Garnier-Laplace & Roussel-Debet 2001; Colle & Mourlon 2002; Beaugelin-Seiller *et al.* 2005; Beaugelin-Seiller *et al.* in-press]; [b]values are specific of the Loire river [Ciffroy et al, 2001].

[17] It is also possible that the kd values are not appropriate for the prediction of total sediment activity concentrations – see Section 8 (eds).

The estimated concentrations in vascular plants, calculated from those estimated in the water, are lower than the available measured values by up to 2-3 orders of magnitude for ^{137}Cs and ^{131}I (Table 4.7). Some data are also available from other years for ^{14}C; the use of these data first requires their transformation into the units needed in FASSET, this transformation is subject to assumptions related to the carbon content of the organisms. The concentration of ^{14}C in plants was virtually constant from 1994 to 2003 ranging from 25 to 28 Bq kg^{-1} fw (geometric mean = 27 Bq kg^{-1} fw). The variation between predicted and measured concentrations may be explained by the accuracy of the method used to predict the water concentrations (as discussed above for sediment predictions). The FASSET CR values for aquatic vascular plants are generally higher than those in other data compilations available to the assessors (see IRSN compilation in Table 4.6 for data sources). Again this comparison puts doubt on the validity of assuming equilibrium. Note with the exception of predictions for ^{131}I, estimated activity concentrations are the same in plants as they are in sediments (compare Tables 4.5 and 4.7). This is because the FASSET CR and k_d values are the same in the case of radiocaesium. Whilst in the lack of a ^{14}C CR value for aquatic plants, FASSET guidance has been followed and the ^{14}C k_d used.

Some measurements of radionuclides in fish caught near the Chinon NPP are available; predictions are generally higher than measured values by up to two orders of magnitude (Table 4.8). Three measurements of ^{14}C in fish are available over the period 1994-2003, assuming predicted water activity concentration and FASSET CR values predicted values were *circa* three orders of magnitude lower than observed data. However, the ^{14}C CR value used here (in the lack of a CR value for fish the k_d value was assumed as suggested in Brown *et al.* [2003a]) is 3 orders of magnitude lower than that recommended by the IAEA [1994,2001][18].

No measurements of activity concentrations in reference organisms are available from the estuary; estimates have been made and these are used in the next sections to predict absorbed doses.

Table 4.7: A comparison of measured and predicted activity concentrations in aquatic plants (fw) near the Chinon NPP.

Radioisotope	Modelled concentration from modelled water concentration by:			Measurement	
	Yearly dilution	FASSET approach	CRESCENDO	Upstream (3 samples)	Downstream (5 samples)
^{14}C	7.00x10^{-2}	0.21	3.90x10^{-2}	No data	No data
^{131}I	1.50x10^{-2}	4.60x10^{-2}	4.80x10^{-3}	2.26±0.96^{+}	<2.42^{+}
^{134}Cs	0.12	0.36	4.20x10^{-2}	<0.06-<1.48	<0.09-<1.17
^{137}Cs	0.32	0.98	7.80x10^{-2}	< 2.96 0.59±0.07 0.71±0.06	3.83±0.77 0.95±0.12 1.51±0.12 0.67±0.07 1.30±0.12

$^{+}$n=1

[18] Note the FASSET guidance suggests using the highest available CR or k_d value – CR values are given for two references organisms in Brown *et al.* [2003] both of which are 3 or more orders of magnitude higher than the k_d value.

Table 4.8: A comparison of measured and predicted activity concentrations in fish near the Chinon NPP.

Radioisotope	Modelled concentration from modelled water concentration by: Yearly dilution	FASSET approach	CRESCENDO	Measurement: Upstream (3 samples)	Downstream (5 samples)
^{14}C	7.0×10^{-2}	0.21	3.9×10^{-2}	No data	No data
^{131}I	3.0×10^{-3}	9.2×10^{-3}	9.6×10^{-4}	Not detected	Not detected
^{134}Cs	1.2-1.4	3.6-4.3	0.42-0.50	<0.01-<0.03	<0.03-<0.04
^{137}Cs	3.2-3.8	9.8-12	0.78-0.94	0.05±0.01 0.10±0.02	0.09±0.02 0.13±0.03

Estuary

Radionuclide activity concentrations in water have been estimated using the yearly dilution approach. The annual flow of water in the Loire estuary is estimated to be *circa* 2.7×10^{10} m^3. Predictions compared well with available measurements (Table 4.9); according to FASSET recommendations measurements have been preferentially used in the following assessment. Estimated sediment activity concentrations have been estimated using measured water concentrations and both marine and freshwater k_d values from the FASSET framework (Table 4.10). There is good agreement between measured and predicted ^{14}C and $^{134,137}Cs$ activity concentrations. This differs from the comparison presented previously for the river, for which predictions were considerable underestimates (see Table 4.5). Values of k_d have been shown to vary with salinity in the Loire estuary. For example for Cs equilibrium k_d values varied from 7 l g^{-1} at 22 % salinity to 75 l g^{-1} at 0 % salinity [Ciffroy *et al*. 2003a].

Table 4.9: Releases of radioactivity from the Loire River NPPs in 1999, and estimated (using yearly dilution) and observed activity concentrations in estuarine waters.

Radioisotope	Releases (Bq)	Estimated average concentrations (Bq m^{-3})	Measured concentrations: Mean (Bq m^{-3})	Max (Bq m^{-3})
^{3}H	1.41×10^{14}	5210	8500	2.1×10^{4}
^{14}C	2.07×10^{11}	7.67	4.8*	6.0*
^{131}I	1.13×10^{9}	4.17×10^{-2}	4.7×10^{-2}	0.18
^{134}Cs	1.77×10^{9}	6.54×10^{-2}	3.0×10^{-2}	3.0×10^{-2}
^{137}Cs	4.96×10^{9}	0.18	0.18	0.61

*Inorganic dissolved carbon[19]

[19] Note the FASSET methodology presents ^{14}C transfer values for application with water organic carbon content – the impact of this on the assessment has not been assessed (eds).

Table 4.10: A comparison of observed and estimated activity concentrations in estuarine sediment.

Radioisotope	Estimated concentrations Using freshwater k_ds		Using marine k_ds		Measurement Estuary	
	Mean	Max	Mean	Max	Mean	Max
^{3}H	Not estimated		8.5	21	No data	
^{14}C	2.4×10^{-2}	4.8	4.8	6.0	11	14
^{131}I	4.7×10^{-4}	3.3×10^{-3}	3.3×10^{-3}	1.3×10^{-2}	Not detected	
^{134}Cs	6.5×10^{-2}	0.12	0.26	0.12	0.28	0.40
^{137}Cs	0.18	0.72	0.72	2.4	6.1	10

4.2.5 Dosimetry

As no default values were presented for occupancy factors in the FASSET methodology those of Copplestone *et al*. [2001] were used.

Loire River

Given the sparse data available, all activity concentrations used to estimate doses were modelled using FASSET k_d and CR values and estimated water concentrations (Table 4.11). The main contribution to total estimated unweighted dose rates is internal radiocaesium exposure, which contributes from 43 % (phytoplankton) to 73 % (pelagic fish) of the total dose rate. The highest estimated dose rate of 3.7×10^{-3} µGy h^{-1} was for mammals. As the radionuclides concerned have few low energy β-emission weighting the doses (using an RBE value of 3) has negligible influences on the estimated dose rates.

Estuary

For the estuary measured water activity concentrations were available. Dose rates were estimated assuming using both marine and freshwater transfer parameters from the FASSET framework (Tables 4.11 and 4.12).

Assuming freshwater CR and k_d values and mean water activity concentrations, internal dose rates from ^{137}Cs contribute the largest proportion of total dose, ranging from circa 55 % (gastropod) to 85 % (pelagic fish). The highest estimated dose rate of 5.7×10^{-4} µGy h^{-1} was for mammals. Assuming the highest measured water activity concentrations the contribution of ^{137}Cs incorporated into organisms to the total dose rate increases (to 95 % in the case of pelagic fish) and the total dose rate estimated for mammals is 1.7×10^{-3}µGy h^{-1}.

If marine transfer parameters are used internal doses from ^{14}C contribute the greatest to total dose rates (between 75 and 99 %). The highest dose rate of 7×10^{-3} µGy h^{-1} assuming average water concentrations is estimated for mammals and birds (this increases to 8.8 µGy h^{-1} using maximum water activity concentrations). The comparative importance of ^{14}C and ^{137}Cs using marine or freshwater transfer parameters is largely the result of ^{14}C transfer parameters for FASSET marine ecosystems being *circa* 4-5 orders of magnitude higher than those for freshwaters.

4.2.6 Effects analysis

Freshwater ecosystems

Total dose rates estimated for freshwater reference organism varied from 10^{-6} to 10^{-2} µGy h^{-1}. From the summaries of the FRED there is generally insufficient information on chronic low dose rates to draw any conclusions. The estimated dose rates to freshwater organisms in the Loire River and its estuary are at least five orders of magnitude lower than those at which effects have been reported (Table 4.11). The only exception being radiation-induced genetic damage that is proposed to potentially occur at all dose rates [Woodhead & Zinger 2003].

'Brackish waters'

If the estuary is assessed as a marine system dose rates to reference organisms do not exceed 3.1×10^{-2} µGy h^{-1} (Table 4.12). The FRED database dose not distinguish between marine and freshwater biota, however, estimated dose rates are of the 5 orders of magnitude lower than levels at which significant effects have been reported. As above the exception may be radiation-induced genetic mutation rate increases.

4.2.7 Uncertainties and interpretation

Most uncertainties have been discussed in each of the above steps of the assessment. These were largely associated with transfer modelling, especially the hypothesis of equilibrium and the lack of CR and/or k_d values for many radionuclides.

The estimation of dose rates integrates a number of uncertainties associated with the DCCs, no information is available with regard to this uncertainty and no point of comparison for the dose rates estimated is available.

The interpretation of the results is difficult, but as the highest estimated dose rates are 5 orders of magnitude below the threshold for statistical effects suggested within FASSET a certain amount of confidence can be placed in the conclusion that effects due to discharges from the Loire River NPPs are unlikely.

Table 4.11: Estimated dose rates for freshwater reference organisms in the Loire River and estuary downstream from the Chinon NPP in 1999.

Reference organism	Estimated total weighted dose rates (µGy h^{-1}) Modelled only		Measured & modelled		Lowest observed effect dose rates reported in RED (µGy h^{-1}) with comments from summary of Woodhead & Zinger [2003]
	River	Estuary	Estuary (mean)	Estuary (max)	
Bacteria	3.3×10^{-4}	6.0×10^{-5}	4.6×10^{-5}	1.3×10^{-4}	too few data to draw conclusions
Phytoplankton	9.1×10^{-6}	1.7×10^{-6}	1.4×10^{-6}	3.8×10^{-6}	too few data to draw conclusions
Zooplankton	4.0×10^{-4}	7.3×10^{-5}	6.6×10^{-5}	2.1×10^{-4}	440 (one single value)
Crustacean	1.5×10^{-3}	2.8×10^{-4}	2.4×10^{-4}	7.4×10^{-4}	10000 (no low chronic exposure)
Insect larvae	4.9×10^{-5}	9.0×10^{-6}	8.0×10^{-6}	2.5×10^{-5}	500 (no low chronic exposure)
Vascular plant	3.7×10^{-4}	6.7×10^{-5}	5.2×10^{-5}	1.5×10^{-4}	too few data to draw conclusions
Gastropod	3.1×10^{-4}	5.7×10^{-5}	4.7×10^{-5}	1.4×10^{-4}	10000 (too few data to draw conclusions)
Amphibian	2.4×10^{-3}	4.5×10^{-4}	3.9×10^{-4}	1.2×10^{-3}	too few data to draw conclusions
Bivalve mollusc	3.3×10^{-4}	6.1×10^{-5}	5.0×10^{-5}	1.5×10^{-4}	10000 (too few data to draw conclusions)
Pelagic fish	2.2×10^{-3}	4.1×10^{-4}	3.5×10^{-4}	1.1×10^{-3}	8.3 (exposure increases mutation rate)
Benthic fish	3.1×10^{-3}	5.7×10^{-4}	4.8×10^{-4}	1.5×10^{-3}	8.3 (exposure increases mutation rate)
Mammal	3.7×10^{-3}	6.8×10^{-4}	5.7×10^{-4}	1.7×10^{-3}	100 (for blood parameters & reproductive capacity)
Bird	1.1×10^{-3}	2.1×10^{-4}	1.8×10^{-4}	5.4×10^{-4}	10000 (too few data to draw conclusions)

Table 4.12: Estimated dose rates for marine reference organisms in the Loire estuary in 1999.

Reference organism	Estimated total weighted dose rates (µGy h^{-1})		Lowest observed effect dose rates reported in FRED (µGy h^{-1}) with comments from summary of Woodhead & Zinger [2003]
	mean	maximum	
Bacteria	3.2×10^{-4}	7.8×10^{-4}	too few data to draw conclusions
Phytoplankton	4.9×10^{-4}	7.1×10^{-4}	too few data to draw conclusions
Zooplankton	2.8×10^{-3}	3.6×10^{-3}	400
Mollusc	2.9×10^{-3}	3.8×10^{-3}	10000
Worm	3.0×10^{-3}	4.0×10^{-3}	1700 (too few data to draw conclusions)
Vascular plant	1.6×10^{-3}	2.3×10^{-3}	too few data to draw conclusions
Pelagic fish	2.8×10^{-3}	3.7×10^{-3}	8.3 (exposure increases mutation rate)
Bird	7.0×10^{-3}	8.9×10^{-3}	10000 (too few data to draw conclusions)
Macroalgae	1.6×10^{-3}	2.3×10^{-3}	too few data to draw conclusions
Benthic fish	2.9×10^{-3}	3.8×10^{-3}	8.3 (exposure increases mutation rate)
Crustacean	2.9×10^{-3}	3.8×10^{-3}	10000 (no low chronic exposure)
Mammal	7.0×10^{-3}	8.8×10^{-3}	100 (for blood parameters & reproductive capacity)

5 Oil and gas platforms case study report

5.1 Introduction

In recent years, the focus on discharges of natural radioactivity from non-nuclear industries has increased. The MARINA II study [European Commission 2003], found that non-nuclear industries, especially the phosphate and oil and gas industries, were responsible for a large fraction of the total discharged alpha activity in Northern European marine waters. The discharges of phosphogypsum have largely stopped since 2000. Therefore, the discharge of produced water from the oil and gas industry has become relatively more important.

During oil and gas exploitation, large volumes of water, referred to as produced water, formation water or oilfield brines, are co-produced with the oil and gas and discharged into the sea. This water consists of either naturally occurring formation water or a mixture of formation water and sea water, since sea water is injected in many of the reservoirs to maintain pressure. In 2003, about 135 million m^3 of produced water was discharged to the North Sea from platforms on the Norwegian continental shelf. Due to injection of sea water in the reservoirs, the water-to-oil ratio will increase over time and in 2003 this ratio was about 0.9 for the Norwegian oil and gas industry. The location of oil platforms and major oil fields in the Norwegian sector of the North Sea is shown in Figure 5.1.

One problem associated with produced water is that it may contain elevated amounts of natural radionuclides, mostly radium isotopes, which have been leached from the surrounding geological material in the reservoirs. A survey in 2003 of natural radioactivity in produced water from all 41 Norwegian platforms discharging produced water, found an average activity concentration of ^{226}Ra and ^{228}Ra of 3.3 Bq l^{-1} and 2.8 Bq l^{-1} respectively [NRPA 2005]. For ^{210}Pb, almost all samples were below the detection limit (about 0.5 to 1 Bq l^{-1}) and 12 samples analysed for ^{210}Po had < 10 mBq l^{-1}. Total activity discharged during 2003 was estimated to be 440 GBq for ^{226}Ra and 380 GBq for ^{228}Ra. In the MARINA II report, an activity concentration of 10 Bq l^{-1} for both ^{226}Ra and ^{228}Ra and a water-to-oil (volume) ratio equal to 3 were assumed, which together gives a discharge of these radioisotopes that is a factor of ten higher than was found in the 2003 survey [NRPA 2005].

5.2 Approach in applying FASSET methodology

The approach taken in this study has 2 main components:

1. Testing of the FASSET methodology with respect to the application of k_d and CR values. FASSET k_d values and CR values have been applied to measured activity concentrations in water to derive values for activity concentrations in sediments and reference flora. The predicted values have been compared with empirical data.
2. Derivation of a dose rate "excess" (i.e. above background) for selected reference organisms and comparison with background dose rates and dose estimates from other similar studies.

FASSET provides no guidance for the application of the environmental impact methodology to cases where technologically enhanced naturally occurring radioactive material (TENORM) forms the focus of the study. Therefore, the approach outlined here, whilst following the general methodology of FASSET, includes some assumptions and approaches which are not considered within FASSET. To address the impact of discharges of radionuclides from oil platforms specifically, we concentrated on activity concentration and dose rates in "excess", i.e. over that existing naturally. ERICA should consider whether this approach is tenable and make recommendations regarding TENORM environmental impact assessments.

Selection/screening criteria for initial source characterisation, as recommended in Larsson *et al*. [2004], have not been applied as the focus was on the testing of the assessments handbook [Brown *et al*. 2003a].

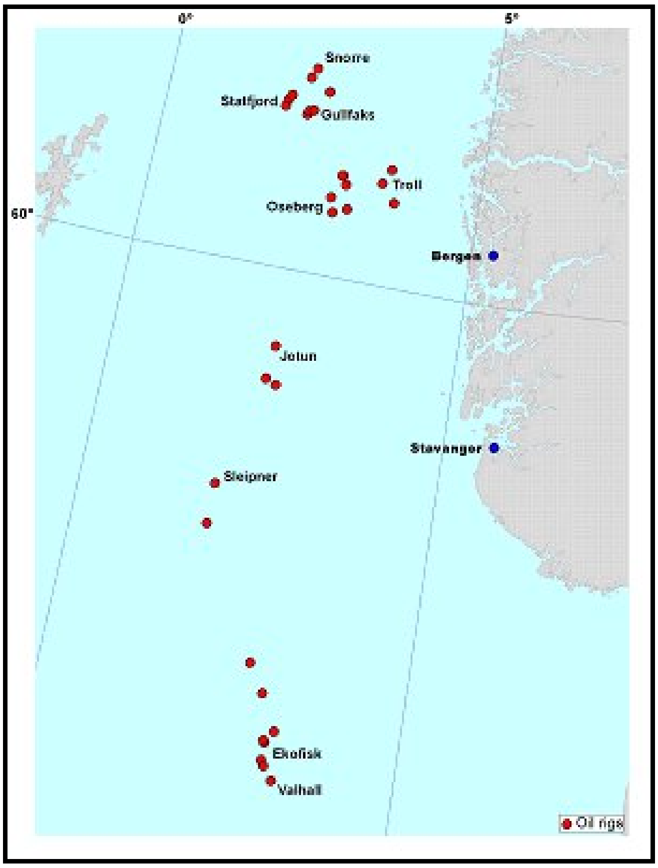


Figure 5.1: Overview of platforms discharging produced water and major fields on the Norwegian continental shelf.

5.3 Problem formulation

The flow diagram in Brown *et al*. [2003a] shows the stages in exposure assessment (Figure 1-2 in Brown *et al*. [2003a]). The initial stage of the assessment involves a problem formulation analysis where radionuclides and ecosystems are selected, spatial and temporal scales are defined and source terms and hazards are identified. Other themes that may require attention under problem formulation including *decision on level of simplification*, *biosphere and exposure pathways*, *object of protection*, *biological effects and data availability and requirements* were either unclear concerning the information requested or redundant within the context of this case study. For example, decisions have already been made within FASSET regarding the degree of simplification as equilibrium CR and k_d values are given.

The focus for this particular study is the naturally occurring radioactive material (NORM) discharged from oil and gas platforms in the Norwegian sector of the North Sea. The source term is the produced waters discharged as part of the routine operations at the stations. The produced waters can be enriched in radionuclides from the ^{232}Th and ^{238}U decay series, but most notably ^{226}Ra and ^{228}Ra. From some locations, elevated amounts of ^{210}Pb and ^{228}Th have also been reported [NRPA 2005].

Information on Ra activity concentrations in water was considered in the process of defining spatial scales appropriate for the implementation of the impact assessment. Results from model simulations (Figure 5.2) indicated that slightly elevated concentrations of Ra isotopes may be observed at distances of up to 100 km from the platforms but that significantly elevated concentrations, i.e. >10 % of naturally occurring ^{226}Ra activity concentrations, may occur in very localised areas typically much closer to the platforms (see Figure 5.2). This, and other information has been used to demarcate:

(i.) a near impact zone – taken to be an area within a few 10s of km from the oil platforms;

(ii.) a regional background area taken simply to be the North Sea as a whole [UK oil and gas production contributes to TENORM levels in this area];

(iii.) a "true" background – taken to encompass European or global sea areas external to the case study area. In view of the numerous other sites in Europe discharging enhanced levels of NORM to the sea, such as the phosphogypsum company, Allbright and Wilson in Cumbria, UK [European Commission 2003, Betti *et al.* 2004], care should be taken in interpreting such background activity concentrations

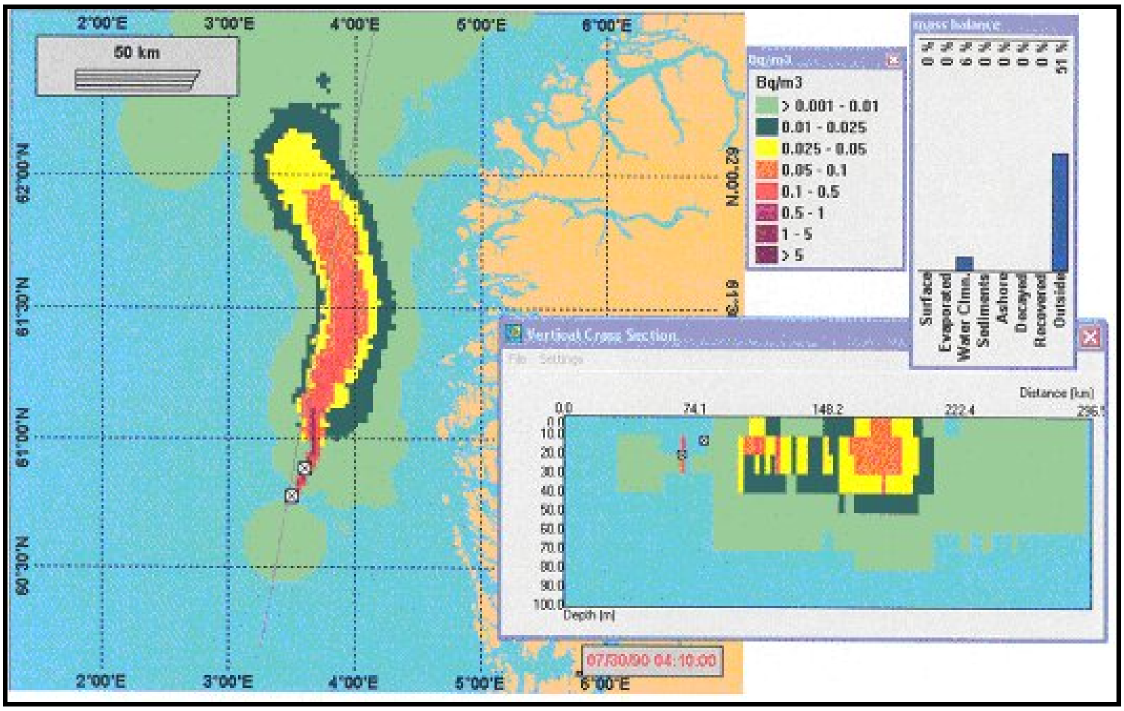


Figure 5.2: Results from dispersion modelling of ^{226}Ra. The figure shows the concentration field of ^{226}Ra after 7 months of continuous discharge of produced water from the Troll B and C platforms.

FASSET provides no advice on appropriate temporal scales or how to average data temporally. Guidance is given in Copplestone *et al.* [2001]: "concentrations should be averaged temporally over a period of at least one-year". In the survey of radium in produced water from Norwegian platforms in 2003, some platforms showed a relatively large temporal variation in the activity concentration of radium during a five month period [NRPA 2005]. In view of the fluctuating input of production water to some areas, it may be inappropriate to consider long-term averages since organisms may be exposed to an acute flush of relatively high loads of contaminants. Nevertheless, from a pragmatic perspective, the process of temporal averaging is driven by data availability and so, in this case, a limited number of empirical/monitoring data were assumed to represent longer term averages.

The issue of which temporal and spatial scales to consider may also be guided to some extent by the particular biota type considered. For example, some species of fish are highly mobile such as the Arcto-Norwegian cod (*Gadus morhua*) which undertakes spawning migrations of approximately 1500 nautical miles [Sazykina 1998]. The spatial scale over which activity concentrations should be collated is large and would require knowledge of the migration route. In contrast, the spatial scale considered for sessile biota, such as adult Blue mollusc (M*ytilus edulis*), might be limited to the local area in close proximity to the source. Life history data may provide useful insight in these cases although for this case study we have not considered this.

The problem formulation stage of the assessment requires refinement. Each component forming the basis for this stage of the methodology should be evaluated in terms of its usefulness to the overall assessment and clear definition and guidelines provided where the component is deemed important enough to consider.

5.4 Exposure pathways analyses

The data on radium in produced water obtained in the 2003 survey [NRPA 2005], together with discharge volumes for 2003 have been used as input data to model the dispersion of radium in the North Sea using the three-dimensional DREAM model [Reed *et al.* 1995; 2001]. One simplification assumed in the calculation is that the produced water discharged has the same temperature and salinity as the surrounding seawater. It is also assumed that all discharged radium is in a soluble form. The ocean current data used were generated from the ECOM-3D model developed by the Norwegian Meteorological Institute [Engedahl & Martinsen 1996]. The data used are from a simulation carried out for the North Sea in 1990. The currents are depth and time variable throughout the whole year, and the horizontal resolution is 20 km.

Two scenarios have been modelled:

1) One year of continuous discharge of produced water from the Troll B and C platforms. Results presented in Figure 5.2 show ^{226}Ra activity concentrations in July.
2) One year of continuous discharge of produced water from all Norwegian production platforms in the North Sea (starting in January). Results presented (Figure 5.3 and 5.4) show ^{226}Ra activity concentrations in the seawater in July (7 months elapsed) and December. Similar modelled data are available for ^{228}Ra.

Figures 5.2 to 5.4 show the concentration range for ^{226}Ra at the depth with the highest radium concentration in Bq m^{-3}. A vertical cross section is also included. In the maps of the North Sea area, a cross section from the Ekofisk area through the Troll area is shown. In the maps of the Troll area, a cross-section to the Troll area and northwards is shown. The mass balance shows the percent of the total annual discharged activity that has been dispersed. The column marked “outside” shows the percentage that is dispersed outside the area shown in the map.

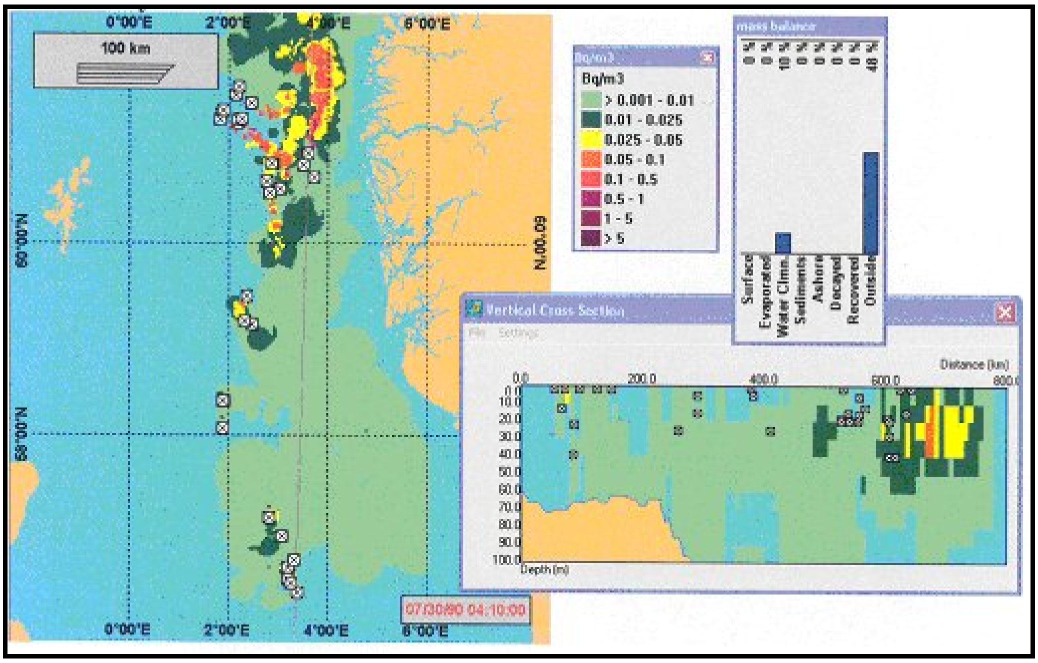


Figure 5.3 Results from dispersion modelling of ^{226}Ra. The figure shows the concentration field of ^{226}Ra after 7 months of continuous discharge of produced water from all Norwegian platforms in the North Sea.

