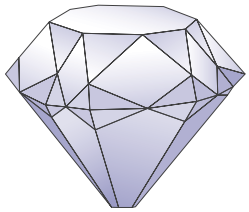
FASSET



Framework for Assessment of Environmental Impact

Deliverable 5

Handbook for Assessment of the Exposure of Biota to Ionising Radiation from Radionuclides in the Environment

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A project within the EC 5th Framework Programme

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FASSET will bring to radiation protection a framework for the assessment of environmental impacts of ionising radiation. The framework will link together current knowledge about sources, exposure, dosimetry and environmental effects/consequences for reference organisms and ecosystems. Relevant components of the framework will be identified on an ecosystem basis through systematic consideration of the available data. The application of the framework in assessment situations will be described in an overall report from the project. The project started in November 2000 and is to end by October 2003.

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German National Centre for Environment and Health	GSF
Spanish Research Centre in Energy, Environment and Technology	CIEMAT
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Executive summary

The approach taken in FASSET to perform an exposure assessment involves the derivation of (1) activity concentrations in biota and their habitat and (2) the concomitant doses received, from a starting point defined by a release into the environment. Detailed information is provided in this handbook on the application of a general methodology to address these requirements (Main Report). Look-up tables (Appendix 1) and detailed underpinning information (Appendix 2) used to support the discussion and to derive values are also provided.

Eight ecosystems (i.e. forest, semi-natural, agricultural, wetlands, freshwater, marine, brackish water and rivers) and radioisotopes of 20 elements (H, C, K, Cl, Ni, Sr, Nb, Tc, Ru, I, Cs, Po, Pb, Ra, Th, U, Pu, Am, Np, Cm) are considered in this report. Through an analysis of the behaviour and fate of these radionuclides in the ecosystems specified, reference organisms were previously selected, as described in FASSET Deliverable 1 (Strand *et al*., 2001). The generic reference organism list has been used as a basis for deriving appropriate environmental transfer data information and selecting suitable target geometries/phantoms for dosimetric modelling. The identification of actual species (or in some cases families or classes of organisms) representing each of the broadly defined groups was helpful in some instances. In the assessment process, it is thus recommended that an appropriate list of "representative" reference organisms is specified and that basic ecological information is collated for each of these flora and fauna. The specific organism attributes, that should be considered, relate directly to the subsequent assessment of exposure. For example, a description of the organism's habitat and, where applicable, the fractional occupancy within parts of this habitat, should be provided. For the purpose of illustration, Life History data sheets have been compiled and are provided in Appendix 2 of this report.

The total absorbed dose to the organism can be split into components of internal and external dose. Furthermore, it may be necessary to introduce radiation weighting factors to take account of the differing biological effectiveness of different types of ionising radiation. The basic components of information that are required to derive dose-rates to organisms, for the main exposure assessment, are the activity concentrations of radionuclides in (selected) reference biota and their habitat, Dose Conversion Coefficients (DCCs) mapping these activity concentrations onto a dose rate and occupancy factors defining the time spent by biota in various habitats for the parameterisation of external dose calculations. Guidance is provided on the application of DCCs (based on the selection of appropriate source-target configurations), occupancy factors and equations for the derivation of external absorbed doses. Similar guidance is provided for the derivation of internal absorbed doses with specific reference to the application of transfer factors when activity concentrations in reference biota are not known. The starting point for deriving transfer factors is defined by a unit concentration in reference media (unit activity concentration in water (Bq l^{-1}) for aquatic; unit rate of deposition (in units of Bq m^{-2} y^{-1}) and unit activity concentration in soil (Bq kg^{-1} dry mass)). Limitations in the application of concentration ratios have been explored. These essentially relate to problems in applying the method where sources to a compartment are numerous (e.g. plants receiving activity directly *via* interception and also *via* root uptake) and the unsuitability of applying the approach to non-equilibrium situations.

A brief consideration only has been afforded the measurement of activity concentrations in reference organisms (and their habitat), although some of the difficulties that might be encountered in relation to averaging data and furthermore defining maximally exposed individuals (as might be required in certain compliance situations) are addressed.

In order to address uncertainties in a preliminary way, some guidance is given in this report. The application of such methods may allow the identification of components in the assessment where uncertainty is greatest and facilitate the allocation of resources to areas of study (though experiment, further modelling etc.) that will reduce overall uncertainty in the most effective manner.

The derivations of transfer factors and dose conversion coefficients are discussed thematically by ecosystem type (tabulated in the look-up tables presented in Appendix 1). For each ecosystem best estimate transfer values have been derived, with the exception of agricultural ecosystems, the look-up table values for which have been based on a screening methodology. In addition, a confidence level has been attributed to each of the derived values.

For the forest ecosystem ranges of transfer factors, instead of single values, are provided. This is motivated by the high variability of species, and the very large range of variation expressed by transfer factors in forest ecosystems. The values provided are a combination of empirical data collations with values derived with a kinetic-allometric model and using existing ecological models.

The derivation of transfer factors for semi-natural pastures and heathlands has been based on empirical data collations and review and the application of the dynamic model FASTer (FASSET terrestrial model). The review included over 300 publications and included data from European Russia and the Arctic. The FASTer model, derived from established dose assessment methodologies, has been specifically developed for simulating the behaviour of radionuclides in semi-natural ecosystems and predicting transfer to reference biota. For the specific case of ^{3}H and ^{14}C, both macro-elements forming structural components of organisms and for which conventional modelling techniques are not applicable, an approach was adopted whereby activity concentrations in biota were derived from activity concentrations in air as oppose to activity concentrations in soil.

Transfer factors for agricultural ecosystems were derived through the use of a generic model based on the International Atomic Energy Agency's (IAEA) Safety Report Series No 19. The model includes four compartments representing environmental media (atmosphere, soil, water and sediment), two compartments representing concentrations in biota (crop concentration and animal concentration) and two biota final receptors receiving doses (crop total dose and animal total dose).

In the absence of a comprehensive data-set pertaining to transfer of radionuclides in wetland systems, it may be necessary to employ surrogate transfer factors derived for semi-natural and freshwater systems. The derivation of transfer factors for freshwater ecosystems has been based entirely on literature review from which approximately 700 data values were extracted. In some cases, recourse was made to data sets from regions outside of Europe. Such data were normally assigned a low confidence level owing to possible differences between European

and non-European environments. Many data gaps on concentration factors of freshwater biota are identified. Even for the most studied artificial radionuclides, data coverage did not extend to all the reference organisms considered within FASSET.

For the marine ecosystem, transfer factor derivation was based on literature review and the application of a simple biokinetic model, parameterised partly using allometric relationships. Sea mammals and sea birds are particularly poorly characterised in terms of transfer factors. Preliminary corroboration of the models employed using the few empirical data that were available suggested that model predictions were sensible. Extensive data sets exist from monitoring programmes in the Baltic Sea and these have been sued to derive transfer factors for brackish waters. For the specific case of ^{14}C, a model within which the main flows and storages of carbon, both in the physical environment and in the food web, are identified, quantified and dynamically simulated was employed.

Biological transfer factors have not been derived explicitly for river ecosystems. It can be assumed that the CFs recommended for freshwater ecosystems may be appropriately applied in most cases.

Tabulated DCCs themselves (unweighted DCCs only) have been extracted from FASSET Deliverable 3 (Pröhl *et al*., 2003), for easy access by the assessor within this handbook. These data are provided in Appendix 1 of this report.

In order to illustrate the application of the exposure assessment methodology summarized in the preceding paragraphs, three examples of application are provided in Section 5; one for marine environment and two for terrestrial ecosystems. In the first example, the modelling approach for environmental impact assessment is applied to a generic marine box. In the other two examples, assessments of the exposure of biota living in wetland and semi-natural areas are conducted.

Glossary

The following terms and definitions have been adopted or modified from; FASSET Deliverable 2, R&D Publication 128, ICRU report 65 (2001) and USDoE-STD-1153-2002.

Absorbed dose
Quantity of energy imparted by *ionizing radiation* to unit mass of matter such as tissue. Unit *gray*, symbol Gy. 1 Gy = 1 joule per kilogram.

Actinide
A group of 15 elements with atomic number from 89 (actinium) to 103 (lawrencium) inclusive. All are radioactive.

Activity
Attribute specifying an amount of a *radionuclide*. Describes the rate at which transformations occur. Unit *Becquerel*, symbol Bq. 1 Bq = 1 transformation per second.

Allometric
The allometric approach is based on the observation that many metabolic parameters, including basal metabolic rates, ingestion rates, biological half times etc., are related (as power functions) to the masses of organisms.

Alpha particle
Is a helium-4 nucleus consisting of two protons and two neutrons, given off by the decay of many heavy elements, including uranium and plutonium.

Assessment endpoint
The biological effect inferred from the measurement or predictions and which the *assessment framework* is designed to study.

Assessment framework
Identification and demarcation of the assessment boundaries. In FASSET, the framework contains the process from problem formulation through the characterization of the effects of radiation on individuals. The overall assessment system describes the tools. Methods and information flow used to carry out the impact assessment.

Benthic
Pertaining to, or with the characteristics of, the benthos; also, the bottom region of a lake or sea.

Bioaccumulation
The process whereby and organism accumulates substances in living tissues to concentrations higher than those existing in the surrounding media (e.g., soil, water and water).

Biological diversity (biodiversity)
The number and abundance of species found within a common environment. This includes the variety of genes, species, ecosystems, and the ecological processes that connect everything in a common environment.

Biological half life
The time required for a biological system (e.g. animal or animal tissue) to eliminate, by natural processes, half the amount of a substance that has been absorbed into that system.

Bioturbation
Perturbation or disturbance of sediments of soils by one or more biological mechanisms.

Chronic
Refers to an extended continuous exposure to a stressor or the effects resulting from such an exposure.

Concentration factor (CF)
In this report, the term has been applied specifically for aquatic ecosystems and is defined as the ratio of the concentration of the radionuclide in the organism or tissue (normally fresh weight) to that in water (normally filtered), assuming the system is under equilibrium.

Concentration ratio (CR)
In this report, the term has been applied specifically for terrestrial ecosystems and is defined as the activity density of *reference organism* relative to that of soil (ICRU, 2001).

Conceptual model
Is a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed.

Cytogenetic damage
Damage to chromosomes that can be detected on the microscopic level.

Decay
The process of spontaneous transformation of a radionuclide. The decrease in the activity of a radioactive substance.

Desorption
Removal of sorbed material from surfaces.

Detritivores
Organisms that feed on dead organic matter.

Distribution coefficient (k_d)
Is the ratio of the mass of solute species absorbed or precipitated on the soil or sediment to the solute concentration in the water.

Dose
Normally relates to the term *absorbed dose* as specified above.

Dose conversion coefficient (DCC)
Represents the instantaneous dose rate per unit activity concentration of the radionuclide in an organism or in the environment.

Dose rate
Dose (normally *absorbed dose*) received over a specified unit of time.

Dose-effect
The relationship between dose (or dose-rate) and the gradation of the effect in an exposed individual or population, that is a biological change measured on a graded scale of severity.

Dynamic model
A mathematical model which incorporates time as an independent variable.

Ecological risk assessment
The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors.

Ecosystem
The interacting system of a biological community and its non-living surroundings.

Endpoint

1. The final stage of a process, especially the point at which an effect is observed.
2. A radiological or other measure of protection or safety that is the calculated result of an analysis or assessment.

Environment

Water, air, land, plants and man and all other organisms living therein, and the interrelationships which exist among them.

Epipelagic

Of or relating to the part of the oceanic zone into which enough sunlight enters for photosynthesis to take place.

Equivalent dose

The quantity obtained by multiplying the *absorbed dose* by a weighting factor (*radiation weighting factor*) to allow for the different effectiveness of the various ionizing radiations in causing harm to tissue. Unit sievert, symbol Sv.

Exposure

The co-occurrence or contact between the endpoint organism and the stressor (e.g., radiation or radionuclide)

Exposure assessment

The process of measuring or estimating the spatial and temporal distribution of contaminants present in the environment or arising from future releases and deriving the concomitant levels of exposure, in this case through appropriate dose models, received by flora and fauna.

Fallout

Atmospheric deposition of particles resulting from a nuclear explosion or accidental release.

Fecundity

The number of viable offspring produced by an organism; mature seeds produced, eggs laid, or live offspring delivered, excluding fertilized embryos that have failed to develop.

Fertility

In sexually reproducing plants and animals it is the number of fertilized eggs produced in a given time.

Food chain

A linear series of species linked by specific *trophic* or feeding relationships, e.g. plant-herbivore-carnivore.

Food web

Interlocking pattern formed by a series of interconnecting *food chains*.

Gamma rays

High-energy electromagnetic photons similar to X-rays which are highly penetrating.

Half-life (Physical half-life)

The time taken for the activity of a radionuclide to lose half its value by decay. Symbol $t_{1/2}$.

Heavy metals
The term heavy metal, as widely understood[1], refers to any metallic chemical element that has a relatively high density and is toxic, highly toxic or poisonous at low concentrations. Examples of heavy metals include mercury (Hg), cadmium (Cd), arsenic (As), chromium (Cr), thallium (Tl), and lead (Pb).

High level waste (HLW)
The radioactive liquid containing most of the fission products and actinides present in spent fuel, which forms the residue from the first solvent extraction cycle in reprocessing, and some of the associated waste streams. This material following solidification; spent fuel (if it is declared a waste); or any other waste with similar radiological characteristics.