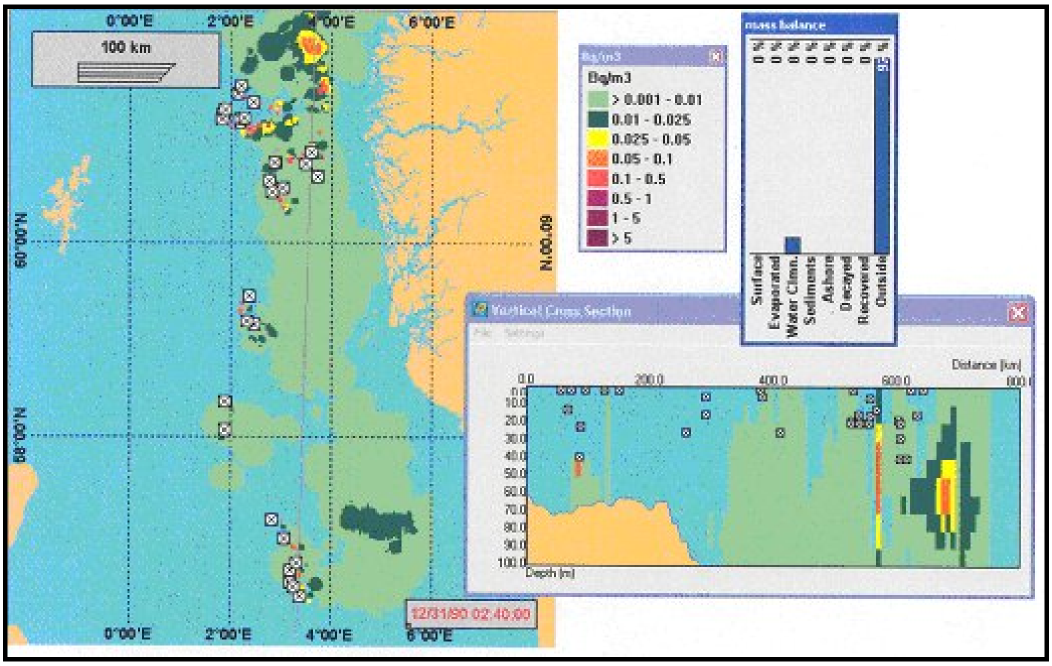


Figure 5.4. Results from dispersion modelling of ^{226}Ra. The figure shows the concentration field of ^{226}Ra after 12 months of continuous discharge of produced water from all Norwegian platforms in the North Sea.

5.4.1 Activity concentrations in seawater

Few relevant data have been identified and collated in this case study reflecting the limited amount of information that is currently available for oil and gas production facilities and their regional environment. An overview of empirical data on activity concentrations in seawater is presented in Table 5.1. Ancillary information has been provided on year of sampling, depth of measurement and the form of the sample (filtered, total etc.).

Table 5.1 : Activity concentrations (Bq m^{-3}) of radionuclides in seawater (empirical data).

Radionuclide	Impact zone mean concentration (range) and time period	Area North Sea mean concentration (range) and time period	Background range in concentrations
Ra-226	1.6 (1.4-2.5)[a] Bq m^{-3} n=8; 2002	(1.5-8.5)[b] Bq m^{-3} n=3; pre 1990	1.3-3.1[d] Bq m^{-3}
Ra-228	n/d	1[c] Bq m^{-3} pre 1990	0.04-3.7[d] Bq m^{-3}
Po-210	n/d	0.8[e] Bq m^{-3} pre 1980	0.19-3.7[d] Bq m^{-3}
Pb-210	n/d	0.72[e] Bq m^{-3} pre 1980	0.4-5.0[d] Bq m^{-3}

[a]Surface waters in proximity to platforms [NRPA,2005]; [b]IAEA [1990]; [c]van der Heijde *et al.*, [1990] [d]IAEA [1988]); [e]Spencer et al. [1980]; n/d = No data

Deriving the excess component of TENORM radionuclides in seawater

The data from the impact zone and North Sea reflect both natural and excess contributions for the radionuclides considered. Ra-226 water phase activity concentrations in the impact zone fall within the range of concentrations observed naturally so the input from the platforms was not easily discernable from background at the time of sampling. The only clue that suggests an anthropogenic influence in this area relates to the ^{226}Ra activity concentration profiles with depth. Under natural conditions ^{226}Ra is often depleted in surface waters [Cherry & Shannon 1974] whereas for the samples taken close to the oil and gas platforms this trend is reversed with activity concentrations of ^{226}Ra over 40 % lower in deep waters (approx 100 m) compared to surface waters around some platforms.

It has been difficult to derive excess activity concentrations from the empirical data alone. For this reason, modelling data were used to supplement the data to derive a ^{226}Ra excess. Analyses of model output show that activity concentrations of Ra isotopes in water can occasionally attain levels of 1 Bq m^{-3}. This conservative value has therefore been used to establish as the concentration of excess ^{226}Ra. To perform a robust dose calculation, the excess component of the activity concentration also requires calculation for ^{228}Ra, ^{210}Pb and ^{210}Po. The excess of ^{210}Pb and ^{210}Po has not been derived directly through the application of model runs but through assumptions related to the relationship of these radionuclides to ^{226}Ra. Ra-226 is often depleted in surface water owing to biological accumulation and ^{228}Ra concentrations decrease with increasing distance from the sediment water interface [IAEA 1988]. A fraction of gaseous ^{222}Rn generated from the decay of non-volatile ^{226}Ra diffuses to the atmosphere where it decays, through a series of short-lived radionuclides to ^{210}Pb and returns to earth by precipitation or dry deposition. Aerosol ^{210}Pb falling directly on surface waters is subsequently removed to sediments where it is eventually buried by accumulations. The final distribution of excess ^{210}Pb in sediments above that in secular equilibrium with *in situ* ^{226}Ra is governed by the rates of sedimentation, mixing (though physical disturbance and bioturbation) and radioactive decay [Hamilton *et al.* 1994]. Pb-210 decays to ^{210}Po which is, despite its short half-life of 138 days, frequently out of equilibrium with ^{210}Pb. Even for a purely theoretical consideration of 1 kBq ^{226}Ra discharged into a closed system with an assumed initial ^{210}Pb concentration of 0, ^{210}Pb reaches only transient equilibrium with ^{226}Ra following an elapsed period exceeding 200 years (Figure 5.5).

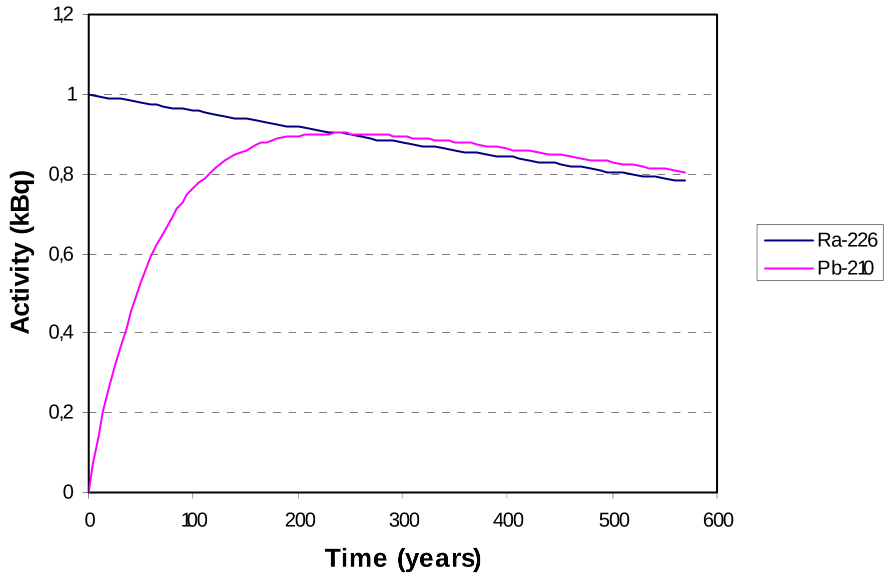


Figure 5.5: Ingrowth of ^{210}Pb following instantaneous release of ^{226}Ra to a closed system.

In view of the complexity of the behaviour of the natural decay series radionuclides in marine environments, estimates of the excess component of ^{210}Pb and ^{210}Po have been made based on the quotient of the average activity concentration of ^{226}Ra and the limit of detection level for ^{210}Pb (not all ^{210}Pb activity concentrations in the 2003 study were below detection limits) in produced waters from the data presented in NRPA [2005]. It has thereafter been assumed that ^{210}Pb and ^{210}Po are in secular equilibrium. A ^{226}Ra:^{210}Pb ratio of 1:0.15 has thus been obtained. A summary of the "excess" activity concentrations in seawater used in this case study are given later in Table 5.2.

Earlier work indicates that disequilibrium normally exists between ^{226}Ra and it's gaseous daughter, ^{222}Rn. Near surface water tends to be depleted in ^{222}Rn with $^{222}Rn/^{226}Ra$ activity ratios of around 0.5 whereas bottom water can be significantly enhanced in ^{222}Rn owing to the upward diffusion of this radionuclide from the seabed with activity ratios of around 8 [Cherry & Shannon 1974]. In dosimetric calculations [Pröhl 2003] in FASSET, secular equilibrium between $^{226}Ra \rightarrow ^{214}Po$ is assumed, the DCC of all daughter nuclides are incorporated within that for ^{226}Ra. FASSET (see Table 4-2 in Pröhl [2003]) is a little unclear regarding this point – DCC is derived for ^{226}Ra "*including all daughter products*". Empirical evidence suggests that this assumption of secular equilibrium as adopted in FASSET is inaccurate for marine environments. This might be addressed by either splitting the DCCs into more discrete components or by demonstrating that the observed disequilibrium makes little difference to calculated doses (using the aggregated DCCs) for specified cases. Nonetheless, for this particular study the original DCCs have been applied.

5.4.2 Activity concentrations in sediments

The FASSET methodology recommends the use of distribution coefficients to derive sediment concentrations from water activity concentrations. However, no explicit guidance is given specifying that filtered seawater activity concentrations are normally required as input to these calculations. Furthermore, the units for k_d have not been explicitly provided in FASSET (see Table 4-7 of Brown *et al.* [2003a]).

The FASSET k_d values were taken from IAEA [2004] where the unit of k_d is dimensionless or $l\ kg^{-1}$. In addition, for the impact zone activity concentrations are for total (unfiltered) water samples. Because of the lack of more detailed information, it has been conservatively assumed that these total concentrations are equivalent to filtered ^{226}Ra in the water samples. In relation to the derivation of the activity concentration excess, the model runs are not explicit with regard to whether the activity concentrations are for filtered or unfiltered water. Again, it has been conservatively assumed that all model are filtered water.

Chemical reactions between ions in seawater and in the discharged produced water may also occur, leading to the formation of insoluble compounds. The formation of $BaSO_4$ from Ba^{2+} ions in the produced water and SO_4^{2-} ions in the seawater is important for radium. If $BaSO_4$ is formed near the discharge point, some of the Ra^{2+} ions may co-precipitate and be removed from solution and transferred to the sediments [Hamilton *et al.* 1991]. For produced water with a high Ba^{2+} concentration, the fraction of radium that co-precipitates with $BaSO_4$, may be significant. Jerez Vegueria *et al.* [2002] investigated sediments and seawater close to two offshore platforms (discharging about 30 MBq d^{-1} and 41 MBq d^{-1}) at the Bacia de Campos oilfield (Brazil) and concluded that both sediments and seawater showed normal background levels, even at the closest sampling distance of 250 m from the platforms.

Testing k_d values

Although no sediment samples have been collected in proximity to the platforms we can test whether sediment concentrations, predicted from the application of FASSET guidance, fall within the range of natural (background) variability given that activity concentrations in seawater fall within the range of natural variability.

Applying the recommended k_d of 2000 for Ra from Brown *et al*. [2003a] (see p. 72) to the maximum activity concentration in water (empirical data) taken in proximity to the platforms, i.e. 2.5 mBq l^{-1} (from Table 5.1), we attain a sediment activity concentration of 5 Bq kg^{-1} dw. This activity concentration specifically represents the exchangeable phase component of the radionuclide in the sediment. The relationship between the activity concentration in filtered seawater and that in the "total" sediment cannot therefore be directly derived from the application of a k_d in this case. The FASSET methodology may be improved by providing the relevant information for this calculation process thereby avoiding the need to access the original IAEA reference. Activity concentrations of ^{226}Ra in sediment were presented in Brown *et al*. [2004]. Typical values are about 30 Bq kg^{-1} dw (ranging from 15 - 60 Bq kg^{-1} dw). These values are somewhat higher than the activity concentration predicted using FASSET methodology. The discrepancy might be explained by the fact that the sediment has a substantial non-exchangeable component associated with it of (mainly) natural origin. Although this comparison cannot be termed a validation of the methodology, it shows that this assessment stage is producing sensible values for ^{226}Ra in sediments.

A similar approach has been adopted in testing the application of FASSET k_d values for ^{210}Po and ^{210}Pb. Empirical data for the North Sea provide values of 0.8 mBq l^{-1} and 0.72 mBq l^{-1} for ^{210}Po and ^{210}Pb respectively (these values are pre 1980s). It has been assumed that these values relate to filtered seawater. Sediment activity concentrations of 16 000 Bq kg^{-1} (dw) ^{210}Po and 72 Bq kg^{-1} (dw) ^{210}Pb are predicted. The prediction for ^{210}Po in sediment is several orders of magnitude higher than measured values typical of European coastal environments [Brown *et al*. 2004; McDonald *et al*. 1991]. The predicted value for ^{210}Pb falls within the range of activity concentrations that might be expected under natural conditions.[20]

Deriving the technological enhanced ('excess') component of radionuclides in sediment

For ^{226}Ra and activity ^{228}Ra concentrations we have assume 1 mBq l^{-1} as the excess activity concentration in seawater predicting an (exchangeable) sediment concentration of 2 Bq kg^{-1}

[20] Grøttheim [1999] provides activity concentrations of ^{210}Po from various northern European seas. Assuming ^{210}Pb activity concentrations are of a similar magnitude (close to equilibrium conditions might be expected between ^{210}Pb and ^{210}Po in sediments), it can be seen that activity concentrations range between 25 and several hundred Bq kg^{-1} dw.

dw. The excess component of derived water activity concentrations, have also been used to calculate sediment activity concentrations for ^{210}Pb and ^{210}Po. An overview of the data set used in subsequent dosimetric calculations is provided in Table 5.2

Table 5.2: "Excess" activity concentrations in the near impact zone of ^{226}Ra, ^{228}Ra, ^{210}Pb and ^{210}Po in water and sediments derived from anthropogenic activity alone

Radionuclide	Seawater activity concentration* (mBq l^{-1})	Sediment activity concentration (Bq kg^{-1} dw)**
^{226}Ra	1	2
^{228}Ra	1	2
^{210}Pb	0.15	15
^{210}Po	0.15	3000

* assumed filtered
** exchangeable activity assumed to represent the total sediment-associated activity arising from the discharge

5.4.3 Selection of reference biota and representative flora and fauna

The information collated during the initial stage of the assessment is intended to allow an informed decision to be made in the selection of reference organisms. The FASSET marine organism suite can be adopted directly or modified accordingly. In this case study we are dealing with three relatively particle-reactive radionuclides that are not biomagnified[21] to any great extent in marine food-chains. Furthermore, because these radionuclide decay predominantly by emission of either α– or β–radiation, the main exposure pathway will be due to internal irradiation and not to external irradiation from the habitat. An exception to this generalisation may be for phytoplankton for which a large component of the exposure load is likely to arise from radionuclides adsorbed onto the cell surfaces. In a closed, equilibrated system, the highest Ra concentrations might be expected in phytoplankton, the highest Po concentrations in phytoplankton, crustaceans (pelagic and benthic)[22] and molluscs and highest Pb concentrations in crustaceans and molluscs. Fish might also be considered a suitable reference biota type due to its high CR values for Po [IAEA, 2004] and relatively high radiosensitivity [UNSCEAR 1996].

Macroalgae and vascular plants may be justifiably deleted from the reference organism suite for this case study due to their relatively low uptake of all radionuclides considered and their high radioresistance. Mammals and wading birds constitute the most radiosensitive faunal groups and by this virtue alone might be justifiably included in the reference organism suite despite their generally low uptake of the radionuclides considered. However, because it was anticipated that little empirical information would be available for these organism types with which to compare values predicted from the FASSET methodology, sea mammals and seabirds were excluded from further consideration.

On inspection, therefore, it appears that the subset of the FASSET suite of reference organisms suitable for application in this marine case study includes phytoplankton, mollusc, crustaceans (pelagic-zooplankton and benthic) and fish.

Recommendations for improvements to methodology

With reference to the Flow Diagram (Figure 1-2 of Brown *et al*. [2003a]), it appears more logical to place the stage of identifying reference organisms and representative species before

[21] Biomagnified = The increase in concentration of a chemical in the tissue of organisms along a series of predator-prey associations, primarily through the mechanism of dietary accumulation.
[22] A Relatively high CR for ^{210}Po have been tabulated for wading birds in Brown *et al*. [2003a] but this value is associated with a high uncertainty/low confidence.

the collation of activity concentration information for biota. It is recognised that the exposure pathways analyses could include a component which deals with transfer to, and activity concentrations within, biota but this should be focused on a more general assessment, such as consideration of generic CR values from, for instance, IAEA [2004]. Such modifications to the Flow Diagram might include the addition of an extra data collation stage below the reference organisms and representative species consideration. More explicit guidance might also be provided on the types of data that might be collated at the various assessment stages. Furthermore, some rationalisation of the components used at this stage of the analyses is required. It may be appropriate to delete the notion of representative species and replace with “feature” organisms that have a more practical applicability in an assessment setting. The representative species approach was primarily designed as a provisional assessment component to facilitate the development of the FASSET methodology (i.e. primarily to provide a range of geometries suitable for reference organism).

5.4.4 Biota

Data on activity concentrations in marine biota have also been placed in categories of impact zone, North Sea and European background (Table 5.3). Some data on activity concentrations of TENORM radionuclides in the vicinity to the oil platforms are available for selected fish species including pelagic/demersal types Cod (*Gadus morhua*), Haddock (*Melanogrammus aeglefinus*), Mackerel (*Scomber scombrus*), Herring (*Clupea harengus*) and benthic types, i.e. Lemon Sole (*Microstomus kitt*) and Dab (*Limanda limanda*). Data are also available on ^{210}Po in fish, molluscs and crustaceans from the North Sea, ^{210}Pb in crustaceans and fish and ^{226}Ra in molluscs and pelagic crustaceans.

Data for “background” activity concentrations of ^{226}Ra and ^{210}Po in pelagic crustaceans[23], benthic crustaceans, molluscs and fish have been taken from Brown *et al*. [2004]. We are aware that there may be some (slight) duplication of data for the North Sea within this background database. These background values provide a useful insight into whether the activity concentrations from biota in the impact zone and North Sea constitute elevated (i.e. excess) levels.

In cases where activity concentrations are registered on a dry weight basis, conversion to wet mass has been made using the factors provided by Cherry & Shannon [1974]. The relevant wet/dry factors are 3.7 for fish (entire organism); 6 for zooplankton (euphausiids and mysids[24]) and 3.7 for marine invertebrates (excluding zooplankton, whole body).

There is an inhomogeneous distribution of the selected radionuclides within the bodies of some marine organisms (e.g. the activity concentrations of ^{210}Po in hepatopancreas of crustaceans and pyloric caecum of fish is much higher than that of flesh [Brown *et al*. 2004]). Where possible data are presented in a comparable form for a radionuclide over the three contamination areas.

It is not possible to assign an activity concentration “excess” to biota from the impact zone and North Sea from a consideration of the empirical data alone. Activity concentrations for all radionuclides in biota from the impact zone and North Sea fall within the range observed for natural background.

[23] It is assumed that data for the general category zooplankton can be sensibly applied to the pelagic crustaceans group.
[24] The pelagic crustaceans considered in this work, e.g. *pandalus borealis*, do not fall under the order/suborders mysids and euphausiids. Nonetheless these taxonomic groups were considered most representative of the biota of interest in this work.

Table 5.3: Mean activity concentrations in biota – empirical data

Radionuclide	Reference biota	Impact zone Bq kg^{-1} fw *(range)*	Area North Sea Bq kg^{-1} fw *(range)*	Background Bq kg^{-1} fw *(range)*
Ra-226	Pelagic fish	n.d.	n.d.	0.2 [b,e] (*0.06 – 0.5*)
	Benthic fish	n.d.	n.d.	0.2 [b,e] (*0.06 – 0.5*)
	Pelagic crustacean/ zooplankton	n.d.	0.59[d] (*0.52-0.69*)	0.2[b,e] (*0.1-0.9*)
	Benthic crustacean	n.d.	n.d.	0.74 [b,e] (*0.07 – 2.1*)
	Mollusc	n.d.	0.85 [d] (*0.39-1.30*)	0.74 [a,e] (*0.07 – 2.1*)
Po-210	Pelagic fish	1.13[a,c] (*0.38-2.03*)	0.64 [d] (*0.28-2*)	1.6[a,f] (*0.05-7*)
	Benthic fish	1.26 [a,c] (*0.97-1.54*)	2.45 [d] (*0.81-4.4*)	1.6[a,f] (*0.05-7*)
	Pelagic crustacean/ zooplankton	n.d.	47.5 [d] (*43-52*)	30[b,f] (*0.7-89*)
	Benthic crustacean	n.d.	2.44 [d] (*1.1-3.48*)	33[b,f] (*4-89*)
	Mollusc	n.d.	37.4 [d] (*22-56*)	42[a,f] (*10-120*)
Pb-210	Pelagic fish	n.d.	0.013 [d]	n.d.
	Benthic fish	n.d.	0.042 [d]	n.d.
	Pelagic crustacean/ zooplankton	n.d.	n.d.	n.d.
	Benthic crustacean	n.d.	0.12 [d] (*0.013-0.23*)	n.d.
	Mollusc	n.d.	n.d.	n.d.

n.d. = No data; [a] = muscle or soft tissue/flesh; [b] = whole body; [c] NRPA [2004]; [d]Dahlgaard [1996] &/or Young *et al.* [2002]; [e] "Typical values" from Cherry & Shannon, [1974]; [f]Brown *et al.* [2004]

Testing CR values

Total filtered activity concentrations (Table 5.1) from the impact zone and North Sea have been used to predict activity concentrations in biota for these areas. If filtered seawater activity concentrations were not available, total activity concentrations in seawater have been taken to be representative of filtered values.

Table 5.4 : Predicted activity concentrations in reference flora and fauna (Bq kg^{-1} fw) using FASSET CR values and empirical data for activity concentrations in seawater.

Radionuclide	Location	Reference organism	Seawater concentration (Bq l^{-1})	Predicted biota concentrations Bq kg^{-1} (whole body fw)
^{226}Ra	Impact Zone	Phytoplankton	$1.6x10^{-3}$	3
		Pelagic crustacean/ zooplankton		0.16
		Benthic crustacean		0.16
		Mollusc		0.16
		Pelagic fish		0.16
		Benthic fish		0.16
^{210}Po	North Sea	Phytoplankton	$0.8x10^{-3}$	56
		Pelagic crustacean/ zooplankton		24
		Benthic crustacean		16
		Mollusc		11
		Pelagic fish		5
		Benthic fish		5
^{210}Pb	North Sea	Phytoplankton	$0.72x10^{-3}$	7
		Pelagic crustacean/ zooplankton		0.7
		Benthic crustacean		65
		Mollusc		1
		Pelagic fish		0.15
		Benthic fish		0.15

The predicted activity concentrations of ^{226}Ra in pelagic crustaceans/zooplankton and molluscs are slightly below the range of empirical data for corresponding organism groups in the North Sea, but fall within the range expressed by empirical data for corresponding organism groups collated for "natural background" (Table 5.4). The FASSET methodology predicts ^{226}Ra activity concentrations in benthic crustaceans and pelagic and benthic fish that fall within the range documented from natural background based on field observation. No empirical data were available for phytoplankton with which to test FASSET predictions. Generally speaking, the FASSET methodology predicts ^{226}Ra activity concentrations in reference organisms reasonably well.

Predicted ^{210}Po activity concentrations in fish are above the ranges expressed by empirical data from the impact zone and the North Sea (Table 5.4). However, predicted values do fall within the greater range associated with natural background empirical datasets. A similar pattern is observed in the case of benthic crustaceans. In the case of pelagic crustaceans/zooplankton and mollusc, the predicted activity concentrations are below the range in the data set from the North Sea but again fall within the range provided for the empirical "natural background" data set. The FASSET methodology appears to provide reasonable estimates for transfer of ^{210}Po to reference organism groups.

For ^{210}Pb, the empirical dataset is limited to a few samples for benthic crustaceans and fish in the North Sea only. In all cases, the FASSET predicted whole body concentrations are overestimates. In the case of benthic crustaceans this over-prediction is in excess by two orders of magnitude.

Deriving the technological component of activity concentrations in biota

The excess activity concentrations of ^{226}Ra, ^{210}Pb and ^{210}Po in seawater (Table 5.2) have been used to derive the activity concentration arising specifically from discharges of radionuclides from the oil and gas platforms (Table 5.5).

Table 5.5: "Excess" activity concentrations of ^{226}Ra, ^{210}Pb and ^{210}Po in biota derived from anthropogenic activity alone in the impact zone.

Radionuclide	Reference organism	Seawater activity concentration (Bq l^{-1})	Biota activity concentration Bq kg^{-1} (whole body fw)
^{226}Ra	Phytoplankton	$1x10^{-3}$	2
	Pelagic crustacean/ zooplankton		0.1
	Benthic crustacean		0.1
	Mollusc		0.1
	Pelagic fish		0.1
	Benthic fish		0.1
^{210}Po	Phytoplankton	$0.15x10^{-3}$	11
	Pelagic crustacean/ zooplankton		4.5
	Benthic crustacean		3
	Mollusc		2.1
	Pelagic fish		0.9
	Benthic fish		0.9
^{210}Pb	Phytoplankton	$0.15x10^{-3}$	1.5
	Pelagic crustacean/ zooplankton		0.15
	Benthic crustacean		13.5
	Mollusc		0.23
	Pelagic fish		0.03
	Benthic fish		0.03

5.5 Application of impact assessment methodology

5.5.1 Dose rate derivation

In this section estimated unweighted absorbed doses have been derived; consideration of weighted doses is presented later.

For the aquatic environment, in the derivation of external DCCs it has been assumed that radionuclides are uniformly distributed in an infinite absorbing medium. In our case, the activity concentrations of technologically enhanced ^{226}Ra, ^{210}Pb and ^{210}Po might decrease with depth to a constant level in the sediment profile. This prediction is purely based on the observation that these radionuclides are being supplied from the water column and that the top layers of sediment may therefore be expected to incorporate an unquantified fraction of unsupported radionuclide that declines to a supported level at depth. However, even where such a vertical distribution exists, the influence on dose rate is unlikely to be significant given the overall negligible importance of external irradiation (see Figure 5.6).

Generic occupation factors have been used based on information provided in life history data sheets [Brown *et al*. 2003a]. It has been assumed that phytoplankton, zooplankton and pelagic fish spend their entire time in the middle of the water column. An infinite absorbing medium model therefore fits the target source configuration in this case. For benthic molluscs, benthic crustaceans and benthic fish it has been assumed that these organisms spend their entire time at the sediment water interface. A semi-infinite absorbing medium model, with exposures arising from both the underlying sediment and the overlying water column, is most appropriate in this case. Phytoplankton, zooplankton/pelagic crustacean and pelagic fish are assumed to spend 100% of their time in the water column, and benthic crustacean, mollusc and benthic fish 100 % of their time at the sediment surface.