Ionisation
The process by which a neutral atom or molecule acquires or loses an electric charge.

Ionising radiation
High-energy radiation capable of producing *ionisation* in substances through which it passes.

Kinetic model
A mathematical model which incorporates rate equations and is dynamic (time-dependent).

Leaf area index (LAI)
The assimilative leaf area relative to the projected ground area for a plant community (one-side area for broad-leaved trees and curve surface are exposed to sunlight for coniferous trees).

Linear energy transfer (LET)
A measure of how, as a function of distance, energy is transferred from radiation to the exposed matter. Radiation with high LET is normally assumed to comprise of protons, neutrons and alpha particles (or other particles of similar or greater mass). Radiation with low LET is assumed to comprise of photons (including X-rays and gamma rays), electrons and positrons.

Macrophyte
A macroscopic plant.

Meristem
The undifferentiated plant tissue from which new cells are formed, as that at the tip of a stem or root.

Meroplankton
Any of various organisms that spend part of their life-cycle, usually the larval or egg stages, as *plankton*.

Monte Carlo method
Of or relating to a problem-solving technique that uses random samples and other statistical methods for finding solutions to mathematical or physical problems.

Morbidity
A loss of functional capacities generally manifested as reduced ‘fitness’, which may render organisms less competitive and more susceptible to other stressors, thus reducing the life span.

Mortality
Death; the death rate; ratio of number of deaths to a given population.

[1] The term "heavy metal" has never been defined by any authoritative body such as IUPAC. Over the 60 years or so in which it has been used in chemistry, it has been given such a wide range of meanings by different authors that it is effectively meaningless. No relationship can be found between density (specific gravity) or any of the other physicochemical concepts that have been used to define heavy metals and the toxicity or ecotoxicity attributed to heavy metals. http://www.iupac.org/publications/ci/2001/november/heavymetals.html

Mycelium
Mass of hyphae that make up the vegetative portion of fungi.

Natural radionuclide
Radionuclides that occur naturally in significant quantities on Earth.

Nekton
The collection of marine and freshwater organisms that can swim freely and are generally independent of currents, ranging in size from microscopic organisms to whales.

Nuclear fuel cycle
The stages in which the fuel for nuclear reactors is first prepared, then used, and later reprocessed for possible use again. Waste management is also considered part of the cycle.

Nuclide
A species of atom characterized by the number of neutrons and protons in its nucleus and by the energy content. Sometimes used interchangeably with the isotope of an element.

Occupancy factor
Refers to the fraction of the time that an organism expends in a specified habitat.

Photic zone
Uppermost layer of a body of water through which enough sunlight penetrates for photosynthesis to occur.

Phylogenetic
Refers to the evolution of a genetically related group of organisms as distinguished from the development of the individual organism.

Phytoplankton
Passive or weakly motile suspended plant life; the plant subgroup of *plankton*.

Plankton
The collection of small or microscopic organisms, including algae and *protozoans*, that float or drift in great numbers in fresh or salt water, especially at or near the surface, and serve as food for fish and other larger organisms.

Protozoa
Any of a large group of single-celled, usually microscopic, eukaryotic organisms, such as amoebas, ciliates, flagellates, and sporozoans.

Radiation weighting factor
Its value represent the *relative biological effectiveness* of the different radiation types, relative to X- or gamma-rays, in producing endpoints of ecological significance.

Radiological protection
The science and practice of limiting the harm to environment from radiations.

Radionuclide
An unstable nuclide that undergoes spontaneous transformation, emitting *ionising radiation*.

Rainout
Removal of aerosols from a cloud by rain: specifically where the aerosol particle acts as a condensation nucleus.

Reference medium
Soil for terrestrial ecosystems, water and sediments for aquatic ecosystems.

Reference organisms
A series of entities that provide a basis for the estimation of radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects.

Relative biological effectiveness (RBE)
For a given type of radiation, RBE is defined as:

$$\text{RBE} = \frac{\text{Dose of the reference radiation needed to produce the same effect}}{\text{Dose of the given radiation needed to produce a given biological effect}}$$

Resuspension
The physical transport of soil particles into the air by wind or other physical disturbance, or of bottom sediment particles into suspension by water currents or other physical disturbance.

Runoff
Portion of the precipitation on an area that is not held by the soil, but rather discharged from the area, e.g. through stream channels. That which is lost without entering the soil is called surface runoff and that which enters the soil before reaching the stream is called groundwater runoff or seepage flow from groundwater. In soil science, *runoff* usually refers to the water lost by surface flow, in geology and hydraulics *runoff* usually includes both surface and subsurface flow.

Semi-natural land
Extensively (as opposed to intensively) used land.

Sensitivity analysis
Analysis used to determine the relative influence of different parameters on the model output.

Soil fixation
Process or processes in a soil by which certain chemical elements essential for plant growth are converted from a soluble or exchangeable form to a much less soluble or a non-exchangeable form; for example potassium, ammonium and phosphate fixation.

Stochastic effects
A radiation-induced health effect, the probability of occurrence of which is greater for higher radiation dose and the severity of, which (if it occurs) is independent of dose.

Transfer factor (TF)
Is defined as the ratio of the activity density (Bq/kg or Bq/l) of a radionuclide in the receptor compartment to that in the donor compartment. In this report the term transfer factor is used as a generic term that includes CRs, CFs and activity concentration relative to annual deposited activity.

Trophic level
Functional classification of organisms in an ecosystem according to feeding relationships from first level autotrophs through succeeding levels of herbivores and carnivores.

Washout
Removal of aerosols from the atmosphere by falling rain.

Zooplankton
Sub-group of plankton in aquatic ecosystems and which are animals.

Table of contents

1 Introduction

1.1 Background

This handbook describes an approach to derive dose-rates for biota exposed to ionising radiation. It provides information on the selection and characteristics of reference organisms, transfer factors and dose conversion coefficients, and notes on the application of these tools. The handbook is backed up by scientific documentation in two appendices. The handbook has been produced as part of the FASSET project by Work-Package 2. The output (doses to biota) can be used in conjunction with effects analysis (FASSET Deliverable 4, Woodhead & Zinger, 2003), as parts of the overall framework that is outlined in the final report of the project, i.e. Deliverable 6.

Well established methods exist for predicting transfer of radionuclides in aquatic and terrestrial food chains (see for example, IAEA, 1994 and IAEA, 1985) but concomitant model parameters and empirical derived transfer factors invariably relate to the consideration of food chains that are relevant to human exposure. In particular, agricultural systems in terrestrial environments and marine biota forming commercially-important foodstuffs for man have received much attention. Empirical transfer data for biota extraneous to these food chains are available in the open literature (see for example, Copplestone, 1996; Fisher *et al*., 1999) but this information, from many disparate sources, has not been compiled in an easily-accessible form. Furthermore, many data are available that will allow parameterisation of models for "non-human" food chains (see for example Coughtrey & Thorne, 1983). Information is clearly available, therefore, to form the basis for a robust exposure assessment for non-human biota.

1.1.1 Objectives and Scope of the Handbook

This handbook is designed to provide guidelines and recommendations on the initial part of an environmental impact assessment. Details are provided on the selection of reference organisms, basic ecological information, environmental transfer factors and available models for predicting radionuclide concentrations in flora, fauna and their environment. The original scope of this handbook has been extended to provide guidance on the derivation of dose rate conversion coefficients for external and internal irradiation of biota (taken from FASSET Deliverable 3 - Pröhl *et al*., 2003). This approach facilitates the integration of the exposure assessment into a transparent and applicable final product.

The term "reference organism" has been defined as: "a series of entities that provides a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects." (Larsson *et al*., 2002a). Numerous criteria that might be used in the selection of reference organism types have been previously suggested (Pentreath & Woodhead, 2001) but these have not, at the present time been applied in a systematic and clearly documented manner.

There is a large range of anthropogenic and natural radionuclides which may need to be considered within environmental impact assessments, and in this initial establishment of a framework it is not possible to consider them all. Therefore, a sub-set of radionuclides of twenty elements were selected for initial consideration, based on several selection criteria including radionuclides : (i) that are routinely considered in both regulatory assessments of waste disposal and releases from different facility types, and emergency planning for accidental releases; (ii) with a range of environmental mobilities and biological uptake rates; (iii) that are both anthropogenic and natural in origin; (iv) that are representatives of α-, β- and γ-emitters and (v) for which sufficient data is likely to be available. Subsequently, a framework designed to assess these radionuclides should be sufficiently robust to be readily applicable to the consideration of others. The radionuclides considered in the handbook include radioisotopes of the following elements: H, C, K, Cl, Ni, Sr, Nb, Tc, Ru, I, Cs, Po, Pb, Ra, Th, U, Pu, Am, Np, Cm. Further details are provided in FASSET Deliverables 1 and 3 (Strand *et al*., 2001 and Pröhl *et al*., 2003, respectively).

The different ecosystems considered within FASSET have been described in some detail earlier in the project (FASSET Deliverable 1 - Strand *et al*., 2001). Seven broad groups of ecosystem have been included: Forests, semi-natural pastures and heathlands, agricultural, wetlands, marine and brackish waters. For each of these groups, typical European ecosystems have been broadly described through a consideration of typical biota species, ecological niches and habitats, food-webs and linkage/interaction with other ecosystems. Empirical transfer data, modelling work and expert judgement were subsequently employed in the process of identifying candidate reference organisms. This information will not be repeated here, the reader is referred to FASSET Deliverable 1 if further background details are required. This report will build upon this previous work retaining the same organisational distinction based on ecosystem types. Some additional information on rivers will also be given in this report.

1.2 Overview of the handbook

An overview of the handbook, including appendices is provided in Figure 1-1. In Section 2 of the main report the assessment context is defined through a discussion of the scope of the study and a generic consideration of transfer and exposure to radionuclides in the environment. Information on reference organisms and the application of life history data are also provided in this section. An overview of the assessment methodology, as it applies to all ecosystems, is presented in Section 3 before details concerning transfer factors for specific systems and the application of appropriate dose conversion coefficients are provided in Section 4. Examples of application of the methodology to a generic marine and two terrestrial ecosystems are discussed in Section 5 and concluding remarks are made in Section 6. Appendix 1 should be used in concert with the main report because this contains all tabulated values for transfer factors (covering forest, semi-natural, agricultural, freshwater, marine and brackish water environments) and dose-conversion coefficients (terrestrial and aquatic). Appendix 2 can be referred to in cases when the assessor requires more detailed information in relation to discussions in the main report and the derivation of look-up table values. Appendix 2 also contains examples of life history information for representative species.