The internal component of the dose rate dominates the total dose rate in all cases (Figure 5.6). The predicted unweighted absorbed excess dose rates vary from a minimum of 4.6×10^{-3} µGy h^{-1} for pelagic fish to a maximum of 6.8×10^{-2} µGy h^{-1} for phytoplankton.

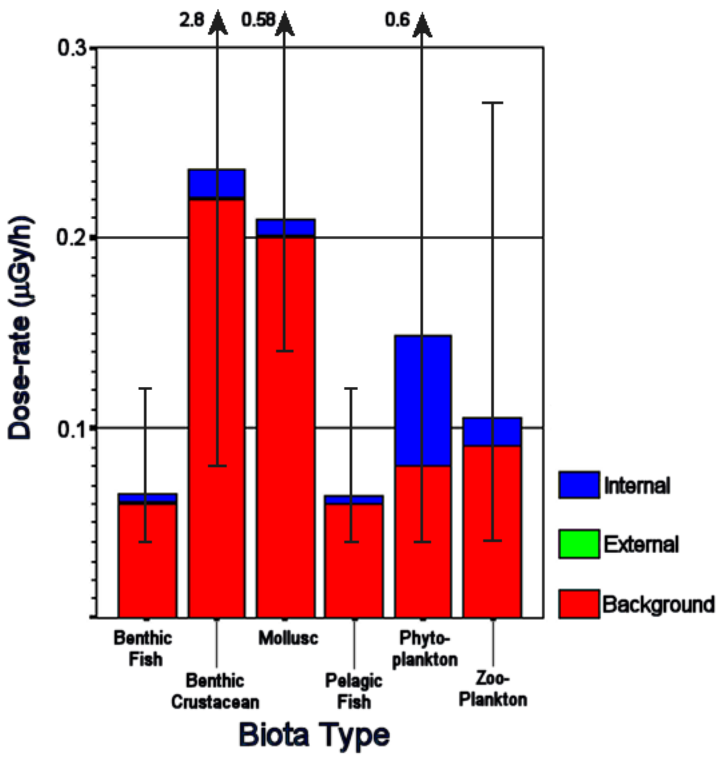


Figure 5.6: Unweighted absorbed dose rates (µGy h^{-1}) to selected reference organisms from the background presence of naturally-occurring radionuclides (red bar), and the internal and external components of technologically enhanced levels of naturally occurring radionuclides (blue and green bars respectively). The bars define the range in dose rates derived for naturally occurring radionuclides from Brown *et al*. [2004].

5.5.2 Comparison with Marina II and background dose estimates

In the MARINA II project [Betti 2004; European Commission 2003] internal dose rates arising specifically from oil platform discharges of Ra isotopes were made for mollusc, fish and shrimps. Average activity concentrations and volumes of produced waters per platform were used as input data for a model simulation which involved the use of a simple box model with nominal size 1x1 km, 20 m mixing layer and a water exchange with the open sea of approximately 0.5-1 times per day.

To make a sensible comparison, weighted DCCs need to be applied within the FASSET methodology [Pröhl *et al.* 2003]. Because ^{228}Ra was not included in FASSET, internal weighted ^{228}Ra DCCs have been taken from Brown *et al.* [2003b]. A weighting factor of 20 has been used for alpha radiations in these calculations to be consistent with the weighting factor used in MARINA II [Sazkyina & Kryshev 2003]. The predicted excess activity concentrations of ^{226}Ra and ^{228}Ra in the vicinity of the oil platforms, i.e.1 mBq l^{-1} in both cases, has been used in subsequent calculations.

Amiro [1997] and Golikov & Brown [2003] provide a internal DCC for ^{226}Ra of 2.46×10^{-5} Gy $year^{-1}$ per Bq kg^{-1} fw (2.8×10^{-3} µGy h^{-1} per Bq kg^{-1} fw) assuming all alpha radiation emitted is absorbed by the target. The corresponding value in FASSET is 1.7×10^{-2} µGy h^{-1} per Bq kg^{-1} fw because FASSET DCC values include all daughter radionuclides down to ^{214}Po whereas Brown *et al.* [2003b] and Amiro [1997] both use a separate DCC for ^{226}Ra and its daughter ^{222}Rn.

Using the FASSET methodology an internal dose rate of 0.03 µGy h^{-1} from excess Ra isotopes for all biota groups has been derived reflecting the fact that (i) most of the dose is derived from the weighted alpha component of ^{226}Ra and daughters (therefore the effect of a variable geometry is negligible) and (ii) all biota have the same CR value and therefore the same (predicted) body concentration of ^{226}Ra. This value compares to the corresponding values calculated in MARINA II [Betti *et al.* 2004] of 1.25-2.92 µGy h^{-1} to mollusc, 0.71-1.42 µGy h^{-1} to fish and 0.15-0.28 µGy h^{-1} to shrimp. There is an approximately 2 orders of magnitude difference between the the estimates which can be partly explained by the seawater concentrations used as input to the biota activity and dose rate calculations. A radium excess of 5-10 mBq l^{-1} was used in MARINA II, which is approximately one order of magnitude above the level used as input to the calculations performed here. As the current assessment is based on measurements of produced waters and a site specific transport model we conclude that the current assessment is more realistic than that of MARINA II. Note also that higher Ra CR values were used for the MARINA II assessment in the case of fish and mollusc (based on IAEA [1985]).

Further analyses demonstrates that the predicted excess dose rate is in most cases < 20 % of that arising from the occurrence of natural radionuclide in the marine environment (Figure 5.6). An exception to this appears in the case of phytoplankton for which the excess dose rate is similar to natural background. It is apparent from the graph that the dose rate arising from exposure to emissions from oil and gas production would be difficult to discern from the variability in dose rates observed in natural marine systems (as defined by the bars in Figure 5.6). The dose rate excesses derived here are orders of magnitude below those for which biological effects have been observed in aquatic systems under controlled conditions. For example, morbidity effects have been observed on fish/fish eggs (arguably the most sensitive truly-aquatic reference organism) at dose rates as low as 8.3 µGy h^{-1} [Woodhead & Zinger 2003] compared to the dose rates that we have derived for this reference organism group in the approximate order of 5×10^{-3} µGy h^{-1}.

5.6. Conclusions

The following conclusions have been drawn with respect to the utility of the FASSET assessment methodology within a marine TENORM case study. On the positive side :

- The methodology is easy to apply and the rationale behind each step is transparent;
- For the marine system, coverage of CR values is complete.

However, several problems have been identified, notably:

- No explicit guidance has been provided by FASSET for TENORM cases. It would be useful to provide a dose rate excess in such cases. Further guidance for this type of special case would be useful;
- Better guidance at the "*Problem formulation*" stage of the assessment is required;
- Disequilibria in the natural decay series between ^{226}Ra and ^{214}Po makes the application of an aggregated DCC inappropriate (but whether this inaccuracy is acceptable deserves consideration). The problem could be mitigated by splitting the ^{226}Ra DCC (perhaps applying a cut-off for inclusion of daughter radionuclides within the parent DCC to 1 day);
- More detail may be required on (i) input data (i.e. filtered seawater concentrations) (ii) the quantity actually being derived by the application of distribution coefficients. Application of k_d values will result in the exchangeable component of activity concentration as oppose to the whole-sediment concentration being predicted;
- There is no marine DCC for ^{228}Ra;
- The stages in the assessment may require modification (refer to the flow diagram Figure 1-2 in Brown *et al.* [2003a]). A data collation stage should also be placed following the identification of reference organisms;
- It may be useful to consider the use of "Feature species" as opposed to the representative species mentioned in Brown et al. [2003a]. It seems more likely that, in routine cases, the assessor will be concerned with named plants or animals specified by, for example, national legal stipulation or international agreement.

6 Chernobyl exclusion zone case study report

The 30 km exclusion zone around the Chernobyl nuclear power plant represents an area contaminated by a large number of radionuclides for which there are comparatively large datasets on the activity concentrations in a number of wildlife species and reported biological effect which may be attributable to the effects of ionising radiation. The exclusion zone contains both terrestrial and freshwater ecosystems and the FASSET framework has been applied to both. As this is not a regulated site, the testing of the FASSET framework has been restricted to the dose assessment (including testing the CR values) and effects analyses stages (see Section 2).

6.1 Terrestrial ecosystems

The most appropriate terrestrial ecosystems types considered in the FASSET framework for the Chernobyl exclusion zone are semi-natural pastures/heathlands and forests. The reference organisms for these ecosystems are: soil micro-organism, soil invertebrate, burrowing mammal, carnivorous mammal, herbivorous mammal, detritivorous invertebrate, bird egg, grass/herb, shrub and tree.

6.1.1 Data compilation

Data on the activity concentrations of radionuclides in biota within the Chernobyl exclusion zone and observations of the effects of radiation exposure on biota were compiled from four sources:

1. The database described by Gaschak *et al*. [2003] which contains a total of *circa* 700 measurements including ^{137}Cs and ^{90}Sr activity concentrations in the muscle and bone respectively of wild animals sampled throughout the Chernobyl exclusion zone between 1988 and 2000. These include species of predominantly large herbivorous (*Alces alces*, *Capreolus capreolus*, *Cervus elaphus* and *Lepus* spp.) and carnivorous (*Vulpes vuples* and *Canis lupus*) mammals. The herbivorous mammal data are dominated by results for *C. capreolus* (63 of 68 ^{137}Cs measurements). Muscle ^{137}Cs activity concentrations were assumed to be representative of whole-body as radiocaesium is distributed relatively homogenously throughout body tissues [Coughtrey & Thorne 1983]. Bone was assumed to contribute 97 % of the body ^{90}Sr and comprise 10 % of the animals' live-weight [Brown et al. 2003b].

2. The compilation of Russian language publications summarising studies conducted between 1986 and 1998 on the effects of ionising radiation on wildlife described by Sazykina *et al*. [2003]. Caesium-137 activity concentrations are also presented for a range of rodent species. Here we have only used data collected from 1988 onwards. Caesium-137 activity concentrations are presented for a range of rodent species. Whilst it is not always evident as to if these are for whole-body or muscle on the basis of the above argument we have assumed that this does not matter.

3. A review of English language publications. Useful data arising from this were restricted to ^{90}Sr and radiocaesium activity concentrations in the skeleton and muscle of small mammals reported by US workers [Baker *et al*. 1996;1999; Chesser *et al*. 2000; Holloman *et al.* 2000; Matson *et al*. 2000] from studies conducted at six, predominantly highly contaminated, sites between 1994 – 1996 investigating the effects of ionising radiation on small mammal populations. To convert the dry weight muscle results to fresh weight values, a dry matter content for muscle of 25 % was assumed; other assumptions

used during data manipulation were as for the data of Gaschak *et al*. [2003] discussed above.

4. Studies conducted by the University of Liverpool between 2001 and 2003 which measured ^{90}Sr and ^{137}Cs activity concentrations in a range of invertebrate species and rodents from an number of collection sites across a gradient of soil activity concentrations [Gilhen *et al*. 2001-2003]; these studies also consider the effects of radiation on invertebrates and small mammals, and on soil biological activity.

Prediction of soil radionuclide activity concentrations

A number of the available data did not have corresponding radionuclide activity concentrations in soil which could be used to test the FASSET CR values. Furthermore, there were no data reporting activity concentrations of any radionuclides other than ^{134}Cs, ^{137}Cs and ^{90}Sr in animals. As an aim of the assessment was to estimate total doses to reference organisms in the Chernobyl exclusion zone, a method of estimating activity concentrations of important radionuclides in soil was required. As most available data were reported with a location and an interpolated surface of ^{137}Cs deposition was available to us (data from measurements made in 1991-2 block kriged to a resolution of 58 m x 58 m pixels [Beresford *et al.* in-press), predicting soil activity concentrations using a geographical information system (GIS) was the most effective option. Soil activity concentrations of ^{89}Sr, ^{90}Sr, ^{91}Sr, ^{95}Zr, ^{95}Nb, ^{99}Mo, ^{103}Ru, ^{106}Ru, ^{129m}Te, ^{132}Te, ^{131}I, ^{133}I, ^{134}Cs, ^{136}Cs, ^{140}Ba, ^{141}Ce, ^{144}Ce, ^{154}Eu and ^{239}Pu were estimated using distance varying relationships of these radionuclides to ^{137}Cs deposition [Mück *et al*. 2002]; activity concentrations of ^{238}Pu, ^{241}Am and ^{241}Pu were estimated from their total depositions within the exclusion zone relative to that of ^{137}Cs [UIAR 2001]. No correction of soil ^{241}Am activity concentrations was made to account for in-growth from ^{241}Pu decay as this was assessed as being insignificant over the time period considered.

Activity concentrations in soil were predicted for an area around the sampling location with a radius typical of the home range for the species in question; mean, minimum and maximum soil activity concentrations for the pixels falling within this area were recorded. Where reported, sampling locations were not given as latitude-longitude co-ordinates but as, for example, a village name, the soil activity concentration in an area of 1 km radius of the village was estimated. For a limited number of collated sampling sites, soil activity concentrations were reported for ^{90}Sr, radiocaesium, ^{154}Eu, ^{238}Pu, ^{239}Pu and ^{241}Am. Predictions are generally within an order of magnitude of the observed mean (and in the range of available measurements). Figure 6.1 compares predicted and observed radiocaesium and ^{241}Am soil activity concentrations for some of these sites. This approach makes the assumption that activity concentrations in soil remain constant other than for radioactive decay (i.e. movement of radionuclides down the soil profile is not taken into account). A 0-10 cm layer soil bulk density of 1110 kg m^{-3}, as determined from soil profile measurements made within the exclusion zone [UIAR 2001], was used in these calculations.

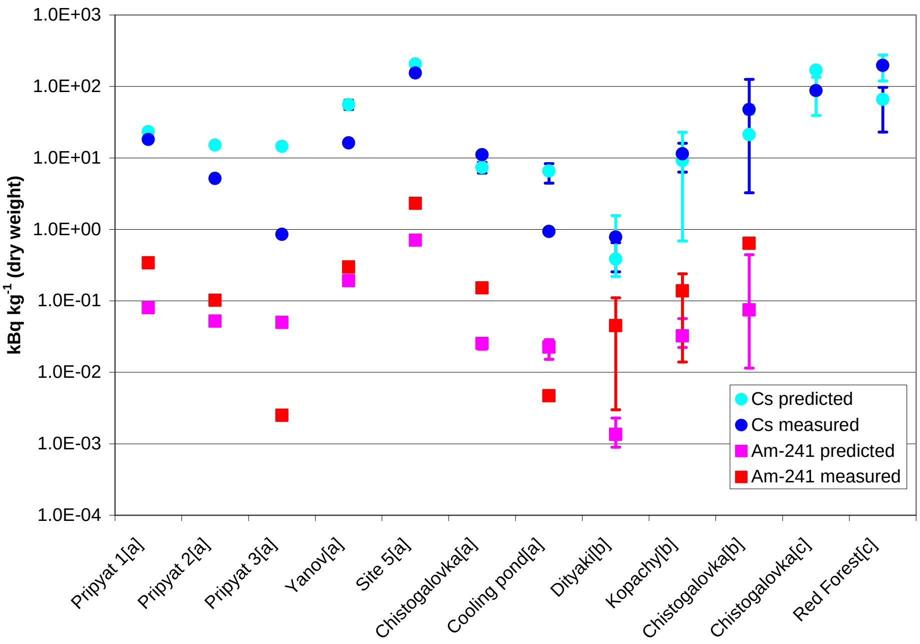


Figure 6.1: Comparison of measured radiocaesium and ^{241}Am activity concentrations in soils with GIS predictions: (a) [Gilhen *et al.* 2001-2003], ^{137}Cs reported; [b] [Baker *et al.* 1996; Chesser *et al.* 2000], $^{134+137}$Cs reported; [c] [Jones 2004], ^{137}Cs reported. Error bars represent minimum and maximum with the exception of measurements from Baker *et al.*/Chesser *et al.* which are ±SD.

6.1.2 Estimating whole-body radionuclide activity concentrations

Whilst CR values for terrestrial plant reference organisms are presented, these are not subsequently used within the FASSET methodology to estimate internal radiation exposure of plants [Brown *et al.* 2003a]; no CR values are recommended for soil micro-organisms as absorbed doses will be predominately defined by the activity concentrations in the surrounding medium. Consequently, we will only consider transfer to animal reference organisms in this paper; recommended CR values for FASSET animal reference organisms provide estimates of whole-body radionuclide activity concentrations. The look-up tables provided for semi-natural pastures/heathlands within the FASSET framework are considerably more comprehensive than those presented for forests. Given that in the collation of empirical CR values for semi-natural reference organisms no differentiation was made on the basis of habitat, because of the paucity of data [Brown *et al.* 2003a], many of the semi-natural pasture/heathland CR values are also likely to be applicable to the same reference organism in forest ecosystems. Here, we concentrate on application of the semi-natural ecosystem CR values although limited comment is given to comparative results if forest ecosystem values were used.

Whole-body activity concentrations were estimated as the product of measured or predicted soil activity concentration and the appropriate CR value selected. The CR values within the FASSET framework are not complete for all reference organism–radionuclide combinations. For this assessment, CR values were not available for: ^{90}Sr and radiocaesium for burrowing mammals; ^{90}Sr and ^{106}Ru for detritivorous and soil invertebrates; and bird egg, Pu isotopes for soil invertebrate and bird egg; and ^{241}Am for bird egg. In the case of burrowing mammals, the highest available CR value for a similar reference organism was used (herbivorous mammal in the case of ^{90}Sr and carnivore in the case of radiocaesium). For the other reference-organism-radionuclide combinations, the highest available CR values were used (an exception was that for actinide elements CR values for bird egg, the highest available values for mammalian reference organisms were assumed and not those for invertebrates) as recommended in the FASSET framework[25]. Concentration ratio values available for radiocaesium and ^{90}Sr (the nuclides for which comparisons of predicted and observed activity concentrations were possible) are presented in Table 6.1.

Table 6.1: Strontium-90 and radiocaesium concentration ratio values recommended in the FASSET framework for reference organisms considered within this case study.

Reference organism	FASSET CR value ^{90}Sr	FASSET CR value Radiocaesium
Detritivorous invertebrate	none given	0.085
Herbivorous mammal	1.96	1.84
Carnivorous mammal	1.3	4.96
Burrowing mammal	none given	none given

6.1.3 Comparison of measured and predicted whole-body activity concentrations

Comparisons of predictions with observed ^{90}Sr and radiocaesium activity concentrations reported in a range of rodent species (*Apodemus* spp., *Sorex* spp., *Microtus* spp. *Clethrionomys galreolus*, *Muscardinus avellanarius*) by Gilhen *et al.* [2001-2003] and Chesser *et al.* [2000] are presented in Tables 6.2 and 6.3 respectively. The mean ratios of predicted to observed activity concentrations for the Gilhen *et al.* data were 5.7 (range 0.05-28) for ^{90}Sr and 25 (0.02-294) for ^{137}Cs; for the data of Chesser *et al.* [2000] the mean ratios were 8.4 (0.5-39) and 11 (0.1-62) respectively. The minimum and maximum observed activity concentrations are also presented for each sampling site in Tables 6.2 and 6.3. Variation in predicted values is a consequence of the different CR values used dependent upon the reference organism selected to best represent the different species sampled. However, predicted values vary little between species as the recommended CR values for different mammalian reference organisms are similar (see Table 6.1). Considerable variation has been reported for rodents at sites in the Chernobyl zone (see 'Observed' in tables 2 and 3). Whilst, Chesser *et al.* [2000] report that overall *C. glareolus* and *A. falvicollis* had the highest radiocaesium activity concentrations of the species they sampled, there was no consistent rank order for radiocaesium activity concentration in species across their sampling sites.

Data summarised for rodent species by Sazykina *et al.* [2003] provide only single values of radiocaesium activity concentration in soil and animals at a range of sites; species sampled were similar to those for the other datasets above with the addition of *Mus* spp.. Predictions were on average 14-37 times higher than observed data depending upon the CR value used. Predictions of whole-body ^{90}Sr and ^{137}Cs activity concentrations (using mean GIS estimated radionuclide activity concentrations in soil for the home range area of each species) by reference organism group are compared to the database of Gaschak *et al.* [2003] in Table 6.4. Whilst predicted means and ranges were close to observed values with predicted to observed ratios (for all species) of 12 (0.01-110) and 3.6 (0.09-64) for ^{137}Cs and ^{90}Sr respectively, there was relatively poor agreement for individual data (R^2<0.17) (Figure 6.2). Predictions were also made on the basis of the maximum activity concentration in a given pixel in the home range area. For ^{137}Cs, only 23 of the 156 maximum predictions were less than the observed value, however, for ^{90}Sr, 48 of the 119 maximum predictions were below the observed value.

Predicted ^{137}Cs and ^{90}Sr activity concentrations for *detritivorous invertebrates* are compared with the reported values [Gilhen *et al.* 2001-2003] for *Carabidae* and 'ground dwellers' (i.e. assuming these groups are likely to contain detritivorous species) in Table 6.5.

[25] These recommendations are not explicit within the FASSET framework – they represent the assessors interpretation of the recommendations and also similar recommendations of Copplestone *et al.* (2003) (eds).

Table 6.2: A comparison of observed (2001-3) and predicted whole-body ^{90}Sr and ^{137}Cs activity concentrations in rodent sampled at various locations in the Chernobyl exclusion zone [Gilhen *et al*. 2001-2003].

Site name	^{90}Sr (kBqkg^{-1} fw)		^{137}Cs (kBq kg^{-1} fw)	
	Observed	Predicted	Observed	Predicted
Pripyat 1	1.3-2.1	16	0.31-0.79	91
Pripyat 3	0.90	9.6	0.14	4.3
Yanov	19	38	17	81
Chistogalovka	0.35-3.1	4.6	0.30-1.0	55
Cooling pond	1.4-3.7	4.4	0.61-2.9	4.7
Red Forest	55-764	41	180-2260	297
Site 1	4.5-25	16-19	5.3-6300	133-196
Site 2	3.0-48	74	11-260	446
Site 3	5.5-1200	1830-2760	0.62-21000	7440-10950
Site 4	0.13-4.5	1.0-1.5	0.01-61	8.7
Site 5	142	134	66	771

Table 6.3: A comparison of observed (1994-96) and predicted whole-body ^{90}Sr and radiocaesium activity concentrations in rodent species sampled at various locations in the Chernobyl exclusion zone [Chesser *et al*. 2000].

Site name	^{90}Sr (kBq kg^{-1} fw)		Radiocaesium (kBq kg^{-1})	
	Observed	Predicted	Observed	Predicted
Chistogalovka	1.2-70	61-92	0.25-34	436
Glyboke lake	7.6-235	71-107	7.8-6490	621
Orchard	-	-	0.25-42	129
Red forest enclosure	-	-	3.3-42	109
Red Forest grassland (remediated area)	2.2-142	14-22	0.50-2230	109-111
Red Forest woodland	3.8-84	194	22-20520	986

Table 6.4: A comparison of observed (1988-2000) and predicted whole-body ^{90}Sr and ^{137}Cs activity concentrations in mammalian reference organisms sampled throughout the Chernobyl exclusion zone [Gaschak *et al.* 2003]. Predicted values presented were estimated using the mean radionuclide activity concentration in soil within typical home ranges for sampled species.

Reference organism	^{90}Sr (kBq kg^{-1} fw)		^{137}Cs (kBq kg^{-1} fw)	
	Observed	Predicted	Observed	Predicted
Herbivorous mammal	5.9 (0.07-43.7)	5.3 (0.16-106)	21.2 (0.03-266)	14.2 (0.60-299)
Carnivorous mammal	0.59 (0.13-1.1)	0.45 (0.20-0.97)	8.7 (0.69-44.0)	20.2 (3.3-66.9)

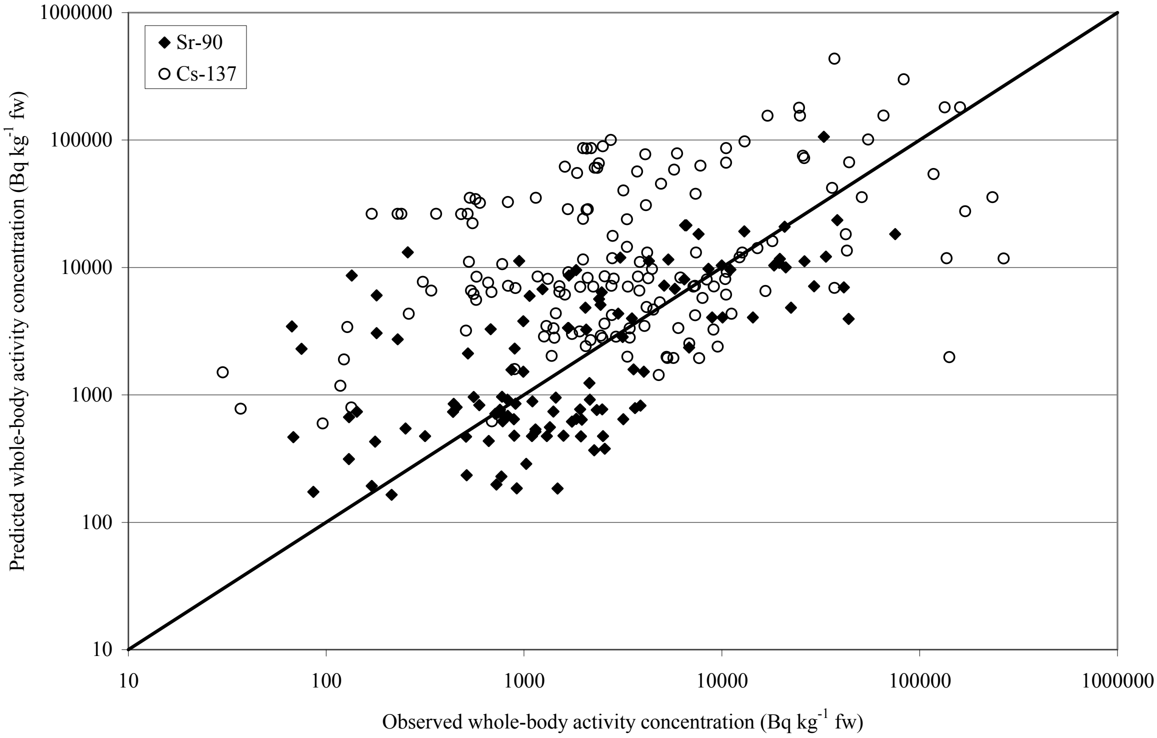


Figure 6.2: Comparison of observed [Gaschak et al. 2003] and predicted (from mean soil activity concentration in home range) whole-body ^{90}Sr and ^{137}Cs activity concentrations for large mammalian species sampled throughout the Chernobyl exclusion zone. For comparative purposes a 1:1 (predicted : observed activity concentration) line is also shown.

Table 6.5: A comparison of observed (2001-2) and predicted whole-body ^{90}Sr and ^{137}Cs activity concentrations in invertebrates sampled at various locations in the Chernobyl exclusion zone [Gilhen *et al*. 2001-2003].

Site name	^{90}Sr (kBq kg^{-1} fw)		^{137}Cs (kBq kg^{-1} fw)	
	Observed	**Predicted**	**Observed**	**Predicted**
Pripyat 1			0.19	1.5
Pripyat 3			0.18	0.07
Yanov			27	1.4
Chistogalovka			7.3	0.94
Cooling pond			0.84	0.08
Red Forest			1100	5.0
Site 1	0.53-72	19	0.29-210	3.3
Site 2			6.7-30	7.6
Site 3	21-4600	2800	1.2-180	190
Site 4			0.08-0.15	0.15
Site 5			12	13

6.1.4 Estimation of absorbed dose rates

Absorbed dose rates were preferentially estimated using measured radionuclide activity concentrations in biota and soil and appropriate DCCs. Where soil activity concentrations were lacking, these were predicted from spatially variable deposition as described above, and CR values were used to estimate whole-body activity concentrations when measured values were not available. Of the radionuclides considered in the FASSET framework, it was possible to estimate exposure to ^{89}Sr, ^{90}Sr, ^{106}Ru, ^{134}Cs, ^{137}Cs, ^{238}Pu, ^{239}Pu, ^{241}Pu and ^{241}Am (i.e. it was not possible to estimate absorbed dose rates for a number of the radionuclides known to have been deposited in the exclusion zone, although many of those for which soil concentrations could be estimated are short-lived isotopes). Estimated doses were weighted assuming weighting factors of 10 for α-radiation, 3 for low-energy β-radiation (<10 keV), and 1 for β-radiation (>10 keV) and γ-radiation. Where applicable, to estimate the external dose received by burrowing mammals it was assumed that small rodents spend 60 % of their time underground and carnivorous mammals 40 %. Absorbed dose rates have been estimated for both plant and animal reference organisms. To estimate external dose rates the FASSET DCCs were applied to dry weight soil activity concentrations; the FASSET methodology does not specify whether dry or wet weight soils activity concentrations should be used.

Doses were estimated for selected references/sampling sites from the available data to cover a range of reference organisms, observed effects and contamination levels. Estimates were made assuming mean and maximum observed/estimated activity concentrations in biota and soil. No studies reporting observations prior to 1988 were selected in an attempt to avoid the consequences of short-lived radionuclides and the passage of the contaminated air mass.

For small mammals, γ- and high energy β-doses resulting from ^{90}Sr and radiocaesium contributed >99 % to the total estimated whole-body absorbed dose. The estimated contribution of internal dose to the total absorbed dose rate for small mammals was generally greater than that of external exposure (Tables 6.6-6.7). Alpha-doses contributed significant proportions of the estimated total dose received by invertebrate reference organisms but not mammals or bird egg (Table 6.8). From the maximum measured soil ^{241}Am activity concentrations of Gilhen *et al.* [2001-2003] an ^{241}Am whole-body activity concentration in herbivorous mammals at Site 5 (Figure 6.1; Table 6.2) of 4 Bq kg^{-1} (fw) is estimated. For the sampling sites of Gaschak *et al.* [2003] ^{241}Am and ^{239}Pu whole-body activity concentrations in herbivorous mammals of up to 0.5 and 2 Bq kg^{-1} (fw) respectively can be estimated (using GIS predicted soil activity concentrations averaged over typical home ranges).

6.1.5 Effects analysis

Table 6.6 presents estimated dose rates for various entries from the collation of Sazykina *et al.* [2003] comparing the reported biological effects for plants with summarised information from the FASSET framework [Larsson *et al.* 2004]. The reported biological effects were observed within the range of estimated dose rates at which radiation induced effects may have been expected (on the basis of information summarised in the FASSET framework).

There are considerable data on biological effects observed in a range of rodent species in the compilation of Sazykina *et al.* [2003]. Table 6.7 compares the potential range in dose rates estimated for 1988 with observed effects from Sazykina *et al.* and summarised information from the FASSET framework [Larsson et al. 2004]. For reasons of brevity, and the nature of how some observations have been reported, this comparison is presented for an area of 10 km radius around the Chernobyl plant. Where it is possible to comment, the observed effects are as would be expected, from the range of estimated dose rates and the FASSET summary of biological effects.

Table 6.6: Observed biological effects in plant species within the Chernobyl exclusion zone [Sazykina *et al.* 2003] compared to absorbed dose rates estimated using the FASSET framework and summarised biological effects [Larsson *et al.* 2004]. Note the absorbed dose rates estimated by the FASSET methodology do not include an internal dose contribution.

Species (year observed)	Absorbed dose rate µGy h^{-1}		Observed effects	FASSET framework summarised effects
	Mean	**Maximum**		
Arabidopsis thaliana (mouseear cress) (1988)	5.7	11	47±5.0% of plants mutated *cf.* <5% in unexposed populations	i) 5.5 µGy h^{-1}) - decreased seed weight ii) *c.* 40 µGy h^{-1} - increased mutation rate in micro-satellite DNA iii) >100 µGy h^{-1} - morbidity responses iv) *c.* 1000 µGy h^{-1} - mortality (pines)
Taraxacum officale (dandelion) (1988)	33	170	Seed germination 40+2.4% *cf.* 94+2.5% for control	
Pinus sylvestris (Scotch pine) (1990-91)	5.9	43	56 % of pollen tubes branched and 2 % multi-branched *cf.* 29 % and 0.1 % in control samples	

A series of studies investigating small mammal species within the Chernobyl exclusion zone conducted between 1994 and 1996 are reported by Baker and co-workers [Baker et al. 1996;1999; Chesser *et al.* 2000; Holloman *et al.* 2000; Matson *et a*l. 2000]. Table 6.8 presents estimated absorbed does rates for rodents at four of their study sites based predominantly on reported whole-body and soil activity concentrations [Chesser *et al.* 2000]. Whilst the authors stated that there was little evidence to suggest persistent impaired performance of populations and communities [Chesser *et al.* 2000], some biological effects, which may be attributable to exposure to ionising radiation, were reported [Baker et al. 1996;1999; Chesser *et al.* 2000; Holloman *et al.* 2000; Matson *et a*l. 2000]. The only morphological abnormality observed in small mammals (n>300) collected at these sampling sites was an enlarged spleen in a few individuals and examination of karyotypes did not show gross chromosomal rearrangements [Baker *et al.* 1999]. Genetic diversity was significantly higher in the bank vole (*Clethrionomys glareolus*) population of the Red Forest site than at a reference site outside of the Chernobyl exclusion zone [Matson *et al.* 2000]; diversity was also greater at the Glyboke Lake site than at the reference site but this was not significant. Whilst this may have been the result of exposure to ionising radiation the authors noted that it could be due to the recolonisation from different founding populations; in 1986 reduced numbers of rodents had been recorded in some areas of the exclusion zone [Sazykina *et al.* 2003]. Mitochondrial DNA heteroplasmy (intra-individual DNA sequence variation) was compared in female *Microtus arvalis* (common vole) and their embryos from the Red Forest site and a control site [Baker *et al.* 1999]. Heteroplasmy levels were greater in the sample from the Red Forest but not significantly different to results for the control site. However, transversions and the presence of multiple base pair substitutions in a single fragment were only observed in the Red Forest sample. Mice and vole (*Apodemus agrarius*, *Apodemus sylvaticus* and *C. glareolus*) liver samples collected from these study sites in 1996 were analysed for oxidative stress enzyme activity [Holloman *et al.* 2000]. Only *A. agrarius* demonstrated significantly reduced oxidative stress enzyme activity compared to animals collected from less contaminated areas. The authors make reference to studies which demonstrate the development of radioresistance in *C. glareolus* and *A. sylvaticus* within the Chernobyl zone and speculate that these species may have developed a higher radioresistance than *A.*

agrarius. As for the comparisons within Tables 6.6 and 6.7, the observed effects reported for the sampling sites in Table 6.8 are as expected from the estimated dose rates and comparison with the FASSET effects summary (see Table 6.7).

Table 6.7: Summary of observed biological effects in rodent species within the Chernobyl exclusion zone [Saykina *et al*. 2003] compared to absorbed whole-body dose rates estimated using the FASSET framework and summarised biological effects [Larsson et al. 2004]. Observed data and dose rate predictions are for 1988; mean and range (in parenthesis) dose rates are estimated for area of 10 km radius around the Chernobyl plant.

Absorbed dose rate (μGy h^{-1})		Observed effects	FASSET framework summarised effects
Internal	41 (0.09-770)	i) Liver abnormalities in a few animals (more abnormalities were observed in 1986/87)	i) Growth (rat) not affected at 16 μGy h^{-1} (affected at >3000 μGy h^{-1})
External	8.1 (0.02-200)	ii) Leucocyte concentrations lower than in control animals	ii) 180-850 μGy h^{-1} some blood parameters affected
Total	**49 (0.1-980)**	iii) Increased embryonic mortality at some sites[1] up to 43 % before implantation *cf*. 15 % in controls; after implantation up to 28 % in contaminated area and 3.4% in controls	iii) No effect on thyroid function at *c*. 10^{4} μGy h^{-1}
		iv) Increased frequency of abnormal sperm heads at some sites[1] up to 17% *cf*. 1.3% for controls	iv) No effect lifespan (mice) - 460 μGy h^{-1}, (significant reductions >*c*. 10^{3} μGy h^{-1} (mice, goat, dog))
		v) Increased frequency of reciprocal translocations in male mice at some sites[1] *cf*. controls	v) *c*. 100 μGy threshold for reproductive effects
		vi) Increased levels of chromatid and genome aberrations in bone marrow cells at some[1] *cf*. controls	vi) Mutation LOEDR[2] >420 μGy h^{-1} (mice)

[1]No significant differences were observed for these parameters at other sites studied within the Chernobyl exclusion zone compared with controls; [2]LOEDR – lowest observed effect dose rate.

Table 6.8: Estimated mean and maximum absorbed dose rates for rodents at sites described by Chesser *et al*. [2000] and Baker *et al*. [1996].

Site	Estimated absorbed dose rate mean (maximum) μGy h^{-1}		
	Internal	**External**	***Total***
Chistogalovka	5.3 (47)	12 (18)	**17 (65)**
Glyboke Lake	94 (1150)	17 (17)	**111 (1170)**
Red Forest	214 (3320)	26 (37)	**241 (3360)**
Orchard	8.6 (11)	3.5 (5.2)	**12 (17)**

Gilhen *et al*. [2001-2003] report studies at a number of sites within the Chernobyl exclusion zone conducted between 2000 and 2003. Estimated absorbed dose rates to rodent species at their most contaminated site were the highest estimated for the data considered within this assessment (Table 6.9). A significant increase in mitotic index for *C. glareolus* was recorded at this and other sites, which was related to increasing contamination level. Whilst no differences in rodent reproductive organ or spleen size compared with control sites were observed, post-mortem examination of *C. glareolus* specimens found some individuals exhibiting epithelial metaplasia and hyperplasia, and small aggregations of lymphocytes. Measurements of soil biological activity at this site, measured using bait lamina which predominantly reflect the biological activity of soil invertebrates, were significantly lower than at less contaminated sites [Gilhen *et al*. 2001-2003]. The summary of effects data in the FASSET framework contained too few data for invertebrate species to comment on the impact of chronic doses in the range estimated here. Table 6.9 presents estimated mean absorbed dose rates for all FASSET reference organisms at this site. Dose rates for detritivorous invertebrates and burrowing mammals were estimated using measured whole-body ^{137}Cs and ^{90}Sr activity concentrations. For comparison to the other reference organisms, if recommended CR values had been used to derive dose rates, the resultant estimates would have been *circa* 2 and 4-fold higher for detritivorous invertebrates and burrowing mammals respectively. Estimated mean dose rates for most mammalian reference organism are in excess of those above which we may expect shortening of life, reduced growth rates and impacts on reproduction to occur [Larsson *et al*. 2003]. Dose rates estimated for plant reference organisms are sufficient that some reproductive, mutation and morbidity responses may be predicted (*cf*. Table 6.7). There are too few effects data for the remaining reference organisms to enable comment [Larsson *et al*. 2003].

6.1.6 Discussion

FASSET CR values

Predictions have been compared to measured values for radiocaesium and ^{90}Sr in a range of FASSET animal reference organisms encompassing a wide range of species (e.g. in the case of mammals from *Sorex* spp. to *Alces alces*). In all cases, predicted values were generally within the range of observed values which is encouraging, especially given the requirement to estimate soil activity concentrations at some sampling sites. Furthermore, mean predictions were either close to observed means, or were conservative estimates being approximately an order of magnitude greater, lending confidence to the overall FASSET approach. However, for a number of individual animal, or site-species mean, predictions were more than one order of magnitude lower than measured values. Brown *et al.* [2003a] suggest that maximum CR values (and hence expected predicted activity concentrations) are generally within an order of magnitude of the mean CR values recommended within the FASSET framework. Some of the variation in agreement between predicted and observed data at different sites may be explicable as a consequence of variables, such as soil type [Gillett *et al* 2001] or in the Chernobyl zone the density and nature of deposited fuel-particles (the predominant source of ^{90}Sr) [Kashparov 1999; 2000]. Some of the large mammal data [Gaschak *et al* 2003] were used, together with that from other sources, in the derivation of the FASSET CR values [Brown *et al.* 2003a]. It does not therefore represent a truly independent test of the FASSET CR values; especially for carnivorous mammals as Gaschak *et al*. [2003] provided most of the radiocaesium and ^{90}Sr CR values.

Given the time span of observed data against which comparisons have been made (1988-2003), it is perhaps worth noting that Gaschak *et al*. [2003] did not find a long-term temporal decline in either ^{90}Sr or ^{137}Cs activity concentrations of wild mammals in the Chernobyl exclusion zone.

Table 6.9: Mean estimated absorbed dose rates for all FASSET reference organisms assuming soil activity concentrations for the most contaminated site of Gilhen *et al*. [2001-2003] in 2003.

	Detritivorous invertebrate	**Soil invertebrate**	**Burrowing mammal**	**Small herbivorous mammal**	**Large herbivorous mammal**
Representative geometry	Woodlouse	Earthworm	Mouse	Rabbit	Roe deer
Absorbed dose rate ($\mu Gy\ h^{-1}$)	569	1938	603	3100	2932
Percentage of total absorbed dose rate from α-radiation[3]	25%	7%	0.4%	<0.01%	0.1%
	Small carnivorous mammal	**Large carnivorous mammal**	**Bird egg**	**Herb/ Shrub[1,2]**	**Tree[2]**
Representative geometry	Weasel	Fox			
Absorbed dose rate ($\mu Gy\ h^{-1}$)	3100	3763	189	343	281
Percentage of total absorbed dose rate from α-radiation[3]	<0.01%	<0.01%	1%	n/a	n/a

[1]Estimated dose rates are the same for these two reference organisms; [2]External dose rate only; n/a – not applicable as only external doses estimated; assuming RBE =10 for α-radiation.

Predicted ^{90}Sr activity concentrations in detritivorous invertebrates, made using the CR value for herbivorous mammals (given the lack of specific values for this reference organism) were within the range of the limited data available. However, predictions were perhaps not as conservative as expected on the basis of previous observations of the comparative transfer of Cs and Sr to invertebrates [Crossley 1963].

Predictions were made here using FASSET CR values for semi-natural pasture/heathland ecosystems which were based (predominantly) on environmental measurements. Recommendations are also made for mammalian reference organism in forest ecosystems [Brown *et al*. 2003a]. These are presented as ranges in aggregated transfer parameters and are predominantly based upon model predictions. If these had been used in the assessment, predicted whole-body ^{90}Sr and radiocaesium activity concentrations would be up to *circa* 2 orders of magnitude higher than those in Tables 6.2-6.4 (i.e. using the upper range values). Whilst this would result in no large individual underestimations compared to observed data, mean predictions would be considerably higher than observed values.

Doses and effects analysis

Whilst we can compare predicted and measured activity concentrations in biota, and therefore have some confidence in the internal doses estimated by the FASSET framework (assuming the internal DCCs are robust), we cannot 'test' our ability to measure external doses received by biota. We are making assumptions about how animals interact with their environment which could significantly influence external doses received.

It was not possible to assess all of the radionuclides which biota may be exposed to within the Chernobyl exclusion zone, either because the FASSET framework does not consider them (e.g. ^{154}Eu) or due to a lack of deposition data. However, the few available measurements for some of these radionuclides suggest that their contributions to doses would be low.

The dose rates estimated using the methodology presented in the FASSET framework for some reference organisms at some sampled sites within the Chernobyl exclusion zone are sufficient to predict that some biological effects may occur on the basis of the FASSET summaries [Larsson *et al*. 2003]. Although paucity of observations under conditions of chronic irradiation makes direct comparison difficult, the biological effects observed in the Chernobyl exclusion zone over the period considered here (1988-2003) are broadly in agreement with those which may have been expected. However, we should acknowledge that the summary of radiation effects within the FASSET framework [Larsson *et al*. 2004] include some observations from the Chernobyl exclusion zone which may relate to the same original Russian language works as collated by Sazykina *et al*. [2003]. In considering observations of the longer-term effects of radiation in the Chernobyl exclusion on biota it is often difficult to separate the consequences of residual acute exposure in the immediate aftermath of the accident from what may be responses to chronic radiation exposure. Some of the studies discussed here investigating potential responses to radiation in the Chernobyl exclusion zone have reported responses in some of the parameters investigated but not others.

Estimates of absorbed dose rates to large mammalian reference organisms in Table 6.9 are in excess of that at which mutations, shortening of life, reduced growth rates or impacts on reproduction may be expected. However, few, if any, studies have considered radiation effects in large wild mammals inhabiting the exclusion zone. From reported measurements of ^{90}Sr and ^{137}Cs activity concentrations in large herbivorous and carnivorous mammals sampled throughout the exclusion zone (1988-2000) (Table 6.4), mean absorbed dose rates from internal exposure of 2.3 and 9.2 $\mu Gy\ h^{-1}$ respectively can be estimated; maximum estimates are 10 and 95 $\mu Gy\ h^{-1}$ respectively.

The FASSET framework does not consider all organism types, birds and reptiles are notable exceptions which may require assessment under environmental protection legislation in many countries (e.g. see Sellafield case study above). Increased albinism, depressed immunoglobulin levels and reduced populations (between 1986 and 1996) were reported for barn swallows (*Hirado rustica*) in the Chernobyl exclusion zone [Ellegren *et al*. 1997; Camplani *et al*. 1999]. Reported whole-body ^{90}Sr and ^{137}Cs activity concentrations in *Lacerta agilis* (sand lizard) samples from one of the sites studied by Gilhen *et al*. [2001-2003] suggest dose rates comparable to those estimated above for rodents for this reptilian species.

Whilst not the aim of this case study, we should acknowledge that probably the largest overall impact of the Chernobyl accident on the ecology of the exclusion zone was brought about by the removal of the human population with the consequent cessation of activities such as agricultural production and the associated use of herbicides, pesticides and fertilisers. As a result, floral and faunal biodiversity and abundance has increased considerably [Baker & Chesser 2000]. However, as demonstrated above, effects characteristic of those expected from chronic exposure to ionising radiation continue to be observed. Estimated doses are considerably in excess of suggested international guidelines in some areas (e.g. the IAEA [1992] suggest that limits of 1 or 10 $mGy\ d^{-1}$ should not result in harm for terrestrial animal and plant populations respectively) and above the dose rate of *circa* 0.1 $\mu Gy\ h^{-1}$ (2.4 $mGy\ d^{-1}$) above which statistically significant effects were recorded in the studies summarised in the FASSET framework. In some cases, estimated doses are greater than thresholds above which shortening of life may be expected.

6.2 Freshwater ecosystems

Whilst the Chernobyl exclusion zone contains many freshwater ecosystems, the Chernobyl cooling pond have been by far the most extensively studied. Furthermore, a scenario for model testing was previously developed under the BIOMOVS programme [BIOMOVS 1996]. The application of the FASSET framework to freshwaters within the Chernobyl exclusion zone has been restricted to the cooling pond; the majority of studies considering biological effects on freshwater organisms in the Chernobyl exclusion zone have also been conducted in the cooling pond [e.g. Sazykina *et al*. 2003]. The BIOMOVS cooling pond scenario has been used here as the prime source of water and biota concentration data for this application and testing.

The cooling pond is located to the south-east of the Chernobyl nuclear power plant; it has a surface area of 22 km^2 and an approximate volume of 1.5×10^8 m^3 (hydrological and hydrochemical information on the cooling pond can be found in BIOMOVS [1996]).

6.2.1 Water, sediment and biota activity concentrations

The BIOMOVS cooling pond scenario presents annual averaged ^{137}Cs activity concentrations in water, phytoplankton (algae), pelagic fish species (silver carp; *Hypophthalmichtys molitrix*) and benthic fish species ('bream') for 1986-1990 (Table 6.10). Data are also presented for bivalve molluscs and sediment in some years (Table 6.10); interpolation and extrapolation have been used to provide values for the missing years (as indicated on Table 6.10). The cooling pond scenario does not present data for activity concentrations of isotopes other that ^{137}Cs in water following 1986. Estimates of the amounts of different radionuclides in the Chernobyl cooling pond (decay-corrected to 30 May 1986) estimated from sampling data of the radioactive contamination of water and bottom sediments [Kryshev 1995] were presented in the BIOMOV scenario (Table 6.11). For the purposes of this assessment, the activity concentration of radionuclides other than ^{137}Cs have been estimated assuming their ratios in water to (measured) ^{137}Cs remained constant to those derived from Table 6.11 with appropriate correction for radioactive decay. The BIOMOVS scenario does present limited data for some radionuclides additional to ^{137}Cs in bivalve molluscs, phytoplankton and fish species (see Table 6.14 later) and isotopic ratios in a range of biota for 1986 (Table 6.11).

The database described by Gaschak *et al*. (2003) contains ^{90}Sr and ^{137}Cs activity concentration data for a few mammal and bird species collected at the cooling pond between 1996 and 2003. These few data are summarised in Table 6.12.

Table 6.10: Average concentrations of the ^{137}Cs in media and reference organisms (adopted from BIOMOVS [1996]).

Year	Mean fresh weight activity concentration of ^{137}Cs ± 2SD (number of samples)					
	Sediment ($kBq\ kg^{-1}$)	**Water ($Bq\ l^{-1}$)**	**Phytoplankton ($kBq\ kg^{-1}$)**	**Bivalve mollusc ($kBq\ kg^{-1}$)**	**Pelagic fish ($kBq\ kg^{-1}$)**	**Benthic fish ($kBq\ kg^{-1}$)**
1986[a]	170±80 (174)	223[b] (86)	120±40 (7)	25±6 (9)	250±120 (3)	160±70 (4)
1987	155[c]	100±60 (36)	60±30 (28)	21±5 (21)	130±50 (96)	120±30 (54)
1988	140±90 (46)	50±20 (40)	40±20 (5)	14±8 (6)	60±30 (62)	30±10 (17)
1989	125[c]	30±14 (36)	40±12 (6)		30±14 (46)	20±8 (9)
1990	110±60 (18)	14±6 (30)	24±7 (20)		23±11 (48)	13±4 (16)

[a]After May 30, 1986
[b]Arithmetic mean of monthly-average values from 1986
[c]Interpolated linearly

6.2.2 Comparing transfer parameters and whole body activity concentrations

Table 6.13 presents a comparison of the estimation concentration ratio derived for radiocaesium for phytoplankton, bivalve mollusc, pelagic fish and benthic fish from the BIOMOVS cooling pond scenario data with the recommended CR values from the FASSET framework [Brown et al. 2003a]. There is good agreement between the measured and observed CR values for phytoplankton, but for benthic fish and pelagic fish, the FASSET values are *circa* one order of magnitude higher than the values derived from the measurements in the cooling pond. Measured CR values for bivalve molluscs are also considerably lower than the FASSET recommended value.

Comparisons for other radionuclides are not presented because of the lack of measured water concentrations data for radionuclides other than radiocaesium.

Some guidance is given by Brown *et al.* [2003a] on how to overcome the lack of advised CR values for a specific radionuclide-reference organism combination. This recommends that to obtain a rough estimate for CR in aquatic systems the numerical value of the k_d is applied[26]. However, this approach was not applied in this estimation as the assessors felt that even if it provided reasonable results, it was difficult to explain and to follow and it might affect the overall acceptance of the estimation.

A direct comparison between predicted and observed radiocaesium activity concentrations in freshwater bird species is not possible because of the lack of water data corresponding to the biota measurements. However, using the FASSET CR value for freshwater birds of 3000 l kg^{-1} an expected water activity concentration of <0.1 Bq l^{-1} can be estimated from the data presented in Table 6.12. Even allowing for reductions in water concentrations over the 6-9 years between the last water activity concentration (14 Bq l^{-1}) available in Table 6.10 and the time when the measurements of birds were made this estimated water activity concentration seems low. This may be the result of the simplistic assumption that the radiocaesium body burden of the birds is derived solely from the cooling ponds.

Table 6.11: Activity concentrations of different radionuclides relative to ^{137}Cs in cooling pond water, sediments and biota in 1986 (adapted from BIOMOVS [1996]).

	30/05/1986		15/07/1986			
Nuclide	**Sediment**	**Water**	**Phyto-plankton**	**Bivalve mollusc**	**Pelagic fish**	**Benthic fish**
^{90}Sr	0.1	0.45	0.13	2	8.0×10^{-3}	6.3×10^{-3}
^{95}Zr	0.83	11	0.83	1	n/a	n/a
^{103}Ru	0.67	6.4	n/a	n/a	n/a	n/a
^{106}Ru	0.33	2	0.67	1.2	0.14	5.6×10^{-2}
^{131}I	4.2	0.27	n/a	n/a	n/a	n/a
^{134}Cs	0.5	0.55	0.33	0.4	0.48	0.47
^{137}Cs	1	1	1	1	1	1
^{140}Ba	2	3.6	n/a	n/a	n/a	n/a
^{141}Ce	0.83	5.8	n/a	n/a	n/a	n/a
^{144}Ce	0.5	7.8	2.7	4.4	0.16	9.4×10^{-2}
^{239}Pu	3.5×10^{-4}	n/a	n/a	n/a	n/a	n/a

n/a – not available

[26] Brown *et al*. recommended that the available CR values for the radionuclide would first be compared to the value of k_d – the larger number being used in the assessment (eds).

Table 6.12: Summary of available whole-body activity concentration data for birds and mammals collected in the cooling pond area (from database of Gaschak *et al.* [2003]).

Reference organism	Species	Sampling period	^{137}Cs Bq kg^{-1} fw	^{90}Sr Bq kg^{-1} fw
Mammal	*Castor fiber*, *Ondatra zibethicus*	2003	Mean 330 Range 180-460 n=6	-
Bird	*Larus ridibundus*, *Haliaeetus albicilla*, *Gavia* spp.	1996-1999	Range 40 – 250 n=3	Range 10-30 n=2

Table 6.13: A comparison of radiocaesium CR values for different reference organisms estimated from data presented within BIOMOVS [1996] and the CR values recommended in the FASSET framework [Brown *et al.* 2003a].

Nuclide	CR value derived from BIOMOV data 1986	1987	1988	1989	1990	FASSET CR value
			Phytoplankton			
^{134}Cs	390	820	840	1400	2040	1000
^{137}Cs	540	600	800	1330	1710	
			Bivalve mollusc			
^{134}Cs	100	200	230			1000
^{137}Cs	110	210	280			
			Pelagic fish			
^{134}Cs	1180	1750	1150	1050	1530	10200
^{137}Cs	1120	1300	1200	1000	1640	
			Benthic fish			
^{134}Cs	740	1560	610	700	1020	12200
^{137}Cs	720	1200	600	670	930	

6.2.3 Estimating absorbed doses

Absorbed dose rates have been estimated for all of the FASSET freshwater reference organisms. For the estimation of internal doses measured activity concentrations have been used by preference. Where these are not available, but data are reported from 1986 (see Table 6.11) activity concentrations in each assessment year have been estimated assuming physical decay of each radionuclide. If this was not possible, biota activity concentrations have been estimated using the FASSET CR values (where available) and estimated water activity concentrations (see Section 6.2.1). Table 6.14 summarises the availability of biota activity concentrations and FASSET CR values.

For the purposes of selecting appropriate DCCs from the FASSET framework phytoplankton, bivalve molluscs and benthic fish were assumed to inhabit the water-sediment, and birds and mammals the air-water interface. All other reference organisms were assumed to inhabit the water column. As far as possible the information in the FASSET data on life history of the reference organisms has been used to estimate the occupancy rates of different habitats [Brown *et al.* 2003a]. However, the information is given only for a few example species, is of qualitative nature, and not useful for quantitative assessments of times that are spent in specific habitats. All organisms inhabiting the water-sediment interface were assumed to spend 50% of their time in the water column and 50% in the sediment in the estimation of total external absorbed dose rates. Those animals inhabiting the air-water interface were assumed to spend 50 % of their time on the water surface; in this instance the external dose rate was estimated as 0.25×(water column DCC)[27]. The list of radionuclides, for which the DCC are available in the FASSET framework, does not include some radionuclides measured in the cooling pond: ^{103}Ru, ^{132}I, ^{140}Ba, ^{140}La, and ^{141}Ce. For the radionuclides included in this case study, α- and low energy β-emitters are not relevant; therefore, the weighted doses have the same numerical values as the unweighted absorbed dose rates.

Table 6.14: Availability of measured biota activity concentrations and FASSET CR values to estimate internal doses of freshwater reference organism in the Chernobyl cooling pond.