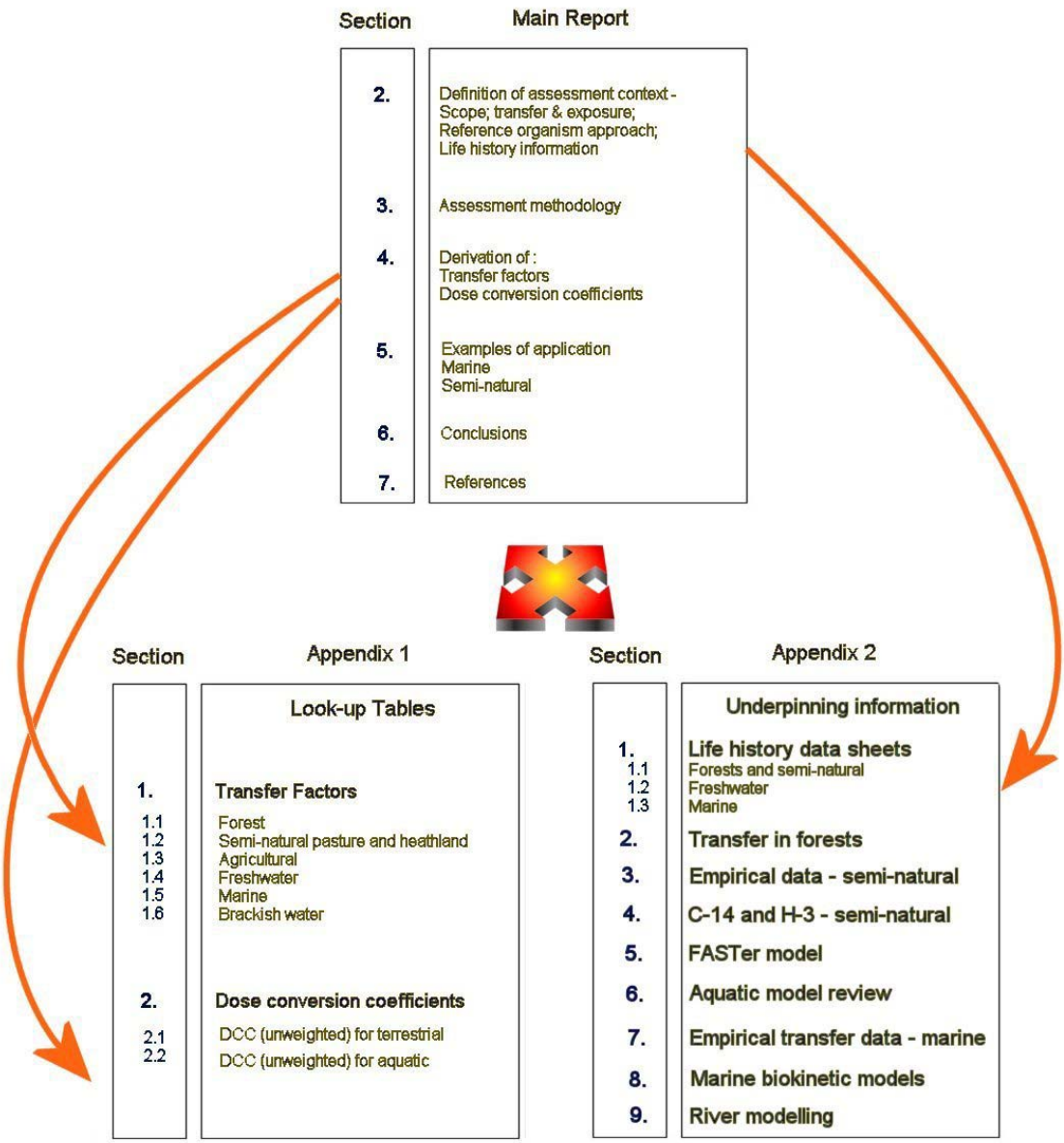

Figure 1-1 Overview of FASSET Deliverable 5.

The stages in the FASSET exposure assessment are presented in the flow diagram shown below (Figure 1-2). This figure also provides further details on where the reader can find relevant information within the handbook and appendices.

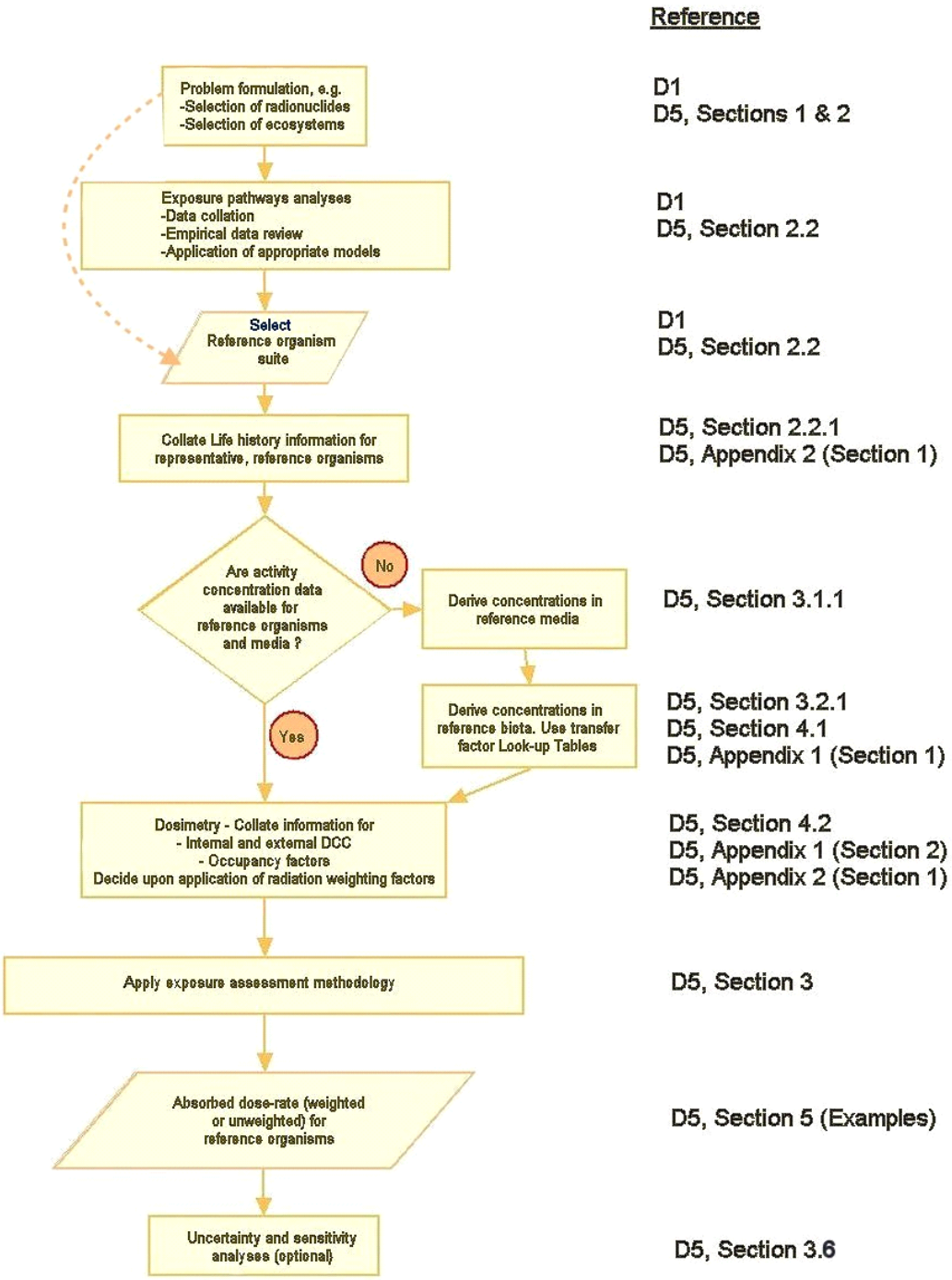

Figure 1-2 Flow diagram showing stages in the FASSET exposure assessment

2 Definition of the assessment context

The purpose of the assessment described in this report is to allow (whole body) dose rates (μGy h^{-1}) to be derived for individual organisms of selected species/types of biota, termed

reference organisms, which have been chosen because effects on individuals of these species/types of biota are thought to be important with respect to effects on the environment as a whole. Although the protection of populations of wild organisms may be the relevant endpoint in many impact assessments, the individual is selected within FASSET as the assessment endpoint because:

(1) many common species, not only those considered threatened or endangered, are protected by national laws at the individual level (Pentreath, 1999);

(2) the most basic, testable, piece of information for which a dose/effect probability factor is derived is that which applies to an individual organism. (Pentreath & Woodhead, 2001);

(3) in relation to the practicalities of performing a dose calculation, the selection of individuals has a distinct advantage, e.g. models for exposure assessment are often derived for individuals (see Sample *et al*., 1997); and

(4) populations are only affected by effects on individuals. Protecting individuals will therefore protect the population (FASSET Deliverable 4 - Woodhead & Zinger; 2003).

The biological endpoints of concern within FASSET (see Woodhead & Zinger, 2003) have been defined under 4 umbrella categories namely:

(1) mortality/lethality;
(2) reproductive success (fertility and fecundity);
(3) morbidity; and
(4) scoreable cytogenetic damage.