Reference organisms	Available data from BIOMOVS [1996]	Available FASSET CR values*
Phytoplankton	^{90}Sr, ^{95}Zr, ^{106}Ru, $^{134,137}Cs$, ^{144}Ce	-
Zooplankton	None	^{89}Sr, ^{90}Sr, ^{131}I, $^{134,137}Cs$
Crustacean	None	^{131}I
Insect larvae	None	^{131}I, $^{134,137}Cs$
Vascular plant	None	None
Gastropod	None	None
Amphibian	None	None
Bivalve mollusc	^{90}Sr, ^{95}Zr, ^{106}Ru, $^{134,137}Cs$, ^{144}Ce	-
Pelagic fish	^{90}Sr, ^{95}Zr, ^{106}Ru, $^{134,137}Cs$, ^{144}Ce	-
Benthic fish	^{90}Sr, ^{95}Zr, ^{106}Ru, $^{134,137}Cs$, ^{144}Ce	-
Mammal	None	None
Bird	None	$^{134,137}Cs$

*Availability of CR values is not indicated when measured data are available.

Estimated annual average absorbed dose rates are presented separately for internal and external exposure for all FASSET freshwater reference organisms in Table 6.15. The estimated internal dose rates shown are based on measured radionuclide activity concentrations for four reference organisms only: phytoplankton, bivalve mollusc, and pelagic and benthic fish. Only for these species can the internal dose rates be considered as realistic. For many other organisms, the internal exposure dose rates are generally underestimated because of the lack of activity concentration data and available CR values. External exposures are considerably higher for species inhabiting the water-sediment interface than other habitats.

The ^{137}Cs and ^{90}Sr activities observed in mammals and birds in the later 1990's and early 2000's (Table 6.12) result in estimated internal absorbed doses in the order of less than 0.5 $\mu Gy\ h^{-1}$.

6.2.4 Effects analysis

The assessments focus on the first years after the accident. After 1990, all estimated exposures are below 100 $\mu Gy\ h^{-1}$ the dose rate below which statistically significant radiation induced effects were largely unobserved in the studies reviewed by Woodhead & Zinger [2003]. However, within the assessment period a number of reference organism had estimated dose rates in excess of 100 $\mu Gy\ h^{-1}$. The summary of the effects of chronic exposure on aquatic reference organism presented within the FASSET framework is shown in Table 6.16; for many freshwater reference organisms there were too few chronic exposure data to make

[27] Note the FASSET framework does not present guidance on how to estimate dose rates for biota inhabiting the air-water interface; the method of calculation used here was based on judgement of assessors (eds).

any conclusion. The compilation of data from the former Soviet Union (fSU) by Sazykina *et al*. [2003] (see also [Kryshev *et al.* in-press]) reports studies on fish in the Chernobyl cooling pond; Table 6.17 summarises observations made on *H. molitrix* from the fish farm at the cooling pond. Note that referring to Kryshev *et al.* [1996] and Kryshev [1998] for dose reconstruction Kryshev *et al*. [in-press] report that the highest dose rates after the Chernobyl accident (1986) were about 8-9 mGy d^{-1}; in 1989-1992 dose rates were 0.4 mGy d^{-1}. These dose rates are considerably higher than estimated for pelagic fish in 1989-90 in Table 6.15; the reasons for this difference have not been investigated.

Table 6.15: Estimated external and internal annual average absorbed dose rates for freshwater reference organisms in the Chernobyl cooling pond.

Reference organisms	**Average dose rate (µGy h^{-1}) in year**				
	1986	**1987**	**1988**	**1989**	**1990**
			External exposure		
Phytoplankton	511	183	88	51	33
Zooplankton	0.39	0.11	4.2×10^{-2}	2.1×10^{-2}	8.8×10^{-3}
Crustacean	0.38	0.11	4.1×10^{-2}	2.1×10^{-2}	8.6×10^{-3}
Insect larvae	0.39	0.11	4.1×10^{-2}	2.1×10^{-2}	8.7×10^{-3}
Vascular plant	0.41	0.11	4.4×10^{-2}	2.2×10^{-2}	9.2×10^{-3}
Gastropod	0.28	7.7×10^{-2}	3.1×10^{-2}	1.6×10^{-2}	6.8×10^{-3}
Amphibian	0.26	7.1×10^{-2}	2.9×10^{-2}	1.5×10^{-2}	6.5×10^{-3}
Mollusc	221	51	27	19	14
Pelagic fish	0.24	6.5×10^{-2}	2.7×10^{-2}	1.4×10^{-2}	6.0×10^{-3}
Benthic fish	194	41	23	16	12
Mammal	9.9×10^{-2}	2.7×10^{-2}	1.1×10^{-2}	6.0×10^{-3}	2.6×10^{-3}
Bird	9.3×10^{-2}	2.6×10^{-2}	1.1×10^{-2}	5.7×10^{-3}	2.4×10^{-3}
			Internal exposure		
Phytoplankton	10	2.8	1.0	0.65	0.28
Zooplankton	49	20	10	6.0	2.8
Crustacean	4.5×10^{-3}	–	–	–	–
Insect larvae	6.5	2.5	1.2	0.68	0.31
Vascular plant	–	–	–	–	–
Gastropod	–	–	–	–	–
Amphibian	–	–	–	–	–
Mollusc	126	68	31	24	20
Pelagic fish	112	44	16	6.7	4.6
Benthic fish	62	38	8.0	4.7	2.8
Mammal	–	–	–	–	–
Bird	250	102	46	25	11

Measurements of chromosomal aberrations in carp (*Cyprinus carpio*) corneal epithelium and developing silver bream (*Blicca bjorkna*) embryos showed no increase over expected ranges. An increased frequency of chromosome aberrations was observed in chironomids (non-biting midge larvae) in the Chernobyl nuclear plant Cooling Pond. In frogs sampled within 5 km of the Chernobyl plant, the frequency of aberrant cells was higher than in controls by 4-7 fold in 1987, 2-5 fold in 1988, and 2-3 fold in 1989. Reductions in the mollusc population of the Chernobyl NPP Cooling Pond were also reported.

In the early 1990's a higher rate of DNA strand breaks was observed in catfish (*Ictalurus punctatus*) collected from the Chernobyl NPP cooling pond compared to control fish (Sugg *et al*. 1996). DNA strand breaks were correlated with the radiocaesium activity concentration in muscle. The mean ^{137}Cs activity concentration was 35 kBq kg^{-1} (dw), which corresponds to a dose rate of a few μGy h^{-1}; as noted in Section 2.4 such cytogenetic effects have previously been observed at dose rates below 100 μGy h^{-1}.

6.2.5 Conclusions

Comments on the application of the FASSET framework for the Chernobyl cooling pond are:

1. External doses are strongly influenced by assumed habitat of organisms. External dose rates for species living near sediment reach values of several hundreds μGy h^{-1} whilst for organisms in open water habitat these dose rates are considerably lower.
2. The life history data provided in FASSET are qualitative rather than quantitative. Further assumptions are necessary to derive quantitative data that can be used in dose assessments.
3. Internal exposures of more than 100 μGy h^{-1} are estimated for biota with high accumulation of caesium in 1986. In general, internal dose rates are underestimated due to lack of measured data on organism's internal contamination and relevant CR values within FASSET.
4. FASSET provides some guidance for the management of missing data. However, the approaches were not applied for this case study. Although the suggestions could possibly provide reasonable results, the procedure is difficult to explain and to follow and it might affect the overall acceptance of the estimation.
5. From the measured data available for the Chernobyl cooling pond, CR values could be derived for caesium for phytoplankton, molluscs, pelagic and benthic fish. Good agreement was found for phytoplankton compared with the CR value recommended in FASSET, for molluscs and fish, the FASSET values are up to one order of magnitude higher than observations made in the cooling pond.
6. The available DCCs and (especially) CR values in FASSET framework for freshwater reference organisms are incomplete and do not allow an assessment for all radionuclides which have been measured in the Chernobyl cooling pond.

Table 6.16: Summary of chronic effects data for fresh water reference organisms based on the FRED (adapted from Larsson *et al*. [2004]).

Wildlife group	Freshwater ecosystem reference organism	Morbidity	Mortality	Reproductive capacity	Mutation
Amphibians	Amphibian	Too few data[1]	No data	No data	Too few data
Aquatic plants	Vascular plant	Too few data	Too few data	No data	No data
Bird	Wading bird	No data	No data	Conclusive dose effects could be drawn for chicken for dose rates $>10^4$ µGy h^{-1}	Too few data
Crustacean	Crustacean	No data	No data	No data	No data
Fish	Pelagic Fish Benthic Fish	One experiment (but not another) indicates effects on immune system at <8.3 µGy h^{-1}	Too few data	One study showing effects on gametogenesis at 230 µGy h^{-1}; otherwise effects at $>10^3$ µGy h^{-1}	Radiation exposure increases mutation rate
Mammals	Mammal	Growth (rat) not affected at 16 µGy h^{-1} (affected at >3000 µGy h^{-1}); 180-850 µGy h^{-1} some blood parameters affected; no effect on thyroid function at $9x10^3$ µGy h^{-1}	No effect lifespan (mice) - 460 µGy h^{-1}, (significant reductions >*c*. 10^3 µGy h^{-1} (mice, goat, dog))	*c*. 100 µGy h^{-1} threshold for reproductive effects; clear effects at $>10^3$ µGy h^{-1}	Too few data. One of nine references gives a mutation LOEDR[2] >420 µGy h^{-1} (mice)
Molluscs	Molluscs	Too few data. One of two references gives LOEDR $>10^4$ µGy h^{-1} for *Physa heterostrophaone*	Too few data. Two references both with LOEDR $>10^4$ µGy h^{-1} for *P. heterostrophaone* & *Mercenaria mercenaria*	Too few data. One of two references gives LOEDR $>10^4$ µGy h^{-1} and HNEDR[3] of $>10^4$ µGy h^{-1} for *P. heterostrophaone*	No data
Zooplankton	Zooplankton	Too few data	No data	Too few data. Only reference gives LOEDR of 440 µGy h^{-1} for *Tetrhymena pyriformis*	No data

[1]Too few data to draw conclusions; [2]LOEDR – lowest observed effects does rate; [3]HNEDR – highest no effects dose rate

Table 6.17: Effects of chronic radiation exposure on reproduction and off-spring of silver carp (*Hypophthalmichthys molitrix*) in the Chernobyl cooling pond (from compilation of Sazykina *et al*. 2003).

General conditions of study	Description of observed effects
Adult fish from fish farm that survived the accident. Some studies were also made of free-swimming fish from the Cooling Pond.	**1989**:There were 5.7 % sterile specimens, and 8.6 % of specimens with gonad asymmetry. According to the data of cytological analysis, 25 % of males had anomalies of sexual cells. In the control population less than 0.25 % of specimens were sterile. **1990**: There were 12.5 % sterile specimens and 16.7 % of specimens with gonad asymmetry; 47.1 % of fish had anomalies of sexual cells. **1991**: There were 23.1 % of specimens with gonad asymmetry. No sterile specimens were detected. Cytological analysis: 68.8 % of fish had anomalies of sexual cells. **1992**: 42.9 % of males had a deformed shape of gonads; 100 % of males and 33.3 % of females had some anomalies of sexual cells. Freely-living fish from the Cooling Pond: 15.4 % of males were partially sterile, and 9.1 % of females had gonad asymmetry; 89.5 % of fish had anomalies of sexual cells.
Off-spring of the above, housed in fish farm enclosures in the Cooling Pond.	**Offspring born 1989**. In 1992, 28.7 % of young fish had anomalies, including 2.8 % sterile bisexual specimens, 11.1 % with anomalies in gonad shape, 8.3 % with anomalies of body shape, 3.7 % with anomalies of the swim bladder, and 2.8 % with other anomalies. **Offspring born 1990**. In 1992, 12.1% of fish had anomalies, including 3.2 % with anomalies of gonad shape and 8.9 % with anomalies of body shape. No sterile specimens were observed in this generation.

*Refer to Sazykina *et al*. for original data sources.

7 Komi case study report

7.1 Introduction

The work conducted in this case study has focussed on the Vodnyi area within the Komi Republic, located in the Ukhta River watershed with geographical coordinates of N 63^0 E 53^0 (Figure 7.1). Industrial operations here have led an enhancement of naturally occurring radionuclides from the decay series of ^{232}Th and ^{238}U in soils, plants and animals. The area was originally associated with oil production with refineries, and oil wells drilled into the banks of the local river. Subsequent analyses of samples from some of these boreholes indicated that groundwater might be a viable source of Ra and from 1931 to 1950 in Vodnyi radium was extracted from groundwaters. From 1947-1956, there was also uranium and radium extraction from imported ores.

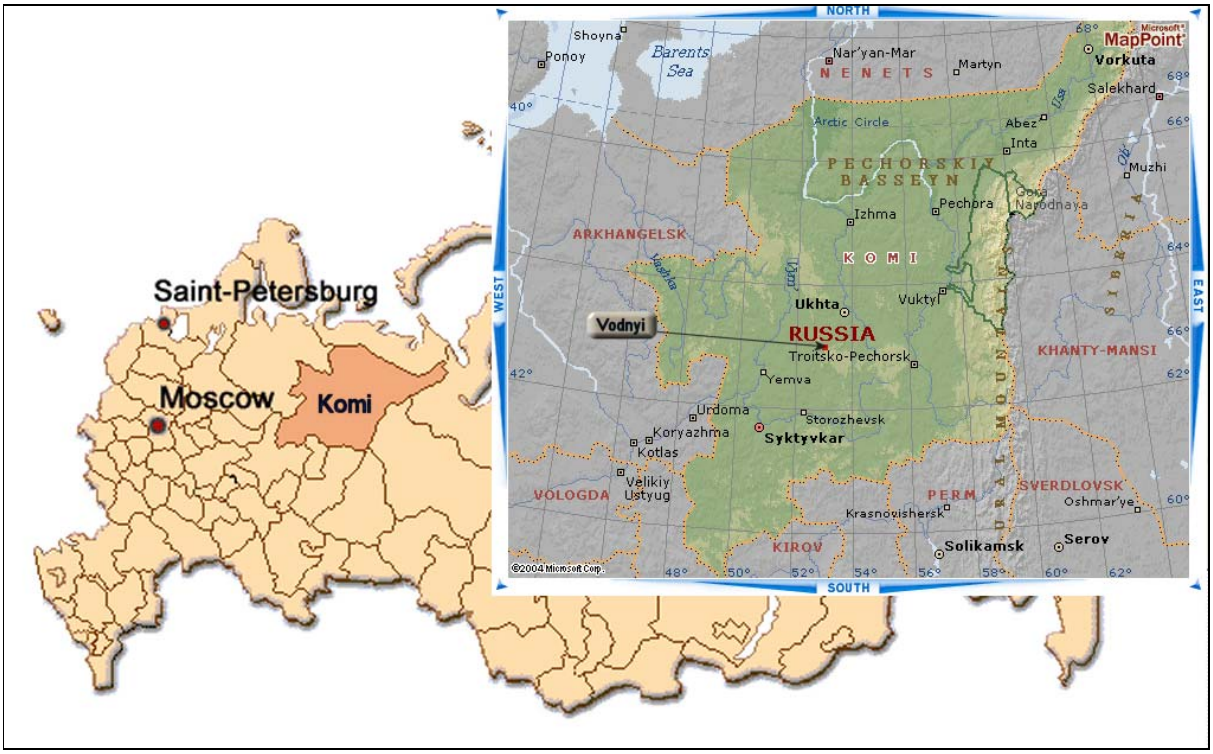


Figure 7.1: Location map of Vodnyi and the Komi Republic, Russia.

The Vodnyi area comprises of smaller contaminated sites including:

(i) Krokhal – contaminated by radium-rich ground water

(ii) Otvally – adjacent to the former central plant in the village of Vodnyi contaminated by tailings from residues of radium extraction from groundwater and from the processing of uranium ores

(iii) Obzhig – contains residues similar to Otvally with the additional presence of large quantities of semi-decomposed woody residues in the soil's A horizon.

A more detailed description of contaminated sites in Vodnyi area of the Komi Republic is provided by Taskaev *et al.* [2003].

7.2 Approach

Initially, soil data were collated from published literature sources from studies that had not been part of the original exercise to establish FASSET method parameters and reference data sets. For example, empirical data were excluded which had already been used [Brown *et al.* 2003a] from the Komi Republic in the derivation of CR look-up tables for several naturally occurring radionuclides.

The components of the FASSET impact assessment methodology that were tested in this case study are summarised in Table 7.1. Using the newly identified soil data, several results could be predicted: (i) activity concentrations in reference biota by application of appropriate FASSET CRs, (ii) external dose rates to reference biota through the application of appropriate external DCCs and (iii) effects for umbrella endpoints and selected organisms via the application of appropriate CRs, DCCs (external and internal), measured activity concentrations and analyses using the FRED database and the general observations made by in the FASSET framework [Woodhead & Zinger 2003]. Each of these predictions was then compared with actual measurements and observations made in the field.

Table 7.1: Components of the FASSET methodology considered in the case study and the datasets used to test them.

Input data set	FASSET component to be tested	Data set used for comparison
Soil concentrations	Semi-natural ecosystem CR values	Activity concentrations in Komi flora and fauna
Soil concentrations	External dose conversion coefficients	TLD, dose rate measurements
Soil concentrations, external dose rate measurements and activity concentrations in flora and fauna	Effects predicted from FRED	Effects observed at Komi

7.3 Data compilation

A literature search was conducted using (i) the INIS database and (ii) ETDEWEB. Relevant data were extracted and placed within EXCEL spreadsheets for subsequent analyses. Few relevant English language references were identified. The datasets were augmented using data collated during the EC Inco-Copernicus EPIC project [Brown *et al*. 2003b].

7.3.1 General assumptions and data manipulation

All data presented in Ci per unit mass were converted to Bq per unit mass.

Available data were often presented as total concentrations of U, Th, Ra by mass (g g^{-1}). Conversion of activity concentrations (Bq kg^{-1}) of various nuclides was carried out assuming that the total concentrations of U, Th, Ra in soil and biota were wholly attributable to ^{238}U, ^{232}Th and ^{226}Ra respectively [Hendersen 1982]. Since no data were available for ^{228}Ra, assumptions were based on its "daughter", ^{228}Th (for which data were available). Due to the similar half lives of the two nuclides – 5.78 y and 1.9 y for ^{228}Ra and ^{228}Th respectively, transient radioactive equilibrium has been assumed.

Some of the soil (and plant) data were in the form of activity concentrations ash weight[28], requiring a correction factor to convert from ash to dry weight. In view of the uncertainty associated with this conversion factor (no site specific ash to dry ratios are available) and the wish to maintain flexibility and transparency (new studies may yield appropriate conversion values in due course) this approach was deemed prudent. We have assumed that the site soil has a 10 % combustible organic material (OM) by mass since this value falls within a range of mineral (typically 1-6 % OM) and organic soils (typically >20 % OM by mass) [Brady, 1990]. For vegetation/plant material, we have assumed an ash to dry weight ratio of 1:40 (i.e. ash material constitutes 2.5 % of the total dry mass) based on the observations of Sheard *et al.* [1986] and used as a default conversion factor by Beresford *et al.* [2003]. A default dry weight to fresh weight conversion factors for vegetation has been selected of 10 % (dry weight as % of fresh) from the value given for "grass" in IAEA [1994]. In some instances, it was unclear as to what basis the observed data were reported (i.e. ash weight or dry weight).

7.3.2 Soil data

The "independent" soil data used as input for FASSET methodology application are summarised in Table 7.2. The data are taken from Titaeva *et al.* [1977], Titaeva *et al.* [1978] and Taskaev *et al.* [2003]; supplementary information is taken from Sazykina *et al.* [2003].

Generally, data from the upper 15 cm of the soil have been collated. However, for one study site [Titaeva *et al.* 1978] no sample depth was specified, instead weighted mean values for the "rhizosphere layer" at each site were given. Consequently, the activity concentrations presented in Table 7.2 are not necessarily consistent. Furthermore, it is unclear in some of the original references whether the data are given as Bq kg^{-1} ash or Bq kg^{-1} dw.

Table 7.2: Activity concentrations of uranium-thorium series radionuclides in soil from different locations in the Vodnyi area of Komi (Bq kg^{-1} dw).

Radionuclide	Radionuclide activity concentration (Bq kg^{-1} dw)		
	Krokhal	Obzhig	Otvally
^{238}U	<4 - 48	950 - 2300	700 - 3600
^{234}Th	<4 - 48	950 - 2300	700 - 3600
^{234}U	<6 - 44	960 - 1200	860
^{230}Th	220 - 2600	6400 - 240000	61000
^{226}Ra	8200 - 68000	7000 - 14000	35000 - 82000
^{210}Pb	3400 - 26000	8400 - 10000	41000 - 48000
^{210}Po	3400 - 26000	8400 - 10000	41000 - 48000
^{232}Th	18 – 27	22	24
^{228}Th	1500 - 4100	10000	3900

In Table 7.2, all data on ^{238}U, ^{232}Th, and ^{226}Ra are directly from measurements. Th-234 is assumed to be in secular equilibrium with ^{238}U. Activity concentrations of ^{234}U, ^{230}Th, and ^{228}Th were, if not measured, estimated using reported activity ratios in soil for the respective Komi sites [Titaeva *et al.* 1977, 1978]. Activity concentrations of ^{210}Pb and ^{210}Po are either measured, or derived assuming secular equilibrium between the two nuclides. Secular equilibrium between ^{226}Ra and its daughters has not been assumed.

[28] Note ashing may result in the loss of some radionuclides including ^{210}Po.

7.4 Uncertainties – ranges, mean and median values

Due to the lack of data for the Komi case study, a probabilistic assessment has not been possible. Similarly, little information is available on spatial distribution so it was not possible to produce a map of contamination or to derive activity concentration distributions. Therefore, we have selected "representative" activity concentrations cited within the open literature.

There is evidence for declining levels of naturally-occurring radionuclides in some environmental compartments with time [Taskaev *et al.* 2003]. Thus, it is clearly advantageous to compare data-sets from approximately the same time period.

In view of the long physical half-lives of most of the radionuclides of interest (i.e. ^{238}U, ^{232}Th, ^{226}Ra) and because the independent soil database (Section 7.3.2) and the empirical test data (including activity concentrations in biota, dose rates and observed effects) were derived from studies conducted roughly in the same time period[29], it has been assumed that the change in activity concentrations with time elapsed between the study periods has been negligible.

7.5 Testing transfer parameters

To provide a rigorous test of the activity concentrations in flora and fauna predicted using the FASSET methodology, a data set that is independent from the values used to derive CRs is required. In the Komi case study, identifying such an independent data set was difficult. In key publications such as Verhovskaya [1972], Komi data have been solely used by FASSET to derive herbivorous and burrowing mammal CR values for U and Th. In the case of Ra, the reported data for FASSET on transfer to carnivorous, burrowing and herbivorous mammals appear to be almost exclusively from Komi [Verhovskaya 1972; Pokarzhevskii & Krivolutzkii 1997]. These two publications used to derive FASSET CR values constitute the main, available source of information of activity concentrations in wildlife of the Komi area and cannot be used as an independent test dataset.

7.5.1 Flora

There are empirical activity concentration data for several plant species at identified locations in the Vodnyi area, Komi [Titaeva *et al.* 1978]. Some of these plant species fall within the broadly defined reference organism groupings such as "grass", although most belong to other families of flowering plants that are grass-like e.g. (clover - *Fabaceae*, yarrow - *Asteraceae*). Data have been collated and summarised by categories relating to location (Krokhal, Obzhig Otvally and "All" sites) and plant type (grass, tree, herbaceous species[30]). There is some justification for separating the data by location. Statistical analyses, using a non-parametric test of whether the samples are from populations with distinctly different distributions (Kruskal-Wallis H Test) confirms that differences between activity concentrations data in vegetation (all groups) from the 3 sites are significant ($p < 0.01$ for ^{226}Ra and ^{232}Th and 0.1 for ^{238}U). The data are summarised in Figures 7.2 and 7.3.

[29] Most of the studies used in this assessment appear to have been conducted between the mid 1960s and late 1970s. Exceptions to this are effects studies on soil invertebrates (from the 1980s) and cytogenetic studies on mammals (1990s).
[30] The herbaceous category has been defined to include all herbaceous plants that did not fall under the grass classification and any non-classified plant types.

Activity concentrations of ^{238}U, ^{232}Th and ^{226}Ra in vegetation have been predicted using FASSET CR values and soil activity concentrations given in Table 7.2. The predicted values are compared with values derived from field studies in Table 7.3.

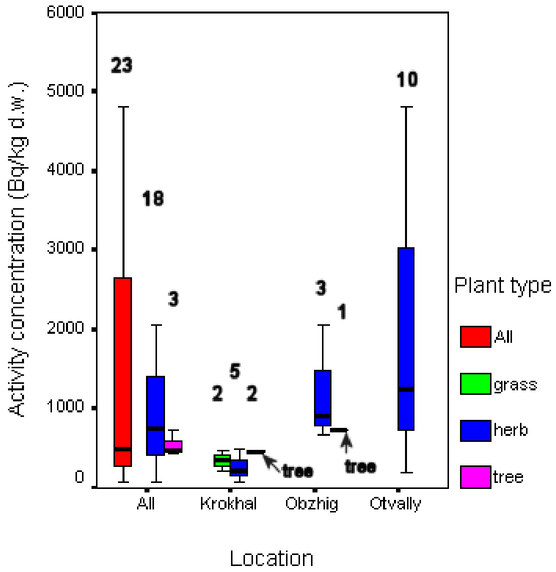
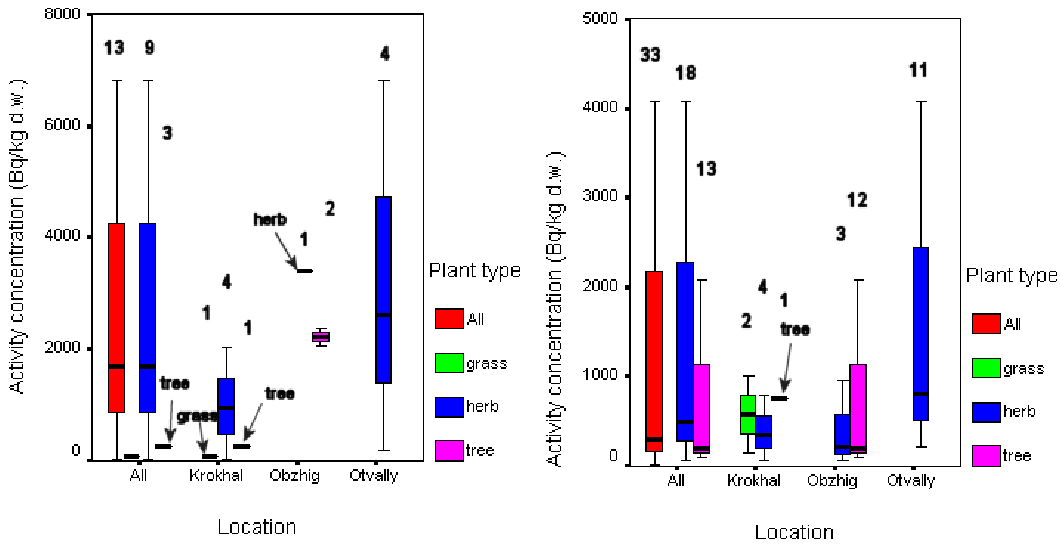


Figure 7.2: Activity concentrations of ^{226}Ra (Bq kg^{-1} dw) in various plant types categorised by location within the Vodnyi site. Median values are plotted within the boxes as horizontal bars, the error bars express the range, numbers of data points per category are shown.



(a) **(b)**

Figure 7.3: Activity concentrations (Bq kg^{-1} dw) in various plant types categorised by location within the Vodnyi site. Median values are plotted within the boxes as horizontal bars, the error bars express the range in levels, numbers of data points per category are shown: (a) ^{238}U and (b) ^{232}Th.

In the case of ^{238}U, there appears to be a significant under-prediction at all 3 contaminated sites (this discrepancy is increased if it is assumed that some of the available soil activity concentrations are reported as ash weight as opposed to dry weight[31]). Observed values are, in some cases, several orders of magnitude above those predicted. Only the ^{238}U activity concentrations for “shrubs” at the Otvally site appear to give reasonable estimates of actual values measured in vegetation. A similar conclusion can be drawn from the inter-comparison of predicted and observed activity concentrations of ^{232}Th in plants. The predicted values are consistently lower than observed values, by more than an order of magnitude, and in extreme cases several orders of magnitude. In contrast, ^{226}Ra activity concentrations predicted for grass/herb appear to compare favourably with empirical data – predicted and observed values are consistently of a similar order of magnitude. In contrast, the predictions for shrub are high, at several orders of magnitude above observed values (this discrepancy is decreased if it is assumed that some of the available soil activity concentrations are reported as ash weight as opposed to assuming that they were reported as dry weight although this would decrease the level of agreement for grass/herb).

Table 7.3: Comparison between activity concentrations of ^{238}U, ^{232}Th and ^{226}Ra in vegetation predicted using FASSET CRs with empirical data for sites in the Vodnyi area of Komi AR

Location	^{238}U (Bq kg^{-1} dw)			^{232}Th (Bq kg^{-1} dw)			^{226}Ra (Bq kg^{-1} dw)		
	Predicted		Observed	Predicted		Observed	Predicted		Observed
	Grass/herb	Shrub	Vegetation[a]	Grass/herb	Shrub	Vegetation	Grass/herb	Shrub	Vegetation
Krokhal	0.05-1.1	0.3-6.9	15-2000	0.2-0.3	1.6-2.4	59-1000	660-5500	23000-190000	78-490
Obzhig	22-52	140-330	2100-3400	0.24	1.9	59-2100	560-1100	19000-39000	670-2100
Otvally	16-83	100-520	180-6800	0.26	2.1	220-4100	2800-6600	96000-220000	210-4800

[a]Vegetation includes grass, herbaceous species and trees

The FASSET ^{226}Ra, ^{238}U and ^{232}Th CRs for grass has been derived using the FASTer model [Brown *et al*. 2003a][32]. Data compiled during the case study could be used to derive CR information that supplements the original model-derived look-up table values with empirical information for these three radionuclides.

There is evidence from the study of Titaeva *et al*. [1978] that ^{226}Ra and ^{222}Rn are not in secular equilibrium in the tissues of plants. In some cases, the activity concentrations of ^{222}Rn in plant tissues exceed those of ^{226}Ra by one order of magnitude. This provides further support for a suggestion to separate the dose conversion coefficient for ^{226}Ra from that of ^{222}Rn by altering the rule by which the dose conversion coefficient of the daughter is amalgamated with the parent nuclei when the half-life is <10 days to a system where this combination occurs only when the daughter half-life is <1 day (see Chapter 5).

Species from the genus *Vaccinium* have been predominately used in the derivation of plant CR values for the shrub reference organism [Brown *et al*. 2003a]. There can be considerable variation in the activity concentrations for some naturally occurring radionuclides within the plants [Titaeva *et al*. 1978]. For example considerable differences in the levels of Th isotopes occur between the bark, xylem and branches of several tree species, including willow (*Salix caprea*), pine (*Pinus silvestris*) and larch (*Larix sibirica*). Guidance may be required on the sampling/consideration of representative plant parts in this case.

[31] It is unclear in some reference if soil data were reported as dry or ash weight, when this was the case initial calculations were performed assuming dry weight.

[32]The FASTer model was used to provide look-up table values for semi-natural ecosystems if measured values were not available.

7.5.2 Fauna

A possible independent data set has been identified in the report by Sazykina *et al.* [2003]. Data were collated from selected contaminated sites at Komi including, burrowing and herbivorous mammals for the radionuclides U, Th, Ra and Po, drawing on several Russian language publications. Unfortunately, it was not possible to evaluate the degree of overlap between the data-set used in the derivation of CR values and this potentially useful new dataset that might be used to test the results of the FASSET method application, because the source data were from Russian language publications. It has therefore been assumed that the datasets are not linked, without evidence to the contrary, but that a caveat should be placed on any conclusions drawn because of the uncertainty surrounding the information source. Furthermore, the data are not provided with information on detailed sample site location.

The test data set only comprises three individual data values from two samples (it has not been possible to identify whether the samples are bulked or individual animals). Observations for Tundra vole (*Microtus oeconomus P.*) provide muscle (assumed here to equate to whole body[33]) activity concentrations of 54 - 90 Bq kg^{-1} fw ^{226}Ra and 0.28 Bq kg^{-1} fw ^{238}U. The samples are believed to be from the Krokhal site (in the Vodnyi area).

Using the activity concentrations in soil at Krokhal presented above ^{238}U and ^{226}Ra activity concentrations (fw) were predicted for a "burrowing mammal". Following the example of Beresford *et al.* [2004], in cases where two reference organism categories were relevant, the category providing the more conservative CR value was used. The predicted values are 500 - 4100 Bq kg^{-1} fw ^{226}Ra and 0.006-0.14 Bq kg^{-1} fw ^{238}U. Application of FASSET CR values to measured soil data produces levels that are slightly below the observed value for ^{238}U although the upper bound of the predicted range provides a very reasonable estimate of empirical values. If it is assumed that some of the available soil activity concentrations are reported as ash weight as opposed to dry this under-prediction would be increased. Predicted activity concentrations of ^{226}Ra in small, herbivorous burrowing rodents are more than one order of magnitude above the observed range. The predictions derived from the FASSET methodology for this very limited data set may be considered adequate although further comparison is clearly needed before any definitive conclusions can be drawn.

Where CR values do not exist, FASSET guidance needs to be more specific. There is some general advice in Brown *et al*. [2003a] on management of information gaps but this needs to be refined, possibly in the form of step-wise advise, such as stage 1 apply a best option method if unsuccessful go to step 2 etc.. With respect to the use of the highest available CR for a selected radionuclide, it might be more sensible to make a distinction, at least, between fauna and flora, e.g. if no CR data for grass are available then guidance is given to use data for other vegetation categories referring to all CR values.

The CR values for ^{238}U, ^{232}Th and ^{226}Ra to plants require further refinement. CR values for natural decay series radionuclides to fauna could only be tested to a limited extent and, therefore, further consideration of these values is recommended. Furthermore, collation of NORM transfer data for other sites would be beneficial.

7.6 Testing dose rate data

Taskaev *et al.* [2003] reported that dose rates at the ground surface were typically 5 μGy h^{-1} with a range of 0.5 to 12 μGy h^{-1} at the Krokhal site. The range of external dose rates at the Otvally site appears to be higher at 2 to 60 μGy h^{-1}. Other data have been collated into a database by Sazkyina *et al.* [2003] drawing on Russian language studies and reviews; external

[33] This assumption is likely to have underestimated whole body activity concentrations (eds).

dose rates for unspecified locations in the Komi Region vary between 0.3 μGy h^{-1} to 80 μGy h^{-1}. The highest recorded external dose rates of 40 to 80 μGy h^{-1} are believed to have been recorded in Vodnyi, most probably (considering the activity concentrations in soil) at the Krokhal site[34].

To transform these data into a format compatible for comparison with data generated by the FASSET methodology, some consideration must be given to the details of measurement. In most cases, the empirical data are cited as dose rates, presumably absorbed dose rates to air at a certain height above ground level (this latter information is often not specified but it can be assumed that kerma rates probably do not vary much in the range 0-1 m, which are typical air kerma measurement heights). A conversion is required to derive the absorbed dose rate in a specified tissue or geometry from the air kerma. Earlier studies [Golikov & Brown 2003] show that for photon energies in the range 0.1 to 1 MeV and for a spherical geometry of radius 15 cm (representing, for example, a small mammal) at the soil air interface, the quotient of the average dose rate in the geometry, D_{geom} and the air kerma rate K_{air}, i.e. D_{geom}/K_{air} does not differ greatly from unity (*c*. 0.8 in this case). For lower energy photons, however, the quotient can deviate markedly from unity. There are large uncertainties generally associated with the dose rate data, and a large proportion of the external dose rate is likely to arise from gamma rays greater than 0.1 MeV (thereby applying a factor of 1 would not to lead to any great error in any case). Therefore, we assumed that measured air kerma rates were equivalent to the absorbed dose rates in the whole bodies of reference plants and animals. We acknowledge that this ignores the contribution of β-emitters (although these would not contribute significantly in this assessment).

External (unweighted) absorbed dose rate data have been generated for the Krokhal and Otvally sites, using appropriate soil activity concentration data, for all reference organism types. These data are presented in Figures 7.4 and 7.5 (see Section 7.1). Predicted external dose rates at Krokhal vary between 0.4 μGy h^{-1} and 17 μGy h^{-1}. These predicted data match closely with the data provided by direct measurement. The external dose rates predicted for Otvally range between 2 and 20 μGy h^{-1}. In this case, the match between predicted and measured values is again good although the highest observed values exceed predictions. According to the predictions, ^{226}Ra contributes almost all the external dose rates at these sites.

External dose rates to soil invertebrates in surface soils at Krokhal have been estimated to be approximately 10 mGy d^{-1} [Taskaev *et al*. 2003]. These values compare less favourably to estimates using FASSET methodology of 1 to 17 μGy h^{-1} or 0.024 to 0.41 mGy d^{-1} (woodlouse and earthworm at three sites). Reasons for this discrepancy are unknown, Taskaev *et al*. do not present methods of estimating exposure.

7.7 Derivation of total dose rate

With respect to the interpretation of reference organism dose rates predicted using FASSET methodology the assessors considered that it was also important to place these values in the context of background dose rates. This is consistent with Pentreath [2002] who stated that, arguably, two points of reference may be used for the purpose of assessing the potential consequences of exposures to radiation on non-human biota. These are (a) natural background dose rates and (b) dose rates known to have specific biological effects on individual organisms.

The FASSET framework currently gives very limited guidance on the application of methodology in cases concerning TENORM or enhanced levels of NORM. Here a decision

[34]A single values of 400 μGy h^{-1} was recorded in the database but there is some doubt over whether this value is correct.

was made to consider only those radionuclides that exhibited significant activity concentrations over background levels. In other words we attempted to derive an activity concentration, and thereafter a dose rate, "excess". This value will then be used in conjunction with the FASSET FRED database to make prediction concerning biological effects[35].

The following radionuclides were initially deemed important at the Komi sites: ^{238}U, ^{232}Th, ^{234}U, ^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po, ^{232}Th, ^{228}Ra, and ^{228}Th. All other radionuclides in the ^{238}U and ^{232}Th decay series were either included in the dose conversion coefficient of their parent radionuclide or were not reported in sufficient detail in the available literature to justify inclusion. In the case of ^{228}Ra, no dose conversion coefficients were available for use and this radionuclide could therefore not be considered further using the FASSET methodology. The activity concentrations of ^{226}Ra and ^{228}Th were sufficiently above expected background levels that no correction for background was made. In some cases, such as for rodents at Krokhal, ^{238}U and ^{232}Th was at background levels and these radionuclides were therefore not included in the derivation of the dose rate excess.

7.7.1 Dose rates for flora and fauna

Soil activity concentrations from studies in the Vodnyi area of Komi (summarised in Table 7.2) have been used to derive unweighted absorbed dose rate estimates for reference organisms through the application of look-up table CR values and the application of appropriate dose conversion coefficients (Figures 7.4 – 7.6). No internal DCC values are given within the FASSET framework for terrestrial plants.

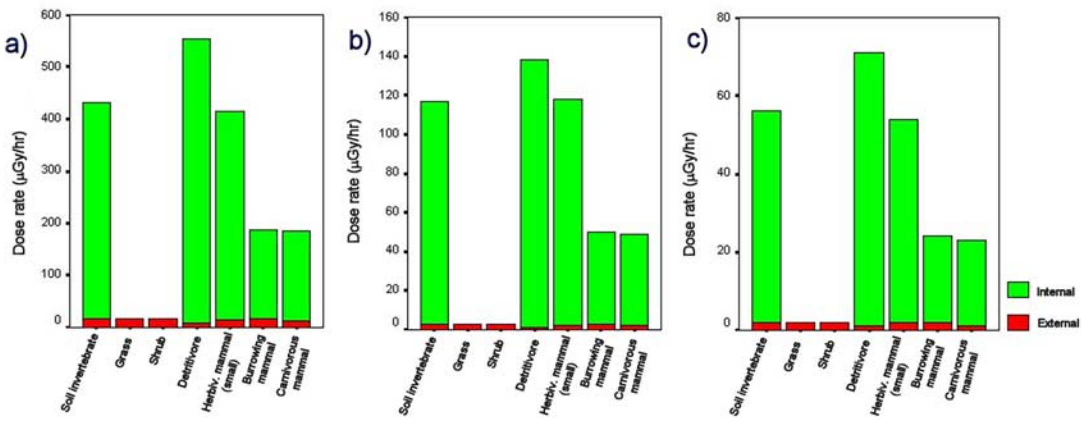


Figure 7.4: Unweighted dose rate excesses (µGy h^{-1}) for reference organisms based on soil data from three studies (a-c) at Krokhal (Vodnyi area within the Komi Republic).

Figure 7.4a is based on soil activity concentrations in the upper 15 cm of the soil, whereas Figures 7.4b, 7.5a, and 7.6a represent activity concentrations (ash weight) in the "rhizosphere layer". All studies were conducted in the late 1970s [see Titaeva *et al.* 1977; 1978]; Taskaev *et al.* 2003]. Figures 7.4c, 7.5b, and 7.6b, however, are based on activity concentrations (presumably dry weight) in surface soil (0-15 cm) from 2001 [see Taskaev *et al.* 2003].

The overwhelmingly predominant component of radiation exposure arises from the presence of internally distributed (alpha-emitting) radionuclides. Weighted doses have been estimated assuming an RBE for α-radiation of 10. The resultant weighted internal dose rates are approximately 10-fold higher than those shown in Figures 7.4-7.6 because of the predominance of internally incorporated alpha emitters. Maximum internal weighted dose

[35] The FRED database is not presented in such a way that collated effects can be compared with 'excess' dose rates (eds).

rates of 8500 μGy h^{-1} were estimated for detritivores at Otvally; dose rates of 6700 μGy h^{-1} were estimated for herbivorous mammals at this site.

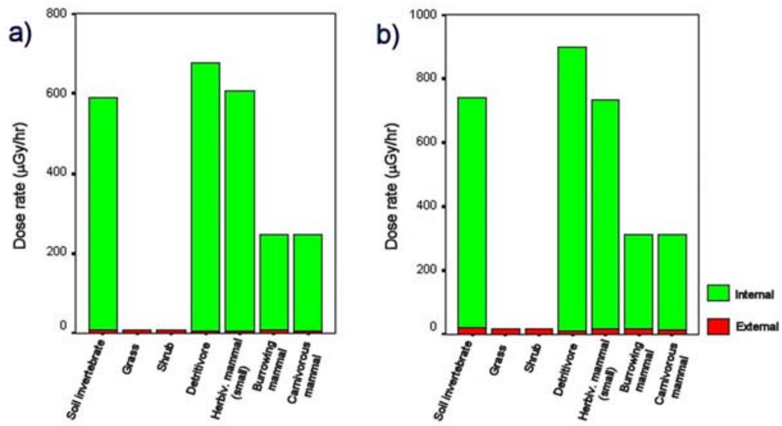


Figure 7.5: Unweighted dose rate excesses (μGy/h) for reference organisms based on soil data from two studies (a & b) at Otvally (Vodnyi area within the Komi Republic).

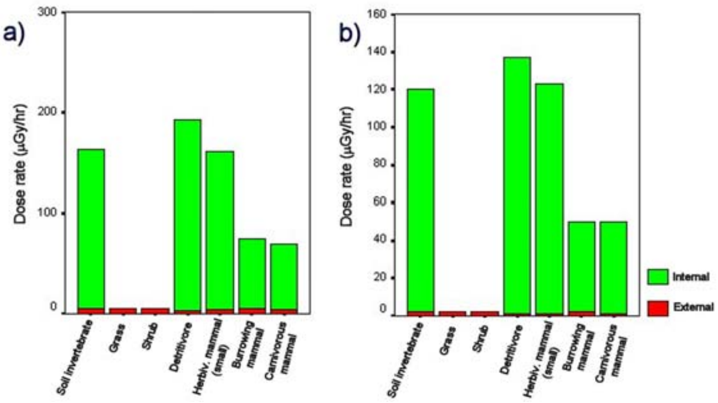


Figure 7.6: Unweighted dose rate excesses (μGy/h) for reference organisms based on soil data from two studies (a & b) at Obzhig (Vodnyi area within the Komi Republic).

Dose rates arising from external exposures are comparatively insignificant because most of the radionuclides considered are predominantly alpha-emitters with an assumed external DCC close to zero. In all cases, detritivores are exposed to the highest dose rates closely followed by soil invertebrates and small herbivorous mammals. For the main internal dose-forming radionuclides of interest (^{210}Po, ^{210}Pb, ^{226}Ra), CR values are highest for these biota groups. The ^{210}Po, ^{210}Pb CR values for detritivore and soil invertebrate, for which transfer data were absent (with the exception of ^{210}Pb to soil invertebrates), have been set equal to those for small herbivorous mammal.

7.7.2 Dose rates for rodents – herbivorous/burrowing mammals

Contamination-induced effects on tundra voles (*Microtus oeconomus*) have been extensively studied at several sites within the Komi AR [Sazykina *et al*. 2003]. As discussed previously, measured external absorbed dose rates at the most contaminated sites in the Vodnyi area were up to 80 μGy h^{-1} although only 3 hectares seems to have such elevated levels.

Implicit within FASSET assessment methodology [Brown *et al*. 2003a] is the guidance that measured data should be used, where possible (Figures 3.1-3.3 in Brown *et al*. [2003a]). This is explicitly considered for activity concentrations but should apply equally to measured dose rates. It is therefore preferable to use the measured external dose rate data in this case. FASSET dose conversion coefficients can be used to derive the internal dose rate to voles arising from a whole body concentration of 90 Bq kg^{-1} (fw) ^{226}Ra at this site (this is the upper value documented by Sazykina *et al*. [2003] as considered above). As no information was available on the body activity concentrations of ^{228}Th, ^{210}Pb and ^{210}Po, these values were derived by applying FASSET CR values to soil concentrations from the Krokhal site and thereafter applying appropriate dose conversion coefficients. Using this approach, an internal dose rate of 48-360 μGy h^{-1} was derived for a small herbivorous mammal.

The total (unweighted absorbed) dose rate is therefore approximately 130 - 440 μGy h^{-1} and is dominated by the internal component of radiation (mainly ^{210}Po). This range is similar to the herbivorous mammal (small) values derived for Krokhal based on soil concentrations alone (approximately 60 – 400 μGy h^{-1} as shown in Figure 7.4). This similarity is because ^{210}Po, the main contributor to internal dose, was derived using CR values in both instances (as no measurements were available for this radionuclide). Weighted doses for herbivorous mammals at Krokhal were estimated to be in the range 560-3700 μGy h^{-1}.

All other radionuclides in the ^{238}U and ^{232}Th decay chains have been measured to be (e.g. ^{238}U concentration = approximately 50 Bq kg^{-1}), or are assumed to be (e.g. ^{232}Th), at background levels and have not been included in the dose rate estimates. To calculate an absorbed dose rate excess the background components of ^{226}Ra, ^{210}Po and ^{210}Pb internal doses and external dose rate should be subtracted in our calculations. However, these components are negligible compared to the field values and their contribution to the total unweighted dose can be ignored (see below for an explicit appraisal of expected background dose rates). Although the internal dose conversion coefficient for ^{226}Ra integrates a component for ^{222}Rn, no explicit consideration of potential build-up of ^{222}Rn in the burrows of these small mammals and subsequent inhalation doses has been made. In a separate study, Sazykina *et al.* [2003] reported total dose rates at these sites in the order of 350 mGy y^{-1} (40 μGy h^{-1}) with a large component arising from ^{222}Rn exposure.

Practical application of the FASSET methodology at Komi has shown that combined data sets are often required to achieve the most realistic dose estimate. Where possible, data that require the least number of assumptions in the derivation of dose rates should be used. Further guidance should be provided by FASSET regarding the use of field data. More specifically - field measurements of gamma air kerma rate or data from thermo-luminescent dosimetry (TLD) should be used where possible (in this case some advice may be required with regard to conversion factors for selected geometries).

The FASSET methodology does not explicitly address ^{222}Rn inhalation doses. Since radon is a radioactive gas, the framework needs to address the air-lung pathway in addition to whole-body dose rates.

7.7.3 Background dose rates

Information on background dose rates to selected reference fauna for typical European environments has been compiled within FASSET by Pröhl [2003]. Typical external dose rates were 0.09 to 0.4 mGy y^{-1}. Typical internal dose rates range between 0.26 and 0.44 mGy y^{-1} for selected measurement endpoints (generic muscle, grain, leafy vegetables and roots). Whicker & Schultz [1982] reported total dose rates (including irradiation from cosmic rays as well as external exposure from terrestrial gamma rays and internal sources[36]) of 1.2 mGy y^{-1} as being typical for terrestrial vertebrates.

Terrestrial background dose rates presented in Pröhl [2003] are not presented in a particularly useful format. Internal dose rate data are presented for muscle (presumably human) and a number of agricultural products. No summarised information, such as total dose rates for a specified organism, is provided. We recommend a modification of this table for the sake of greater compatibility with the assessment methodology. Exposure from cosmic radiation should be included for completeness.

7.8 Testing effects data

7.8.1 Rodents – herbivorous/burrowing mammals

To consider effects, we have compared our estimated dose rates with the chronic effects data compiled within the FASSET radiation effects database - FRED [Woodhead & Zinger 2003]. Significant effects were observed above dose rates of 100 μGy h^{-1} in the FRED as database with a "clear" effect occurring at 1 mGy h^{-1}. The upper end of the weighted dose rate (assuming RBE of 10 for α-radiation) range predicted using the application of FASSET methodology of 3700 μGy h^{-1} is above that of *circa* 1000 μGy h^{-1} at which significant reductions in mammalian lifespan can be expected [Woodhead & Zinger 2003].

According to the review of Sazykina *et al.* [2003] the following radiation effects on the morbidity of rodents were actually observed in the field at contaminated sites in Komi (believed to be Krokhal at Vodnyi):

- (i) Changes in blood indicating chronic radiation sickness[37].
- (ii) High (up to 100%) levels of infestation with parasites
- (iii) Many low-fat (thin) animals in populations, low weight of liver in young mice, abnormalities in liver of adult mice

Effects on reproductive capacity have also been documented at sites within Komi, notably:

- (i) At sites in Krokhal and Otvally (Vodnyi region), decreases in testes weight and pathological changes in the seminiferous tubules of young male voles compared to control sites. The changes suggested impacts on spermatogenesis for voles in the contaminated zone [Taskaev *et al*. 2003].
- (ii) Numbers of females involved in reproduction and numbers of embryos per female were approximately 50 % of those in the control [Sazykina *et al*. 2003].

Cytogenetic studies have been conducted on the red bone marrow of voles from Krokhal and Otvally. The incidence of structural chromosome aberrations in contaminated areas was approximately 3 times greater than those observed in control plots. Genome aberrations, involving the presence of one excess chromosome, were elevated at both Otvally and Krokhal [Taskaev *et al*. 2003].

The observed effects are broadly as expected at the estimated weighted dose rates (assuming RBE=10) from the summaries of the FRED database presented in Woodhead & Zinger [2003].

[36] Internal sources also included ^{87}Rb and ^{3}H not considered in the background internal dose estimation performed in FASSET.

[37] According to the information provided by Sazykina et al. (2003), some of these observations were made at sites with environmental dose rates somewhat lower than the maximally contaminated areas.

However, radiation exposure will not be the only environmental stressor. The chemical toxicity of heavy natural radioisotopes also contributes to, and in some case dominates, the health effects in animals [e.g. Jones *et al*. 2003]. Gilman *et al*. [1998] report that the lowest observed adverse effects level (LOAEL) for experiments involving the oral administration of U to rats was 0.06 mg kg^{-1} body weight d^{-1}. From USDoE [2002] and ICRP [1979] this can be estimated to result in a whole body concentration of 0.015 mg kg^{-1}. Sazykina *et al*. [2003] report activity concentrations in the tissues of small mammals from Komi in the range 0.02-0.2 mg kg^{-1}. Consequently, the effects being observed at some sites in Komi are as likely to be attributable to the chemical toxicity of U as they are to the effects of radiation[38].

7.8.2 Soil invertebrates

The weighted dose rates received by soil invertebrates at the Krokhal site estimated using the FASSET methodology fall in the range 540-4100 µGy h^{-1}. It is not possible to comment upon what effects may be predicted in this dose range because of the limited nature of summary conclusions from the FRED database [Woodhead & Zinger 2003]. Furthermore, results from studies of soil invertebrates in Komi are included in the FRED database so independent testing is not possible. A LOEDR of 10 mGy h^{-1} could be suggested for mortality from FRED based on 9 references.

Studies at the Krokhal site, Vodnyi and its environs, summarised by Taskaev *et al.* [2003], have reported various effects on soil invertebrates including:

(i) Morbidity effects including histological changes in the outer integument and mid-gut epithelium of earthworms compared to control sites. Furthermore, an increase in mucus-producing cells were noted for populations of earthworms in contaminated areas.

(ii) Effects on reproductive capacity involving decreases in population sizes and fecundity (non-specific) of earthworms compared to controls.

(iii) Reduced populations for a variety of soil invertebrates including beetles and spiders at contaminated sites compared to control sites.

In view of the substantial lack of information on dose(rate)-effect relationships for soil invertebrates in FRED, the studies conducted at Komi may prove valuable in augmenting existing datasets, although some of the Komi effects data have already been included within FRED. Whilst at some sites the wider usefulness of data obtained may be compromised because of the chemical toxicity of the radionuclides concerned there are sites were $^{226,228}Ra$ are the main contributors to dose and hence chemical toxicity may not be an issue.

7.8.3 Flora

A limited number of studies have been conducted involving the observation of contamination-induced effects on tufted vetch (*Vicia cracca*) – a species that can be technically placed under the reference organism grouping "Grass". However, the FASSET methodology does not allow the estimation of internal doses to plants. The external dose rates to plant reference organisms are all below 20 µGy h^{-1}. The dominance of alpha emitters at the Komi site will mean that internal dose rates will be much higher than this. It is likely that total doses to plants are above those at which effects on reproductive capacity have been observed in vegetation. According to the FASSET summary table [Woodhead & Zinger 2003], a decrease in the seed weight of a "herb" species has been observed at 5.5 µGy h^{-1}. It is also possible that they would be above the 40 µGy h^{-1} level at which mutation (specifically in changes in micro-satellite DNA) has been seen to increase.

[38]Note the high U concentrations reported here conflict with the observation above that ^{238}U activity concentrations were at 'background' levels. We are unsure as to the location of at which the total U measurements was made (eds).

Several field observations have been made in Komi including [Sazykina *et al*. 2003]:

(i) Mortality effects where survival of tufted vetch under irradiated conditions was determined at 45 % (from maximum number of sprouted vegetation) compared to 83 % in a control for the initial vegetative period

(ii) Cytogentic effects including a greater number of (a) variable mitosis and (b) anaphases with aberrations in irradiated populations compared to controls. In contrast the frequency of chlorophyll mutations was actually lower in irradiated populations compared to controls.

The mortality effects observed may be due to the chemical toxicity of U (and Th) rather than exposure to ionising radiation.

7.8.4 Summarised dose-effects information

Calculated dose rates are presented in Figure 7.7 and put into context with observations form the FRED database.