It is recognised that target organs within reference flora and fauna, for which dose-rates could be calculated, might be best selected to relate closely to these biological endpoints. In particular, reproductive organs might be selected as a target because exposure at this point can be critical in relation to the consideration of reproductive success. However, in most cases, such ambitions have not been achieved because:

(1) With respect to transfer in the environment, few empirical environmental transfer data exist for organs within biota. Where data have been found, and where appropriate, organ-specific transfer information has been occasionally reported. The use of food chain transfer models also lends itself more to the derivation of whole-body concentrations than to specific organ concentrations, e.g. elimination rates derived from allometric relationships are normally derived for whole body.

(2) The preponderance of dose-effects data relate to whole-body exposure from external radiation sources as oppose to experiments where dose-rates from alpha emitters have been calculated and related to a specific biological endpoint. Therefore, even where dose-rate data for particular organs have been derived, interpretation, in terms of expected effects, might not be possible.

The aim of FASSET has been to develop a framework containing assessment tools that can be tailored to a particular purpose, e.g. demonstration of compliance, remediation considerations in the event of an accident etc. Components of the system, most notably the “transfer-exposure pathways” have been developed with primary regard to conditions in European ecosystems.

Finally, the quantity of measurement, the absorbed dose (or dose-rate) in Gy (or Gy per unit time) requires some consideration with respect to the FASSET assessment. When using the dosimetry system in practice, the fact that radiations can differ in their *qualitative effect*, i.e. the same absorbed dose from different types of radiation can produce various degree of effect in the same biological endpoint, must be taken into account. For example, there is a very substantial body of experimental evidence to indicate that the absorbed dose of high linear energy transfer (LET) radiation (α-particles) required to produce a given biological effect is less than that of low LET radiation (β-particles and γ-rays) - the relative biological effectiveness (RBE) phenomenon. For human radiological protection practice, this phenomenon is taken into account by applying dimensionless radiation weighting factors (w_r) to the absorbed doses from the different radiations, and the sum give a quantity called the *equivalent dose.* It should be emphasized, however, that values of w_r defined for the purpose of human radiation protection cannot be applied without reservation to other organisms and biological endpoints. More explicitly, the radiation weighting factor of 20 for α-particles used in protection of humans (ICRP, 1991; NCRP, 1993) may not be appropriate, because this value was intended to represent RBEs for stochastic effects, primarily the induction of cancers (ICRP, 1991).

The fact that the choice of RBE is a contentious issue has been highlighted most recently by Tracy & Thomas (2002). These authors stressed the point that the choice of radiation weighting factor cannot be tied to a unique value of RBE since this quantity varies with species, end-point and dose range. Although examples exist in the literature where alpha RBEs in excess of 350 have been calculated, these derivations have often been associated with a number of problems including poor statistics, high uncertainties or questionable dosimetry. In addition to considerations of alpha RBE, there is evidence to suggest that β-radiations with energies below 10 keV have a higher RBE than electrons with energies above 10 keV (Moiseenko *et al*., 2000; Straume & Carsten, 1993). In view of the various ongoing discussions relating to the theme of RBE, a decision was made to express radionuclide specific Dose Conversion coefficients (DCCs) as 3 absorbed dose-rate components corresponding to low β (< 10 keV), high β (> 10 keV) & γ, and α radiations. No concrete recommendation has been made by FASSET in relation to radiation weighting factors. Instead, it has been suggested that biota specific w_r of 3 for low β and between 5 and 50 for α-radiation may be appropriate for environmental impact assessments (FASSET Deliverable 3 - Pröhl *et al*., 2003).

2.1 Generic consideration of radionuclide transfer and organism exposure

Although many transport processes are common to a large number of radionuclides, the quantitative importance of such processes is often dictated by the unique properties of a particular radionuclide in question. In the following section, some of the processes influencing the environmental behaviour and fate of radionuclides will be considered, in a general way, in order to place the exposure assessment methodology (Section 3) into a broader context. Some of these processes are conceptualised in Figure 2-1.

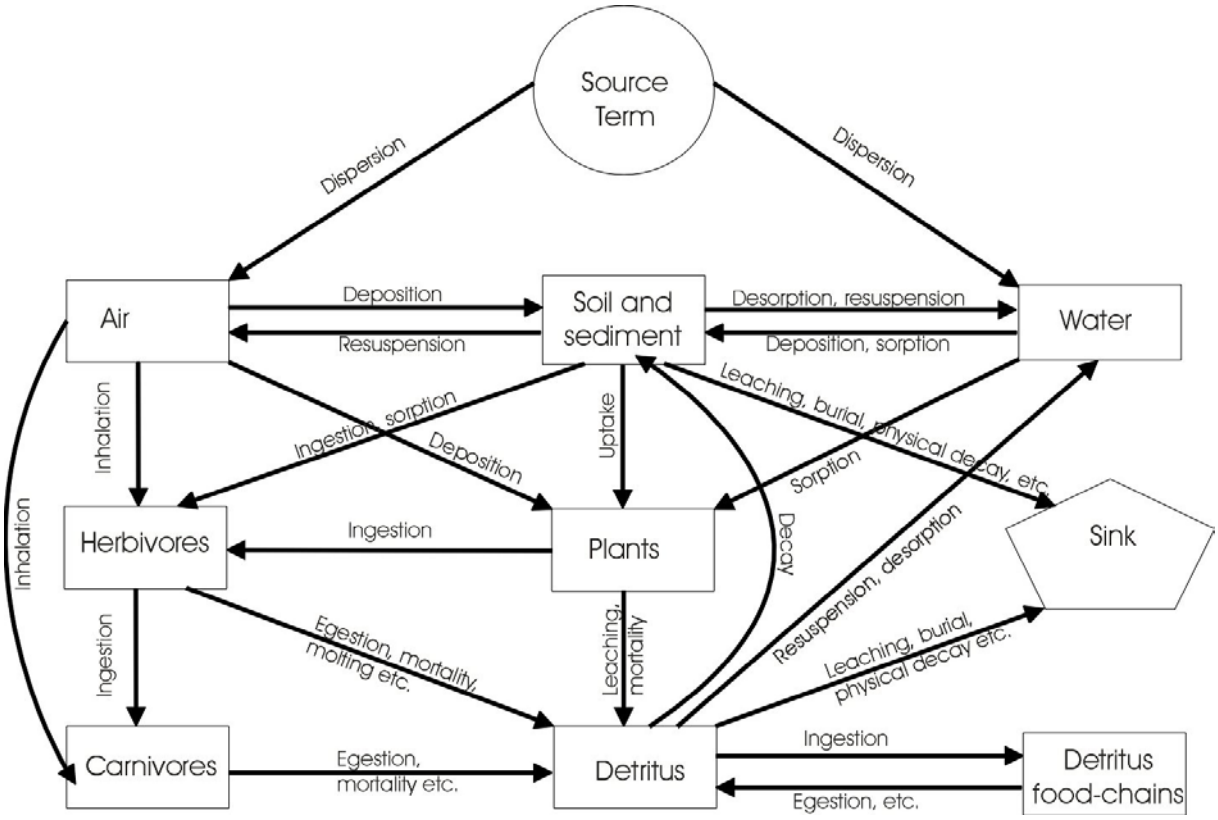

Figure 2-1 Processes affecting radionuclide behaviour in ecosystems (Based on Whicker & Shultz, 1982).