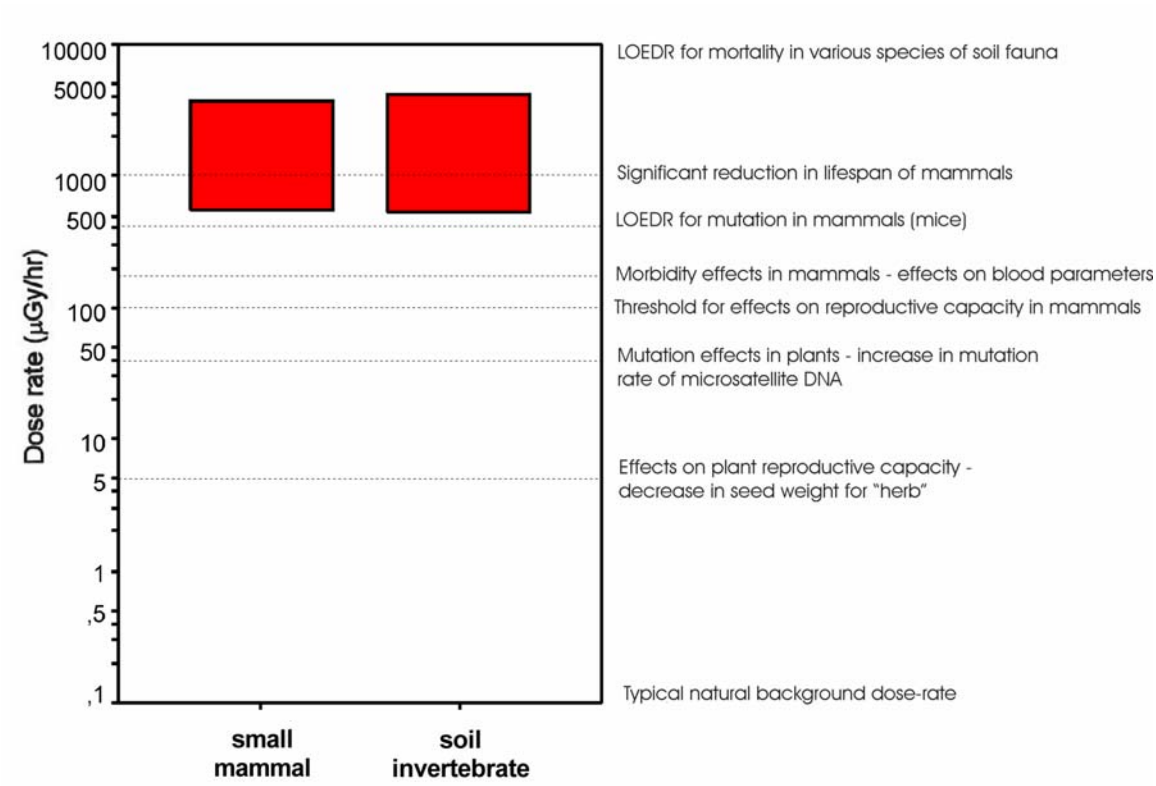


Figure 7.7: "Excess"[39] weighted (assuming RBE=10) absorbed dose rates predicted at the Krokhal site for 2 categories of reference organism compared to typical natural background (for terrestrial vertebrates) and dose rates at which effects might be expected in selected umbrella endpoints for the same biota categories [Woodhead & Zinger 2003]. The calculated dose rates for small mammals are derived from external dose rate measurements (in air), biota activity concentrations for ^{226}Ra, and the application of FASSET CR values to soil activity concentrations for other radionuclides.

[39] Guidance should be given that the total dose rates are compared to summaries from the FRED database not excess dose rates only (eds).

7.9 Conclusions

Aside from the fact that explicit guidance for cases with TENORM are not provided, the application of the FASSET methodology has met with few practical difficulties. The assessment was straight-forward and assessment parameters are generally available or can be easily derived (although in this case there was some doubt as to the method of reporting soil activity concentrations in some references and this has contributed to uncertainty in our assessment). Nevertheless, the following problems have been encountered during the case study:

(i) Evidence showing that the radionuclides ^{226}Ra and ^{222}Rn are often not in secular equilibrium in the tissues of plants leads to a suggestion to separate the dose conversion coefficient for ^{226}Ra from that of ^{222}Rn by altering the rule by which the dose conversion coefficient of the daughter is combined with the parent nuclei, i.e. when the progeny half-life is >1 day the daughter is treated separately (this is in agreement with approaches used by other groups working in this field for instance Amiro [1997] and Golikov & Brown [2003]).

(ii) Advice on the management of information gaps, specifically in relation to CR values, needs to be more helpful. A form of step wise advise, such as stage 1 apply a best option method if unsuccessful go to step 2 etc. is recommended. With respect to the selection of surrogate CR values where data are available for the same radionuclide and other organism groups a recommendation is made to select CRs for related biota types, e.g. if no CR data for grass use data for other vegetation categories before using all CR values.

(iii) There was lack of agreement between observed and predicted CR values for ^{238}U, ^{232}Th and ^{226}Ra to plants. CR values for natural decay series radionuclides to fauna could only be tested to a very limited extent and therefore further consideration of these values is recommended. In this respect, further collation of NORM transfer data for other terrestrial sites would be beneficial.

(iv) Application of the FASSET methodology at Komi has shown that combined data sets (i.e. observed and predicted values) are often required to achieve the most realistic dose estimates. Where possible, data that require the least number of assumptions in the derivation of dose rates should be used.

(v) It recommended that further guidance should be provided by FASSET regarding the use of field data, e.g. field measurements of gamma air kerma rate or from thermo-luminescent dosimeters should be used where possible (in this case some advice may be required re conversion factors for selected geometries).

(vi) A prerequisite for the robust interpretation of calculated dose rates is a comparison with natural background dose rates. Terrestrial background dose rates presented in FASSET [Pröhl 2003] are not presented in a particularly useful format. A modification of the relevant tables, for the sake of greater compatibility with the assessment methodology, is recommended. Exposure from cosmic radiation should be included for completeness.

(vii) In view of the substantial lack of information on dose rate-effects relationships for soil invertebrates in the FASSET effects database, the studies conducted at Komi may prove valuable in augmenting existing datasets assuming sites where chemical toxicity does not dominate are available.

8 Discussion and recommendations

The process of applying the FASSET framework to different case studies has been valuable in highlighting areas of improvement for consideration during the ERICA project. In this section we discuss issues associated with the actual application of the framework, provide feedback on those areas of the assessment which worked and those where we encountered problems. To our knowledge this deliverable represents the most comprehensive assessment of the application of a radiological environmental impact assessment approach.

8.1 Application of the FASSET methodology

The different assessments did not all consider implementation of the same parts of the FASSET framework, some case studies concentrating on the 'assessment phase' (e.g. Chernobyl) whilst others tried to implement the entire FASSET framework (i.e. Larsson *et al*. 2004) in a step by step manner (e.g. Loire River). Some assessors considered Brown et al. [2003] ('the assessment handbook') as the definitive guide and others Larsson et al. [2004] ('the framework'). This led to different interpretations of what to do and different problems encountered. Some assessors felt that the framework guidance was relatively clear and easy to implement (these tend to be those involved in the development of the FASSET framework) others experienced considerable difficulties in interpreting the many reports (these tended to be assessors new to the FASSET methodology). Arguably, this may lead to the suggestion that future case study applications should involve those 'new to FASSET', however, we feel there is merit in the involvement of both groups as they find different sets of 'problems'.

Brown *et al*. and Larsson *et al*. begin the assessment with a 'problem formulation' and 'source characterisation' stage respectively. Whilst differing slightly, these are, in essence, the same thing. Both require considerable data collation (e.g. source term characteristics, chemical speciation of releases, solubility of element, isotopic dilution in receptor ecosystems, reaction of radionuclides with biological ligands, potential chemical toxicity etc.). However, there is little, if any, guidance on how to acquire this data and how to interpret it once obtained. Furthermore, FASSET gives no guidance on what to do if this information cannot be obtained. The Loire River assessment gives the best examples of these problems. We suggest that some of the specified information (<u>if really required</u>) could be provided as default values within the ERICA assessment tool if guidance on their interpretation is given (e.g. chemical analogue list, information on biological activity, solubility constants, energies of radiation).

The presentation of CR and DCC values for reference organisms led to some confusion. One assessment assumed the 'representative species' selected, which in FASSET were used to derive a range of DCC values by providing example geometries, were the 'reference organisms'. It would perhaps have been more appropriate (in FASSET) to define a series of geometries for the estimation of DCCs without identifying them as particular species as this leads to misinterpretations by both stakeholders and operators. Within ERICA it has already been agreed that external doses will be estimated for user defined geometries.

There were some anomalies in the FASSET DCC values for terrestrial ecosystems which resulted in a modified approach being used[40] and assumptions had to be made with regard to

[40]FASSET terrestrial organism external DCCs are for photons only, no contributions from β-emissions or bremsstrahlung being included. In the case of (e.g.) ^{90}Sr, a β-emitter, the only photons contributing to estimated doses are those emitted by its daughter, ^{90}Y, with energies in range 2-18 keV. The range of these photons in air is much longer than that in soil and, the dose for animals "on-soil" is due to photons coming from a larger area that for the case of "in-soil" exposure. This was unclear in the relevant deliverables and led to confusion (eds).

freshwater DCCs which were presented as negative values (see section 2.3.1). These anomalies need to be resolved. One of the assessors noted the requirement to assume that external DCCs for terrestrial systems within FASSET are to be applied to dry weight activity concentrations as the framework does not specify this. It is now apparent that all terrestrial assessments assumed this, but that the DCCs were calculated on a fresh weight soil basis (although this is unlikely to make much difference to the estimated doses).

Issues were raised in a number of assessments regarding the lack of equilibrium (FASSET equilibrium CR values have been used here). When conducting assessments for sites discharging radioactivity, FASSET provides no guidance on temporal integration, for instance is it acceptable to use water concentrations determined on a given day or should annual averages be used (as suggested by Copplestone *et al*. [2001]). The comparison of predicted and observed activity concentrations in vegetation at terrestrial sites within the Sellafield assessment showed a tendency to under predict possibly because these site receive continual inputs via sea spray or tidal inundation. Predictions conducted for the Loire River were based upon estimated water activity concentrations from one years discharges. However, the river at the assessment site has received discharges from a number of nuclear power plants over many years and will also have inputs from the Chernobyl accident and nuclear weapons fallout. Therefore, it could be considered unreasonable to expect the FASSET methodology when applied to the discharges for one site in one year to reliably predict sediment and biota activity concentrations. This perhaps raises a fundamental question – how do we envisage the ERICA tools will be applied? Advice is needed on the use of equilibrium CR values in assessments and their limitations with regard to the potential lack of equilibrium. Both Brown *et al*. [2003a] and Larsson *et al*. [2004] acknowledge this issue and note that the assumption of equilibrium may not always be conservative, however, no guidance on how to address this is given. It is probably a realistic assumption to suggest that the ERICA tools will generally be required for application to sites which are discharging radionuclides to the environment and furthermore as a consequence have the legacy of historical discharges associated with them, as in some of the case study sites here.

The FASSET methodology presents a number of generic ecosystem types. However, for some of these there are comparatively few CR values provided (i.e. forests, wetland and brackish waters). The stakeholders associated with the Sellafield case study expressed the opinion that separation in these ecosystems was artificial and inappropriate (e.g. assessment of agricultural ecosystems within the FASSET would not include wild species (only farm livestock and crops)). As noted in the Chernobyl case study application, the derivation of CR values for animal reference organisms in semi-natural ecosystems did not exclude data obtained from forests or agricultural land (there were too few data for this) and may therefore be equally applicable to all three ecosystems. We suggest that some of the artificial divisions of terrestrial ecosystems (which we cannot actually conduct assessments for) be removed leaving assessment tools for wild species and agricultural species. Furthermore, there needs to be consideration of ecosystem interfaces (animals are not only associated with one type of ecosystem, e.g. birds may be associated with marine, freshwater and terrestrial ecosystems).

The FASSET semi-natural ecosystem CR values did not adequately predict activity concentrations for the saltmarsh site of the Sellafield case study. This is not surprising since the radioecology of saltmarshes is very specific and the data used to derive the FASSET CR values were not representative of this ecosystem. However, this site represents perhaps the most radiologically contaminated ecosystem in western Europe and has considerable nature conservation value. Therefore, the methodologies developed should be able to perform robust assessments for this ecosystem type receiving regulated discharges.

Two of the case study assessments considered NORMS/TENORMS. Whilst some radionuclides associated with NORM/TENORM sites are included within the FASSET framework there is no specific guidance on how to conduct assessments for such sites. For instance, in the absence of guidance, the assessors attempted to estimate the excess contribution to dose over that expected from background levels. Whilst Pröhl [2003] presents some information on background doses rates this is not presented in a form which is very useful, or appropriate, for application within an assessment. The stakeholders associated with the Sellafield case study also noted that they would like to see separation of the contribution of natural background (and other) sources from the contribution of dose caused by the site.

FASSET presents some guidance on how to conduct an assessment when a required radionuclide-reference organism CR is not available (see pages 44-45 of Larsson *et al*. [2004]). All assessors felt that this guidance needs to be more helpful. A form of step wise advice, such as 'stage 1 apply a best option method if unsuccessful go to step 2 etc.' is recommended. With respect to the selection of surrogate CR values where data are available for the same radionuclide and other organism groups the recommendation should be to select CRs for related biota types, e.g. if there are no CR data for grass then use data for other vegetation categories before using all CR values. However one group of assessors did not attempt to apply the existing guidance stating '*Even if it would provide reasonable results, it is difficult to explain and to follow and it might affect the overall acceptance of the estimation*.' As a consortium we need to agree an acceptable approach given the relative importance of this problem.

There is considerable uncertainty in many of the assumptions made with the FASSET framework and its predictions. The FASSET deliverables acknowledge this uncertainty and state that '*We, however, recommend that uncertainty analyses are always conducted as an integral part of the assessment'* (page 50 of Brown *et al*. [2003a]). However, whilst some general guidelines on how to conduct uncertainty analyses are provided the FASSET framework does not provide the ability to conduct uncertainty analyses (e.g. only mean CR values are presented). Only one assessor noted that they had considered the recommendation to conduct uncertainty analyses (although they could not actually do it). The presentation and description of uncertainty was raised by the Sellafield case study stakeholder group who felt the uncertainty evident from the comparisons of predicted and observed activity concentrations for some radionuclides in some biota reduced their confidence in the assessment.

In general, most assessors felt that the guidance on conducting the assessment needs to be much more user friendly. After consultation with potential end users of the ERICA tools we feel that it cannot be assumed that assessors would all have a significant understanding of radioecology. Therefore, the eventual output from ERICA should endeavour to be considerably more succinct than that from FASSET and provide clear guidance on how to conduct all stages of the assessment and be preferably presented as one document (and associated software).

8.1.1 Additional guidance/methodologies required

A number of additional detailed potential improvements of the FASSET methodology and guidance for consideration by the ERICA project were identified:

- The CR values (should) predict whole-body activity concentrations, however, many available measurements are for specific tissues. The inclusion of guidance on how to convert from tissue specific to whole-body activity concentrations would be useful. However, there may be a need to discuss what constitutes 'whole-body';
- The FASSET CR look-up tables have three different terrestrial mammal reference organisms. Given that there is limited data for many radionuclides and CR values for the different mammal groups are generally similar (within the overall variation) we suggest reducing this to one 'terrestrial mammal' reference organism;
- Available air kerma could be used to estimate external gamma dose rather than making predictions using DCCs; thermoluminescent dosimeters could be also be used and would have the advantage of including beta emitters. Alternatively, these

techniques could be used to compare with estimated doses to organisms. Guidance on how to do this would be required;

- There is variation in terminology and units used both within the FASSET deliverable and between FASSET and other radiological environmental assessment approaches (e.g. BAF, CF, BCF etc. are all used to describe what we have here referred to as CR; DCC can appear as DCF). To avoid confusion, this should be standardised in the case of transfer and dose conversion parameters we suggest CR and DCC are used as suggested by the Biota Working Group of the IAEA EMRAS programme (http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm);

- The presentation of all parameters should be clearly defined, for instance, do they relate to dry or fresh weights. Where possible this should be specified to be in the format analytical data is normally reported. The FASSET methodology does not state that k_d values should be applied to filtered waters;

- To apply marine k_d values within the FASSET methodology there is a need to access the original IAEA reference – information should be provided within in ERICA to avoid this. The exchangeable component of sediment activity concentration as oppose to the whole-sediment concentration is predicted by the advised k_d values, this is not made clear. The implication of this is that sediment total activity concentrations are underestimated or water activity concentrations overestimated by using k_d values. Conversely, the freshwater k_ds included within FASSET relate the dissolved radionuclide concentration in water to the suspended sediment concentration (i.e. they are not the same parameter as the marine k_ds);

- Guidance on spatial averaging (as applied in the Chernobyl case study out of necessity) could be considered;

- Based on observations made during assessments of ^{226}Ra in both the TeNORM/NORM case studies consideration should be given to altering the rule by which the dose conversion coefficient of the daughter is amalgamated with the parent nuclei when the half-life is <10 days to a system where this combination occurs only when the daughter half-life is <1 day.

8.2 Gaps in parameter values provided by the FASSET framework

The marine ecosystem CR look-up tables are complete for the radionuclides considered in FASSET [Brown et al. 2003a]; those available for agricultural ecosystems are virtually complete. However, those for the other ecosystems have considerable gaps; Table 8.1 and 8.2 summarise the available CR values for freshwater and terrestrial semi-natural ecosystems as included in the FASSET framework (there are many more data gaps for forest, wetland and brackish water ecosystems). The lack of CR values for freshwater ecosystems severely hampered the ability to conduct assessments for the Loire River and the Chernobyl cooling pond.

A number of radionuclides were identified in the case studies which were not included in the FASSET framework. These were:

- Sellafield ^{35}S, ^{60}Co ^{125}Sb (terrestrial);
- Loire River ^{54}Mn, ^{58}Co, ^{60}Co, ^{110m}Ag, ^{123m}Te, ^{124}Sb and ^{125}Sb (freshwater; brackish);
- Oil and gas platforms ^{228}Ra (marine);

- Chernobyl ^{91}Sr, ^{95}Zr, ^{95}Nb, ^{99}Mo, ^{103}Ru, ^{129m}Te, ^{132}Te, ^{133}I, ^{136}Cs, ^{140}Ba, ^{141}Ce, ^{144}Ce, ^{154}Eu (terrestrial assesment); ^{103}Ru, ^{132}I, ^{140}Ba, ^{140}La, and ^{141}Ce (freshwater assessment);
- Komi ^{228}Ra (terrestrial).

With the exception of the Loire River, the 'missing' radionuclides have been identified from measurements of media concentrations and not discharge information. We are also aware that ^{95}Zr, ^{95}Nb, ^{141}Ce and ^{144}Ce may be present on the Sellafield saltmarshes. Whilst there were a considerable number of radionuclides present in the Chernobyl case study area which could not be assessed, the requirement of including these within the ERICA tools should be considered. If they are unlikely to be required in assessment of regulated sites, do we need to included them within the ERICA tools? Conversely, although the missing radionuclides in the Sellafield, Loire River and Komi case studies have been identified for specific ecosystems it is likely that they will also need consideration for other receiving environments around the sites.

The FASSET framework does not present internal DCCs for terrestrial plants and this was noted as a deficiency in all terrestrial case studies. There is no justification for this omission presented in the FASSET methodology and agreement for inclusion within ERICA has already been reached.

The lack of consideration of amphibians and birds in FASSET terrestrial ecosystems severely restricted the ability to conduct a robust assessment for the Sellafield case study. These species are protected and contribute to the area being a nature conservation site. The lack of these organisms in FASSET was considered unacceptable to the stakeholder group; emphasised by a previous need to assess the impact of Sellafield on birds [Lowe 1991]. There is a general need to ensure that the reference organisms allow assessment for protected species within Europe.

The inclusion of inhalation doses from ^{222}Rn should be considered for burrowing mammals as it may contribute a large component of dose [Macdonald & Laverock 1998; Strong & Baker 1996].

Table 8.1: Available CR values for freshwater ecosystems in the FASSET framework (shading = CR value in FASSET framework) (adapted from Brown *et al*. [2003a].

Reference organism	H	C	K	Cl	Ni	Sr	Nb	Tc	Ru	I	Cs	Po	Pb	Ra	Th	U	Pu	Am	Np	Cm
Amphibians										■										
Birds											■			■						
Bivalve molluscs						■				■		■		■						
Crustaceans				■						■		■		■						
Fish						■				■	■			■	■	■	■			
Gastropods												■		■						
Insect larvae										■	■									
Mammals														■						
Plankton											■	■								
Phytoplankton						■				■				■		■	■			
Zooplankton						■				■						■	■			
Vascular plants				■		■				■	■	■		■	■	■	■			
k_d		■				■		■	■	■	■			■	■	■	■	■	■	■

Table 8.2: Available CR values for semi-natural pastures/heathlands ecosystems in the FASSET framework (shading = CR value in FASSET framework) (adapted from Brown *et al.* [2003a].

Reference organism	H	C	K	Cl	Ni	Sr	Nb	Tc	Ru	I	Cs	Po	Pb	Ra	Th	U	Pu	Am	Np	Cm
Soil Invertebrate (worm)	*	*																		
Lichen & bryophytes																				
Grasses	*	*		*	*		*	*	*	*				*	*	*	*	*	*	*
Shrub																				
Detritivores																				
Carnivorous mammals	*	*		*	*		*	*	*	*							*	*	*	*
Herbivorous mammals	*	*		*	*		*	*	*	*									*	*
Burrowing mammals																				
Bird egg	*	*																		

*CR value based on modelled estimate only.

8.3 Comparison of predictions and observations

8.3.1 Biota activity concentrations

In many instances the lack of available, independent, data which could be used to compare with the predictions of the FASSET framework hampered this task (especially for the Komi, Loire River, and oil and gas platforms case studies). Even within the Chernobyl case study, terrestrial biota activity concentrations were unavailable for radionuclides other than ^{90}Sr and radiocaesium. Participation within the IAEAs EMRAS programme will make additional datasets available in the future. However, comparisons of observed and predicted activity concentrations could be made for some radionuclide-reference organisms at all case studies.

Predicted activity concentrations in small mammals and sheep at the Sellafield saltmarsh assessment site were considerably higher than observations. This is likely to be the result of a lack of equilibrium (the animal will not spend all their time on the saltmarsh) and the low bioavailability of sediment associated radionuclides. The activity concentration of saltmarsh vegetation was under predicted, almost certainly the result of deposition of contaminated sediments onto vegetation surfaces during tidal inundation of the marsh. Predicted activity concentrations in small mammals at the sand dune assessment site were generally of the same order of magnitude as available measured data. The majority of observed data for sand dune vegetation were substantially higher that predicted activity concentrations. This is probably because the site receives continuing deposition via seaspray. Comparison of predicted and observed activity concentrations for the Sellafield agricultural assessment were hampered because most reported measurements were below detection limits. For all vegetation the FASSET CR values under predicted compared to observed data; again this may be because of continuing inputs. Perhaps we should have also applied the available CR values for chronic releases provided within the FASSET methodology at this site. For ruminant livestock predicted radiocaesium activity concentrations compared well with observed data, whilst ^{241}Am was substantially underestimated (this may in part be because tissue specific activity concentrations were compared with whole-body predictions).

Application of the FASSET k_d values for freshwater ecosystems under predicted radiocaesium sediment concentrations in the Loire River. However, this is likely to be because the assessment was made using estimated water activity concentrations based on one years discharges from all nuclear power operating sites on the river. The river at the assessment site has received discharges from five nuclear sites for two decades as well as deposition from the Chernobyl accident and global fallout. Furthermore, there is some confusion as to how the FASSET k_d values should be applied. However, the few radiocaesium activity concentrations reported in fish are 2-3 orders of magnitude lower than predicted, although the predictions of ^{14}C activity concentrations are considerably underestimated. In the Loire estuary the FASSET k_d values for ^{14}C and radiocaesium in marine ecosystems gave predictions which compared well with measured values; predictions using freshwater k_d values were 1-3 orders of magnitude below measured values. The Loire River assessment was limited by a lack of transfer parameter values in the FASSET methodology and a lack of available measurements with which to compare predictions. Furthermore, it was reliant on modelled water activity concentrations.

Only limited comparison of measured and predicted values was possible for the marine oil and gas platforms assessment. Comparisons of measured and predicted concentrations of ^{226}Ra and ^{210}Po were satisfactory. Predictions of ^{210}Pb in benthic crustaceans were 2 orders of magnitude higher than available measurements.

For the Chernobyl case study, it was possible to compare predictions to measured values for radiocaesium and ^{90}Sr in a range of FASSET animal reference organisms encompassing a wide range of species (e.g. in the case of mammals *Sorex* spp. to *Alces alces*). Predicted values were generally within the range of observed values which is encouraging. Furthermore, mean predictions were either close to observed means, or were conservative estimates being approximately an order of magnitude greater, lending confidence to the overall FASSET approach. However, for a number of individual animals, or site-species means, predictions were more than one order of magnitude lower than measured values. Given that FASSET predicts mean activity concentrations in biota and the uncertainties associated with estimation of media concentrations for this case study perhaps a under prediction by an order of magnitude for individual animals is to be expected. Some of the variation in agreement between predicted and observed data at different sites may be explicable as a consequence of variables, such as soil type or in the Chernobyl zone the density and nature of deposited fuel-particles (the predominant source of ^{90}Sr). Predicted ^{90}Sr activity concentrations in detritivorous invertebrates, made using the CR value for herbivorous mammals (given the lack of specific values for this reference organism and applying the FASSET guidance) were within the range of the limited data available. From the data available for the Chernobyl cooling pond, CR values could be derived for caesium for phytoplankton, molluscs, pelagic and benthic fish. These were all within one order of magnitude of those in the FASSET look-up tables (FASSET being conservative). The Chernobyl case study is one in which we could expect the assumption of equilibrium to be appropriate.

Predictions of the transfer of ^{238}U and ^{232}Th to plants in the Komi assessment were several orders of magnitude lower than observations. Predictions were made using CR values derived by the (unvalidated) FASTer model within FASSET rather than being based on empirical data. The transfer of ^{226}Ra to plants was in agreement with or higher than observed data. The limited comparison of predicted and observed activity concentrations in small mammals showed ^{238}U to be well predicted whereas ^{226}Ra was over predicted by an order of magnitude.

In summary the available FASSET CR values predict activity concentrations which compare well with observed data at some sites and for some radionuclides. At other sites there are considerable disparities between measured and predicted activity concentrations. The differences, both under and over predictions can be over several orders of magnitude in the worst cases. Firstly we have to ask what is 'acceptable?' and acknowledge that the FASSET CR values are means derived from empirical data or in some instances (unvalidated) model predictions. Some of the empirically derived CR values are also based on very limited data (e.g. those for actinides to mammals). Therefore, we should expect variation between predictions and observations (both under and over predictions). The values provided are also expected to predict activity concentrations 'anywhere', however, we know that there is systematic variation in radionuclide transfer depending upon, for instance, soil type or water chemistry. In some instances, the lack of agreement is because we are applying the FASSET outputs in a manner where we could perhaps not realistically expect them to predict available measured data. The application of CR and k_d values to the discharges during one year in the Loire River case study is an example of this. That is not to say that it was wrong for the assessors to attempt to assess the impact of an annual discharge from one site but the FASSET methodology could never predict observed environmental concentrations under this scenario. Similarly, the assessments conducted for sand dunes and saltmarshes within the Sellafield case study attempted to predict doses to specific species. This would be a requirement of the regulator for this site (see Copplestone *et al*. [2003]) but the FASSET framework was not intended to do this. The application of FASSET CR values to the Sellafield saltmarshes is also an example where we could not have really expect the tools provided within FASSET to provide realistic predictions. However, as noted above this is an important site (both radiologically and from a nature conservation perspective) where assessments are required.

8.3.2 Doses and effects

It was largely impossible to attempt to validate estimations of dose. For the Komi case study air kerma were available and these compared well with predictions of external dose using the FASSET DCCs. However, this does not test the assumptions we are making with regard to how biota interact with their environment (e.g. time spent in burrows, home ranges etc.). The assumptions with regard to, for instance, relative contributions of sediment and water to the external dose received by aquatic organisms, have a considerable influence on the results obtained. Information provided within the 'life history' datasheets is species specific and was of little value in the assessment.

In general, the assessments have not compared the predictions of dose from FASSET with those of other authors. However, this was done for pelagic fish within the Chernobyl cooling pond assessment and the estimated doses of previous authors were considerably higher than FASSET estimates. Reasons for this were not investigated but there is a general need to compare the predictions of the various 'doses to biota' tools now available and being used. This will be carried out within the IAEAs EMRAS programme in which ERICA is actively participating (see http://www-ns.iaea.org/projects/emras/emras-biota-wg.htm).

It was possible to try to compare observed (potentially) radiation induced effects with the summary tables of chronic exposure effects from the FASSET framework in a limited manner at the assessment sites in the fSU. The paucity of comparable data from observations and the FASSET summaries made this assessment difficult. However, for both the Chernobyl and Komi case study the observed effects were generally in line with those that would have been expected from the FASSET summary. At the Komi site the dose was predominantly due to internally incorporated alpha-emitters; an RBE of 10 for α-radiation was assumed. The chemical toxicity of uranium may have contributed to the observed effects at Komi; reported uranium concentrations in the tissues of small mammals are in excess of those expected to give toxic effects. Consideration of the chemical toxicity of uranium and thorium needs to be included with the guidance requested for assessments of NORM/TeNORM sites.

8.4 Recommendations

The following recommendations arise out of the case study application of the FASSET framework:

- ERICA has to carefully consider the scenarios it expects its tools to address.