2.1.1 Physical and chemical processes

Once released into air or water, radionuclides will be influenced by physical processes that lead to their advection and dispersion in the environment. The physical and chemical form of the radionuclide and the turbulence of the receiving medium play an influential role in relation to these initial transfer mechanisms. Other process will continually cause the transfer of contamination from free air or the water column to the ground or sediment surface. These include:

(1) Gravitational settling of suspended particulate material in atmospheric or aquatic releases. The physical size of the contaminant is clearly an important attribute with respect to this.

(2) Precipitation scavenging, whereby aerosols are washed from the atmosphere by water droplets or ice crystals.

(3) Impaction, whereby suspended particles impinge on solid object within an air/water stream.

(4) Chemical sorption and exchange, dependent on both the chemical and physical form of the radionuclide and the interacting surface.

Radionuclides interact with solid material, such as soil and sediment particles, plankton vegetation etc. by numerous mechanisms including electrostatic attraction and formation of chemical bonds. In many cases, the grain-size dictates the radionuclide activity per unit mass of solid (e.g. Hetherington & Jefferies, 1974; Bonnett *et al*., 1988; Livens & Baxter, 1988) purely because the surface area, available for adsorption, per unit mass or volume is greater for fine grained solids. In most cases solid materials, i.e. soil or sediments, accumulate higher concentrations of radionuclides than air or water with some notable exceptions, e.g. noble gases following atmospheric release.

In the terrestrial environment, interception of radionuclides by vegetation occurs by wet, dry and occult deposition. The remaining fraction of radionuclides introduced to a terrestrial ecosystem may impact the ground directly. Biomass per unit area clearly affects the interception fraction for all deposition categories but other factors, including ionic form, precipitation intensity, crop maturity and leaf area index are especially important when considering wet deposition. Radionuclide concentrations on vegetation may be reduced by a number of physical processes including wash off by rain or irrigation, surface abrasion and leaf bending from wind action, resuspension, tissue senescence, leaf fall, herbivore grazing, addition of new tissue, volatilisation and evaporation. Empirical formulae have been derived to model retention of radionuclides on vegetation, i.e. crop, surfaces (IAEA, 1994).

Resuspension is an important process in both aquatic and terrestrial systems. In aquatic systems, turbulent action of water can remove surface sediments and transport them considerable distances before they are lost from the water column by sedimentation processes. Such processes, for example, appear to be important in the Irish Sea for redistributing historically labelled sediments from open coastal sites to peripheral marine areas where long term sediment accumulation is occurring (Brown *et al*., 1999). Furthermore, contaminated suspended sediments will be available for entry into marine food chains. Filter-feeding organisms, such as blue mussels, are known to ingest large amounts of seston/particulate material (Hawkins *et al*., 1998) and may therefore be exposed to relatively high levels of particle-reactive contaminants. In terrestrial systems, wind action and rain “splash” on the soil “reservoir” reintroduce radionuclides to the air where they can be (re)deposited onto sediment or inhaled by animals. This process can be influenced by factors including the height and type of the plant canopy as well as wind, rain and soil type.

Chemical and physical processes occurring in soil and sediment lead to the further redistribution of radionuclides within these compartments. In soils, radionuclides can migrate downwards by leaching. Rates of leaching appear to be greater under conditions of high rainfall or for soils containing a relatively large proportion of sand particles (Copplestone *et al*., 2001). For freshwater lacustrine sediments, upward and downward diffusional fluxes of radionuclides can result in the redistribution of these contaminants within sediments (this has been observed with radiocaesium - Comans *et al*., 1989). The process of physical disturbance and bioturbation can lead to the mixing of radionuclides in the surface layer of the sediment over short time periods. In the north east Irish Sea for example, mixing of surface sediments (< 13 cm depth) occurs on a time-scale of ca. 1 year (Mackenzie *et al*., 1998). The sedimentation of particulate material will also lead to the burial of contamination. In the terrestrial environment animals relocate material both horizontally and vertically during the

construction of burrows, tunnels and chambers. Earthworms can also be important in terms of redistributing quantities of contaminated soil.

The geochemical phase association of radionuclides in sediments and soils can change with time (see Vidal *et al*., 1993). This affects physical transport within the system and transfer to foodwebs in numerous complex ways. In some cases, a substantial proportion of the radionuclide may become associated with residual phases, and in this way become effectively removed from uptake by organisms. Such behaviour is exemplified by radiocaesium a fraction of which can be fixed by illitic soils (Hird *et al*., 1996). The fixing process leads to a virtually irreversible binding of the radionuclide to the soil matrix. In other cases changes in solid phase chemistry may lead to redistribution between geochemical phases. For example, it has been inferred that Pu is released from organic phases, as organic matter degrades, and is recaptured by sesquioxide phases in salt marsh environments on the UK coast (Brown *et al*., 1997). Transfer within the sediment compartment can, therefore, be influenced by factors including bacterial activity and redox conditions. Fractions of many radionuclides persist in exchangeable phases and in aquatic environments may be prone to re-dissolution processes whereby the contaminant is transferred from the sediment compartment to the water column (see e.g. Hunt & Kershaw, 1990). The fraction of a particular radionuclide present in exchangeable phases will depend on numerous factors including, amongst others, the sediment characteristics, the presence of competing ions, pH and redox conditions.

2.1.2 Biological interaction

Radionuclides can enter the lowest trophic level, characterised by primary producers such as terrestrial and aquatic flora, by numerous processes. In terrestrial systems, these processes include direct adsorption to plant surfaces followed by foliar uptake (e.g. Zehnder *et al*., 1996) and more importantly, for the majority of radionuclides, *via* the passage of atoms (normally ions) or molecules in soil solution through root membranes into the internal organs of the plant. The transfer of many radionuclides from soil to plant is strongly influenced by soil characteristics.

For marine systems, generally, phototrophs[2] and phytoplankton are the most important primary producers and form the base of the foodchain. For coastal environments, however, macrophytes and macroalgae can account for more than 30 % of the primary production. It is notable that the adsorption of radionuclides to phytoplankton can be substantial as exemplified in the relatively high CFs derived for actinides (Fisher *et al*., 1983). This might result in a substantial input of particle-reactive radionuclides to food-webs, especially those for which filter feeders, feeding on microscopic plants and animals, form an important component.