- ERICA should be clear in its output when the methodology will, and will not, be applicable, considering:
    - equilibrium
    - site specific factors
    - historic discharges.
- The guidance produced by ERICA must be much more user friendly (and one document), it needs to clearly guide the assessor through all stages of the assessment providing:
    - interpretation of results at the various stages
    - guidance on how to proceed if required data or parameters are missing
    - guidance on how to take background exposure into account
    - guidance on chemical toxicity
- The ERICA methodology must not include assessment steps it does not provide guidance on how to conduct and interpret.
- A consistent terminology must be used.
- FASTer model requires validation if it is to be used
- The ERICA tool and other outputs presenting guidance must be consistent and their purpose and status clear.
- The reference organism list should encompass protected species
    - terrestrial birds and amphibians should be included.
- The additional radionuclides identified in the case study assessments need to be prioritised for inclusion within ERICA.
- The ecosystems and reference organisms considered by ERICA should be rationalised and consideration given to interface between different ecosystems.
- If ERICA is to continue the FASSET recommendation that uncertainty analyses be conducted as part of the assessment process it should provide the ability to do this.
- ERICA should consider providing guidance on how to present the assessment process and results to an interested but non-technical audience.

We have identified many aspects of the FASSET framework which could be improved during the development of the ERICA tools. However, we have to be realistic and accept that we are not going to be able to address all of these within the resources and timescale of the ERICA project. We therefore need to agree and prioritise the above recommendations. A fundamental question to ask during this prioritisation is: *how (and where) do we envisage the ERICA tools will be applied by end-users and what will they expect of them?* Interaction with end-users within the ERICA project may help in addressing this important question.

References

Amiro BD (1997) Radiological dose conversion factors for generic non-human biota used for screening potential ecological impacts. *Journal of Environmental Radioactivity* 35, 37-51.

Baker RJ, Hamilton MJ, VandenBussche RA, Wiggins L E, Sugg DW, Smith MH, Lomakin MD, Gaschak SP, Bundova EG, Rudenskaya GA and Chesser RK (1996) Small mammals from the most radioactive sites near the Chornobyl Nuclear Power Plant. *Journal of Mammalogy* 77 (1), 155-170.

Baker RJ, Dewoody JA, Wright AJ and Chesser RK (1999) On the utility of heteroplasmy in genotoxicity studies: An example from Chornobyl. *Ecotoxicology* 8 (4), 301-309.

Baker RJ and Chesser RK (2000) The Chernobyl nuclear disaster and subsequent creation of a wildlife preserve. *Environmental Toxicology and Chemistry* 19 (5), 1231-1232.

Beaugelin-Seiller K (2004) *Catalogue des données « Hydrosystèmes » acquises pour l'outil CASTEAUR*. IRSN/DEI, rapport SECRE/04-04.

Beaugelin-Seiller K and Charton J (2004) *ERICA Project – WP4 – the Loire case study*. Rapport IRSN/EDF, DEI/SECRE/04-026.

Beaugelin-Seiller K, Roussel-Debet S, Germain P, Connan O, Boust D and Fiévet B (2005) *Cobalt 60 et environnement*. Fiche radionucléide, IRSN/DIE/SECRE.

Beaugelin-Seiller K, Roussel-Debet S., Germain P, Connan O, Boust D and Fiévet B (in press). *Césium 137 et environnement*. Fiche radionucléide, IRSN/DEI/SECRE.

Beresford NA, Mayes RW, Crout NMJ, Dawson JM, Maceachern PJ, Dodd BA, Soar, JB, Barnett CL and Lamb CS (1997) *Transfer of heavy metals from feeding stuffs and environmental sources into meat, milk and other foods of animal origin. Part 1. Arsenic, Cadmium and mercury*. Final report. Ministry of Agriculture, Fisheries and Food. Centre for Ecology and Hydrology, Grange-Over-Sands, UK.

Beresford NA, Mayes RW, Cooke AI, Barnett CL, Howard BJ, Lamb CS and Naylor GPL (2000) The importance of source dependent bioavailability in determining the transfer of ingested radionuclides to ruminant derived food products. *Environmental Science and Technology* 34, 4455-4462.

Beresford NA, Wright SM, Brown J E and Sazykina T (Eds.) (2003) *Review of approaches for the estimation of radionuclide transfer to reference Arctic biota.* Deliverable 3 for the EC project 'Environmental Protection from Ionising Contaminants in the Arctic' (EPIC) (Contract No. ICA2-CT-2000-10032). EC Inco-Copernicus project ICA2-CT-2000-10032. Centre for Ecology and Hydrology: Grange-Over-Sands, UK.

Beresford NA, Wright SM, Barnett CL, Wood MD, Gaschak S, Arkhipov A, Sazykina TG and Avila R (in-press) A case study in the Chernobyl zone – part 1: predicting radionuclide transfer to wildlife. *Radioprotection (colloques).*

Betti M (2004) The MARINA II study, introduction from the perspective of EC and OSPAR policies with regard to radioactivity in the marine environment. pp. 534-537. In: *Proceedings from the international conference on radioactivity in the environment* , 1st -5th September, Monaco. Børretzen P, Jølle T and Strand P (Eds.). Norwegian radiation Protection Authority, Østerås.

BIOMOVS II (1996) *Assessment of the consequences of the radioactive contamination of aquatic media and biota: model testing using Chernobyl data*. BIOMOVS II Technical Report No.10. Swedish Radiation Protection Institute, Stockholm.

BNFL (2001) *Discharges and monitoring of the environment in the UK*. Annual Report 2000. British Nuclear Fuels Ltd., Warrington.

BNFL (2002) *Discharges and monitoring of the environment in the UK*. Annual Report 2001. British Nuclear Fuels Ltd., Warrington.

BNFL (2003) *Discharges and monitoring of the environment in the UK*. Annual Report 2002. British Nuclear Fuels Ltd., Warrington.

Brown JE, Strand P, Hosseini A and Børretzen P (Eds.) (2003a). *Handbook for assessment of the exposure of biota to ionising radiation from radionuclides in the environment.* Deliverable (including 2 appendices) for the EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Norwegian Radiation Protection Authority, Østerås.

Brown JE, Thørring H and Hosseini A (eds.) (2003b) *The "EPIC" impact assessment framework: Towards the protection of the Arctic environment from the effects of ionising radiation.* Deliverable for the EC project 'Environmental Protection from Ionising Contaminants in the Arctic' (EPIC) (Contract No. ICA2-CT-2000-10032). Norwegian Radiation Protection Authority, Østerås.

Brown JE, Jones SR, Saxén R, Thørring H and Vives i Batlle J (2004) Radiation doses to aquatic organisms from natural radionuclides. *Journal of Radiological Protection* 4, A63-A77.

Brady NC (1990) *The Nature and property of soils* (10^{th} Edition). Macmillan Publishing Company, New York.

Camplani A, Saino N and Møller AP (1999) Carotenoids, sexual signals and immune function in barn swallows from Chernobyl. *Proceedins of the Royal Society London* B 266, 1111- 1116.

Cherry RD and Shannon LV (1974) The alpha radioactivity of marine organisms. *Atomic Energy review* 12, 3-45.

Chesser RK, Sugg DW, Lomakin MD, Van den BusscheRA, DeWoody JA, Jagoe CH, Dallas CE, Whicker FW, Smith MH, Gaschak SP, Chizhevsky IV, Lyabik VV, Buntova EG, Holloman K and Baker RJ (2000) Concentrations and dose rate estimates of $^{134,137}Cs$ and ^{90}Sr in small mammals at Chornobyl, Ukraine. *Environmental Toxicology and Chemistry* 19 (2), 305-312.

Ciffroy P, Garnier JM, and Pham MK (2001) Kinetics of adsorption and desorption of radionuclides of Co, Mn, Cs, Fe, Ag and Cd in freshwater systems: experimental and modelling approaches *Journal of Environmental Radioactivity* 55, 71-91.

Ciffroy P, Garnier JM and Benyahya L (2003a) Kinetic partitioning of Co, Mn, Cs, Fe, Ag, Zn and Cd in fresh waters (Loire) mixed with brackish waters (Loire esturay): experimental and modelling approaches. *Marine Pollution Bulletin* 46, 626-641.

Ciffroy P, Reyss JL and Siclet F (2003b) Determination of the residence time of suspended particles in the turbidity maximum of the Loire river estuary by ^{7}Be analysis. *Estuarine, Coastal and Shelf Sciences* 57, 553-568.

Colle C and Mourlon C (2002) *Iode-129 et environnement*. Fiche radionucléide, IRSN/DEI/SECRE.

Copplestone D (1996) *The food chain transfer of radionuclides through semi natural habitats*. Chapter 4 - A saltmarsh ecosystem - river Esk. (PhD Thesis) University of Liverpool.

Copplestone D, Bielby S, Jones SR, Patton D, Daniel P and Gize I (2001) *Impact assessment of ionising radiation on wildlife*. R&D Publication 128. ISBN 1 85705590. Environment Agency, Bristol.

Copplestone D, Wood MD, Bielby S, Jones SR, Vives J and Beresford NA (2003) *Habitat regulations for Stage 3 assessments: radioactive substances authorisations* R&D Technical Report P3-101/SP1a. Environment Agency, Bristol.

Coughtrey PJ and Thorne MC (eds.) (1983) *Radionuclide distribution and transport in terrestrial and aquatic ecosystems. A critical review of data*. Volume 1. A.A. Balkema, Rotterdam.

Crossley DA, (1963) Movement and accumulation of radiostrontium and radiocaesium in insects. pp103-105. In: *Radioecology* Schultz V and Klement AW (Eds.), Reinhold Publishing Corporation, New York.

Dahlgaard H (1996) Polonium-210 in mussels and fish from Baltic-North sea estuary *Journal of Environmental Radioactivity* 32 (1-2), 91-96.

Ellegren H, Lindgram, G, Primmer CR and Møller AP (1997) Fitness loss and germline mutations in barn swallows breeding in Chernobyl. *Nature* 389, 593-59.

Engedahl H and Martinsen E (1996) *A hindcast archive of currents for the North Sea for 1990-1994*. DNMI Research Report no. 28. ISSN 0332-9879. Det Norske Meterologiske Institutt (DNMI), Oslo.

English Nature (2004) West Cumbria coastal plain, http://www.english-nature.org.uk/science/natural/profiles/naProfile11.pdf accessed 4/04/05.

European Commission (2003) MARINA II. *Update of the Marina project on the radiological exposure of the European Community from radioactivity in North European marine waters.* Office for official publication of the European Communities, Luxembourg.

Food Standards Agency and Scottish Environment Protection Agency (2001) *Radioactivity in food and the envirionment, 2000* RIFE-6 FSA and SEPA, London and Stirling.

Food Standards Agency and Scottish Environment Protection Agency (2002) *Radioactivity in food and the envirionment, 200* RIFE-6 FSA and SEPA, London and Stirling.

Food Standards Agency and Scottish Environment Protection Agency (2003) *Radioactivity in food and the envirionment, 2002* RIFE-6 FSA and SEPA, London and Stirling.

Fuge R and Johnson CC (1986) The geochemistry of iodine, a review. *Environmental Geochemistry and Health* 8 (2), 31-54.

Gaare E and Staaland H (1994) Pathways of fallout radiocaesium via reindeer to man. In: *Nordic Radioecology - The transfer of radionuclides through Nordic ecosystems to man*. pp. 303-334. Dahlgaard H (ed.), Elsevier, New York.

Garnier-Laplace J and Roussel-Debet S (2001) *Carbone-14 et environnement* Fiche radionucléide, IRSN/DEI/SECRE..

Gaschak S, Chizhevsky I, Arkhipov A, Beresford NA and Barnett CL (2003) In: *Conference on the protection of the environment from the effects of ionizing radiation* IAEA-CN-109, p200-203. International Atomic Energy Agency, Stockholm.

Gilhen M, Cooper K, Radbourne K and Carrington R. MSc. Theses. University of Liverpool. 2001-2003.

Gilman AP, Villeneuve DC, Secours VE, Yagminas AP, Tracy BL, Quinn JM, Valli VE, Wiles RJ and Moss MA (1998) Uranyl nitrate: 28-day and 91-day toxicity studies in the Sprague-Dawley rat. *Toxicological science* 41 (1), 117-128.

Golikov V and Brown JE (Eds.) (2003) *Internal and external dose models*. Deliverable 4 for the EC project 'Environmental Protection from Ionising Contaminants in the Arctic' (EPIC), (Contract No. ICA2-CT-2000-10032). Norwegian Radiation Protection Authority, Østerås.

Gray J, Jones SR and Smith AD (1995) Discharges to the environment from the Sellafield site, 1951-1992. *Journal of Radiological Protection* 15, 99-131.

Grøttheim S (1999) *Artificial radionuclides in the Northern European Marine Environment in 1995: Distribution of radiocaesium, plutonium, and americium in seawater and sediments*. Canadian Scientific thesis in Marine Chemistry, Department of Marine Zoology and Marine Chemistry, Biological Institute, University of Oslo.

Hamilton TF, Ballestra S, Baxter MS, Gastaud J, Osvath I, Parsi P and Provinec P (1994) Radiometric investigations of the Kara Sea sediments and preliminary radiological assessment related to the dumping of radioactive wastes in the Arctic Seas. *Journal of Environmental Radioactivity* 25, 113-134.

Hamilton LD, Meinhold AF and Nagy J (1991) *Produced water radionuclide hazard/risk assessment – phase 1*. Prepared for the American Petroleum Institute. BNL 47403. Brookhaven National Laboratory.

Henderson P (1982) *Inorganic geochemistry*. Pergamon Press, Oxford.

Holloman KA, Dallas CE, Jagoe CH, Tackett R, Kind JA and Rollor EA (2000) Interspecies differences in oxidative stress response and radiocesium concentrations in rodents inhabiting areas highly contaminated by the Chornobyl nuclear disaster. *Toxicological Chemistry* 19, 2830-2834.

Horrill AD (1984) Radionuclide levels and distribution in grazed saltmarsh in West Cumbria. *Environmental Pollution* B8, 265-280.

Howard BJ, Livens FR and Walters CB (1996) A review of radionuclides in tide-washed pastures on the Irish Sea coast in England and Wales and their transfer to food products. *Environmental Pollution* 93, 63-74.

Hursthouse AS, Baxter MS, Livens FR and Duncan HJ (1991) Transfer of Sellafield-derived Np-237 to and within the terrestrial environment. *Journal of Environmental Radioactivity* 14, 147-174.

IAEA (1985) *Sediment Kd's and concentration factors for radionuclides in the marine environment*. IAEA Technical Report Series No. 247, International Atomic Energy Agency, Vienna.

IAEA (1988) *Assessing the impact of deep sea disposal of low level radioactive waste on living marine resources*. IAEA Technical Report Series No. 288, International Atomic Energy Agency, Vienna.

IAEA (1990) *The environmental behaviour of radium*. Vol. 1. IAEA Technical Reports Series No. 310, International Atomic Energy Agency, Vienna.

IAEA (1992) *Effects of ionising radiation on plants and animals at levels implied by current radiation protection standards*. IAEA Technical Report Series No. 332, International Atomic Energy Agency, Vienna.

IAEA (1994) *Handbook of parameter values for the prediction of radionuclide transfer in temperate environments*. IAEA TECDOC Series No. 364, International Atomic Energy Agency, Vienna.

IAEA (2001) *Generic models for use in assessing the impact of discharges of radioactive substances to the environment* IAEA Safety Reports Series No 19. International Atomic Energy Agency, Vienna.

IAEA (2004) *Sediment distribution coefficients and Concentration Factors for biota in the marine environment* Technical Reports Series No. 422, International Atomic Energy Agency, Vienna.

ICRP (1973) *ICRP task group on alkaline earth metal metabolism in adult man* International Commission on Radiological Protection, Publication 20, Pergammon Press, Oxford.

ICRP (1979) *Limits of intakes of radionuclides by workers*.International Commission on Radiological Protection, Publication 30, Part 1, Pergamon Press, Oxford.

Jerez Vegueria SF, Godoy JM and Miekeley N (2002) Environmental impact studies of barium and radium discharges by produced water from the "Bacia de Campos" oil-field offshore platforms, Brazil. *Journal of Environmental Radioactivity* 62, 29-38.

Johnson C and Marshall WA (2005) ERICA *project UK case studies - assessment of the effects of ionising radiation on agricultural and semi natural ecosystems near Sellafield*. Westlakes Scientific Consulting, Moor Row, UK.

Jones DJ, Domotor SL, Higley KA, Kocher, DC and Bilyard G (2003) Principles and issues in radiological ecological risk assessment. *Journal of Environmental Radioactivity 66*, 19-39.

Jones HE (2004) *Impact of Anthropogenic Ionisng Radiation on Soil Microbial Communities*. PhD thesis. University of Nottingham.

Jones SR, Willans SM, Smith AD, Cawse PA and Baker SJ (1996) Deposition of actinides in the vicinity of Sellafield, Cumbria; accounting for discharges to atmosphere from the plant. *Science of the Total Environment* 183, 213-229.

JNCC (2004) *Drigg coast special area of conservation citations*. Joint Nature Conservation Committee, http://www.jncc.gov.uk accessed 03/03/05.

Kashparov VA, Oughton DH, Zvarich SI, Protsak VP and Levchuk SE (1999) Kinetics of fuel particle weathering and ^{90}Sr mobility in the Chernobyl 30-km exclusion zone. *Health Physics* 76, 251-259.

Kashparov VA, Protsak VP, Ahamdach N, Stammose D, Peres JM, Yoschenko VI and Zvarich SI (2000) Dissolution kinetics of particles of irradiated Chernobyl fuel: influence of pH and oxidation state on the release of radionuclides in contaminated soils of Chernobyl. *Journal of Nuclear Materials* 279, 225-233.

Kryshev II (1995) Radioactive contamination of aquatic ecosystems following the Chernobyl accident. *Journal of Environmental Radioactivity* 27, 207-219.

Kryshev II, Sazykina TG, Ryabov IN, Chumak VK and Zarubin OL (1996) Model testing using Chernobyl data: II. Assessment of the consequences of the radioactive contamination of the Chernobyl nuclear power plant cooling pond. *Health Physics* 70 (1), 13-17.

Kryshev AI (1998) *Modelling the accidental radioactive contamination and assessment of doses to biota in the Chernobyl NPP's cooling pond*. In: Proceedings of the topical meeting of International Union of Radioecologists Mol, 1st-5th June 1998, BVG, Balen, Belgium.

Kryshev II, Sazykina TG and Beresford NA (in-press) Chapter 8: Effects on wildlife. In: *Chernobyl: Environment, health and society*. Smith JT and Beresford NA (Eds.) Praxis Publishing Ltd., Chichester.

Larsson C-M, Brewitz E and Jones C (2002a) *Formulating the FASSET assessment context.* Deliverable 2: Part 1 for the EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

Larsson C-M, Brewitz E and Jones C (2002b) *Overview of programmes for the assessment of risks to the environment from ionising radiation and hazardous chemicals.* Deliverable 2: Part 2 for the EC Project 'Framework for the assessment of Environmental Impact'

(FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

Larsson C-M, Jones C, Gomez-Ros J-M and Zinger I (2004) *Framework for assessment of environmental impact of ionising radiation in major European ecosystems*. Deliverable 6 for the EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

Loire estuaire (2002) *Approche de la biodiversité des milieux naturels dans l'estuaire de la Loire. La couverture végétale.* http://www.loire-estuaire.org/pdf/plaquette_biodiv.pdf accessed 3/03/2005.

Lowe VWP (1991) Radionuclides and the birds at Ravenglass. *Environmental Pollution Series* 70, 1-26.

Matson CW, Rodgers BE, Chesser RK and Baker R J (2000) Genetic diversity of Clethrionomys glareolus populations from highly contaminated sites in the Chornobyl Region, Ukraine. *Environmental Toxicology and Chemistry* 19 (8), 2130-2135.

McDonald P, Cook GT and Baxter MS (1991) Natural and artificial radioactivity in coastal regions of the UK. pp. 329-339. In: *Radionuclides in the study of Marine Processes*. Kershaw PJ and Woodhead DS (Eds.), Elsevier Applied Science.

Macdonald CR and Laverock MJ (1998) Radiation exposure and dose to small mammals in radon-rich soils. *Archives of Environmental Contamination and Toxicology* 35, 109-120.

Ministère de l'Environnement et du Développement Durable (MEDD) (2004a) *Plan Loire Grandeur Nature*. Available from: http://www.centre.environnement.gouv.fr/plgn. htm accessed 15/05/04.

Ministère de l'Environnement et du Développement Durable (MEDD) (2004b) *Natura 2000, pays de la Loire*. http://natura2000.environnement.gouv.fr/regions/REGFR51.html, http://natura2000.environnement.gouv.fr/regions/REGFR24.html and http://natura2000.environnement.gouv.fr/regions/RGEFR72.hmtl accessed 15/05/04.

Mück K, Pröhl G, Likhtarev I, Kovgan L, Meckbach R and Golikov V (2002) A consistent radionuclide vector after the Chernobyl accident. *Health Physics* 82 (2), 141-156.

NRPA (2004) *Radioactivity in the marine environment 2002 – Results from the Norwegian Marine Monitoring Programme (RAME)*. Strålevern Rapport 2004:10. Norwegian Radiation Protection Authority, Østerås.

NRPA (2005) *Natural radioactivity in produced water from the Norwegian oil and gas industry in 2003* StrålevernRapport 2005:2. Norwegian Radiation Protection Authority, Østerås.

Nuclides 2000 (1999) *An electronic chart of the nuclides*. Institute for Transuranium Elements, Joint research center, European commission.

Pentreath R J and Woodhead DS (2001) A system for protecting the environment from ionising radiation: selecting reference fauna and flora, and the possible dose models and environmental geometries that could be applied to them. *Science of the Total Environment* 277 (1-3), 33-43.

Pentreath RJ (2002) Radiation protection of people and the environment: developing a common approach. *Journal of Radiological Protection* 22, 45-56.

Pokarzhevskii AD and Krivolutzkii DA (1997) Background concentrations of Ra-226 in terrestrial animals. *Biogeochemistry* 39, 1-13.

Pröhl G (Ed.) (2003) *Dosimetric models and data for assessing radiation exposure to biota*. Deliverable Report 3 for the EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

Reed M, French D, Rines H and Rye H (1995) *A three-dimensional oil and chemical spill model for environmental impact assessment.* In: 1995 Oil Spill Conference, 27th February - 2nd March 1995, Long Beach, California, USA.

Reed M, Rye H, Johansen Ø, Hetland B, Johnsen S, Frost T, Hjelsvold M, Salte K, Greiff-Johnsen H, Karman C, Smit M, Giacca D, Bufagni M, Cova A, Gaudebert B, Durrieu J, Røe-Utvik T, Follum OA, Gundersen J, Sanni S, Skadsheim A, Bechman R and Bausant T (2001) *DREAM: a Dose-Related Exposure Assessment Model. Technical Description of Physical-Chemical Fates Components.* In: 5th Int. Marine Environmental Modelling Seminar, 9th -11th October, 2001, New Orleans, USA.

Rivernet (2004) *Loire Nature, c'est quoi* ? Available from: http://www.rivernet.org/loire/lifeloire/wholn_f.htm Accessed 10/05/04.

Sazykina TG (1998) Long-distance radionuclide transfer in the Arctic Seas related to fish migrations. *Radiation Protection Dosimetry* 5, 219-222.

Sazykina TG, Jaworska A and Brown JE (Eds.) (2003) *Dose-effects relationships for reference (or related) Arctic biota*. Deliverable Report 5 for the EC project EPIC (Contract No. ICA2-CT-200-10032). Norwegian Radiation Protection Authority, Østerås.

Sazykina TG and Kryshev II (2003) *Assessment of radiological impact on marine biota in the OSPAR region* (MARINA update project). pp 565-568. In: Proceedings from the International conference on radioactivity in the environment, 1-5 September, Monaco. Børretzen P, Jølle T and Strand P (Eds.) Norwegian radiation Protection Authority, Østerås.

Sheard JW, Swanson SM and Goodwin B (1986) *A study of natural uranium series radionuclides in the upland vegetation of northern Saskatchewan and adjacent Northwest Territories*. Contract report for Environment Canada.

Siclet F, Luck M, Le Dortz JG, Damois C, Ciffroy P, Hendrickx F and Courivaud JR (2002) Radionuclides concentrations in the Loire river system resulting from routine discharges of five nuclear power plants: assessment of dose to man. *Radioprotection* 37 (4), 399-410.

Spencer DW, Bacon MP and Brewer PG (1980) The distribution of ^{210}Pb and ^{210}Po in the North Sea. *Thalassia Jugoslavica* 16, 125-154.

Stewart TH, Fulker MJ and Jones SR (1990) A survey of habits of people living close to the Sellafield nuclear reprocessing plant *Journal of Radiological Protection* 10, 115-122.

Strand P, Beresford NA, Avila R, Jones SR and Larsson C-M (Eds.) (2001) *Identification of candidate reference organisms from a radiation exposure pathways perspective.* Deliverable (includes 2 appendices) for EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Norwegian Radiation Protection Authority, Østerås.

Strong JC and Baker ST (1996) Lung deposition in rodents during exposure to attached radon progeny. *Environment International* 22, S905-S908.

Sugg W, Bickham J, Brooks J, Lomakin M, Jagoe C, Dallas C, Smith M, Baker R and Chesser R (1998) DNA damage and radiocaesium in channel catfish from Chernobyl. *Environmental Toxicology and Chemistry* 15, 1057-1063.

Taskaev AI, Landa ER, Guryev DV, Butler NG and Kraemer TF (2003) Vodnyi: a long-term, low-level radiation exposure field site in Russia. *Health Physics* 38, 332-343.

Titaeva NA, Taskaev AI, Ovchenkov V Ya, Aleksakhin RM and Shuktomova II (1978) Content and characteristics of U, Th and Rn uptake in plants growing under different radioecological conditions. *Ékologiya* 4, 37-44, (In Russian).

Verhovskaya IN (Ed.) (1972) *Radioecological investigations in natural biogeocenoses*. Moskow, (In Russian).

Vives i Batlle J, Jones SR and Gómez-Ros JM (2004) A method for calculation of dose per unit concentration values for aquatic biota. *Journal of Radiological Protection* 24 (4A), A13-A34.

Whicker FW and Schultz V (1982) *Radioecology: Nuclear Energy and the environment*. Vol. 1. CRC press Inc. Boca Raton, Florida.

Williams C (Ed.) (2004) Special Issue: Framework for assessement of environmental impact (FASSET) of ionising radiation in European ecosytems. *Journal of Radiological Protection* 24 (4A).

Wilson MD and Hinton TG (2002) Statistics of extreme values – comparative bias associated with various estimates of doses to the maximally exposed individual. pp.69-76. In: *Proceedings of the Third Interntional Symposium on the Protection of the Environment from Ionising Radiation (SPEIR 3).* 22nd -26th July 2002. Darwin, Australia.

Woodhead D and Zinger I (Eds.) (2003) *Radiation effects on plants and animals.* Deliverable for EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

UIAR (2001) *Contamination of the ChNPP 30-km zone.* CD v2. UIAR, Chabany.

Unesco (2000) *Inscription de la vallée de la Loire au patrimoine mondial.* Available from: http://whc.unesco.org/nwhc/pages/doc/mainf3.htm. Accessed 10/05/04.

UNSCEAR (1996) *Effects of radiation on the environment.* United Nations Scientific Committee on the Effects of Atomic Radiation. Report to the General assembly, Annex 1. United Nations, New York.

USDoE (2002) *A graded approach for evaluating radiation doses to aquatic and terrestrial biota.*Technical Standard DOE-STD-1153-2002. U.S. Department of Energy, Washington D.C., U.S.A.

van der Heide K, Duursma E, de Groot, H and Köster N (1990) Environmental aspects of phosphate fertilizer production in the Netherlands. *Science of the Total Environment* 90, 203-225.

Woodhead D and Zinger I (Eds.) (2003) *Radiation effects on plants and animals*. Deliverable 4 for EC Project 'Framework for the assessment of Environmental Impact' (FASSET) (Contract No. FIGE-CT-2000-00102). Swedish Radiation Protection Authority, Stockholm.

Young AK, McCubbin D and Camplin WC (2002) *Natural radionuclides in seafood*. Environmental Report RL 17/02. Food Standards Agency project R03010m. The Centre for Environment, Fisheries and Aquaculture Science (CEFAS), Lowestoft.