In terrestrial ecosystems, the transfer of radionuclides from plant and soil compartments to herbivores occurs mainly by ingestion. When plants are consumed they are likely to include a component of contamination associated with soil adhered to the plant surface as well as

[2] most primary production in marine waters is believed to be accomplished by single-celled 0.5 to 10 μm phototrophs (bacteria and protists).

contamination incorporated within the plant itself. This may be important because radionuclides that are organically-bound or present in ionic form within the body of the plant may be assimilated, by the herbivore, to a greater degree than those radionuclides adsorbed to soil matrices (Whicker & Shultz, 1982). Nonetheless, for radionuclides that are not easily taken up by plants, the effects of soil adhesion can be as important in terms of the total contamination level associated with the plant (IAEA, 1994). In some instances, soil ingestion by animals may be deliberate, but quantities can also be ingested by foraging on low-growing plants, and licking or preening of fur, feathers or offspring (Whicker & Shultz, 1982). For aquatic organisms the transfer of contaminants from basal trophic levels in epipelagic systems may be best depicted by the ingestion of phototrophs and phytoplankton by protozoa and zooplankton. These organisms in turn provide food for successively higher trophic levels filled by free-swimming organisms, the so-called nekton (see FASSET Deliverable 1 - Strand *et al*., 2001).

The process of predation, whereby herbivorous organisms are consumed by carnivorous or omnivorous organisms, leads to the transfer of radionuclides to successively higher trophic levels. For radionuclides with nutrient analogues, uptake through the gastrointestinal tract of higher animals may be significant. Depending on, amongst other factors, the physico-chemical form of the radionuclide, the contaminant may be channelled to particular organs or body structures. As an example, radiostrontium behaves as an analogue for calcium. It follows the same metabolic pathways, with the result that ^{90}Sr is often incorporated, to a significant degree, into the minerals of skeleton (Coughtrey & Thorne, 1983; Odum, 1957). For other radionuclides, absorption may be minimal resulting in the passage of a very large fraction of the contaminant through the digestive tract. An example is Pu for which absorption is often extremely low for adult animals, although combining Pu with organic chemicals can markedly increase uptake (Coughtrey *et al*., 1984). In the particular case of aquatic environments, it should be noted that substantial proportions of some radionuclides may be transferred to predatory, truly aquatic animals directly from the water column. As an example, the effect of food chain transfer for Pu is insignificant and the observed body burden in organisms from the upper level of the (truly aquatic) food chain (e.g. predatory fish) appears to be almost entirely due to direct adsorption from the water column (Thomann, 1981). In contrast, the transfer of Pu to high trophic level aquatic birds and mammals cannot occur *via* this pathway and food chain transfer constitutes a dominant process in determining body contaminant burdens.

The death of plants and animals, removal of body parts, secretions and excretions will provide inputs of radionuclides to the detritus reservoir in terrestrial and aquatic ecosystems. Detritus can serve as an important reservoir for radionuclides which can cycle within the compartment through linkage to detritus food chains (Whicker & Shultz, 1982). With time, insoluble organic material, containing contamination, is broken down to simpler forms by the action of detritivores and, more importantly, microbes. This leads to the release of radionuclides, to the water column, pore water or air etc., in soluble forms (or associated with very fine detrital material) which may become available, once more, for uptake by primary producers and other biota. Recycling of radionuclides will thus occur. In contrast, deeper soil and sediment layers may act as permanent sinks for contaminants. Some of the processes discussed above including sedimentation in the aquatic environment, leaching and vertical relocation of solid

material in aquatic and terrestrial systems, may lead to removal of contaminants to compartments for which access by organisms is limited and biological uptake is unlikely.

The kinetics of the overall system, defined by rates of transfer between compartments, will determine the temporally-varying and steady-state (if attained) distribution of radionuclides within any given ecosystem. Rates of intercompartmental transport, however, vary with the radionuclides, the nature and activities of the biota and the properties of the ecosystem.

These processes have been considered, albeit in a generic way, during the early stages of the FASSET project. The behaviour and fate of selected radionuclides within various European ecosystems were described in the FASSET project and have been summarised in FASSET Deliverable 1 (Strand *et al*., 2001).

2.1.3 Exposure of biota

Exposure of biota to radiation and transfer of radionuclides in the environment, as discussed above, are, of course, intimately linked. Exposure of biota to ionising radiation occurs when radionuclides, present naturally in the environment or released through man's activities, decay releasing radiation of various types and energies. The pathways leading to exposure in aquatic and terrestrial ecosystems can be split into several categories:

(1) Inhalation of (re)suspended contaminated particles or gaseous radionuclides. This pathway is relevant for terrestrial animals and marine birds and mammals.

(2) Contamination of fur, feathers and skin. This has both an external exposure component, e.g. $\beta-$ and $\gamma-$emitting radionuclides on or near the epidermis cause irradiation of living cells beneath and an internal exposure component as contaminants are ingested and incorporated into the body of the animal.

(3) Ingestion of lower trophic plants and animals. This leads to direct irradiation of the digestive tract and internal exposure if the radionuclide becomes assimilated and distributed within the animal's body.

(4) Direct uptake from the water column, in the case of truly aquatic organisms (e.g. fish, molluscs, crustaceans), leading to direct irradiation of respiratory system, e.g. gills, and internal exposure if the radionuclide becomes assimilated and distributed within the animal's body.

(5) Intake of water contaminated by radionuclides through the gastrointestinal tract, i.e. the organism drinks water. The same exposure categories as discussed in (3) are relevant here.

(6) External exposure. This essentially occurs from exposure to γ-irradiation and to a much lesser extent β-irradiation, originating from radionuclides present in the organism's habitat. For microscopic organisms, irradiation from α-particles is also possible. The configuration of the source relative to the target clearly depends on the organism's ecological characteristics and habitat. A benthic dwelling fish will, for example, be exposed to radiation from radionuclides present in the water column and deposited sediments, whereas a pelagic fish may only be exposed to the former.

In the context of FASSET, inhalation and contamination of fur, feathers and skin (exposure pathways (1) and (2) in the above list) have not been considered explicitly in the derivation of transfer parameters or dose-conversion coefficients. The ingestion and direct uptake from water pathways (points (3) and (4) in the above list) have been considered in so far as they relate to internal body burdens of contaminants normally under equilibrium conditions. Irradiation by unassimilated contaminants in the gastrointestinal tract has not been considered nor has exposure occurring due to the consumption of water (point (5) above). Finally, external exposures have been considered in some detail both in terms of contaminant transfer to terrestrial and aquatic habitats and from the dosimetric perspective, the latter having been described in FASSET Deliverable 3 (Pröhl *et al*., 2003). An example of how exposure is conceptualised for the aquatic environment is shown in Figure 2-2.

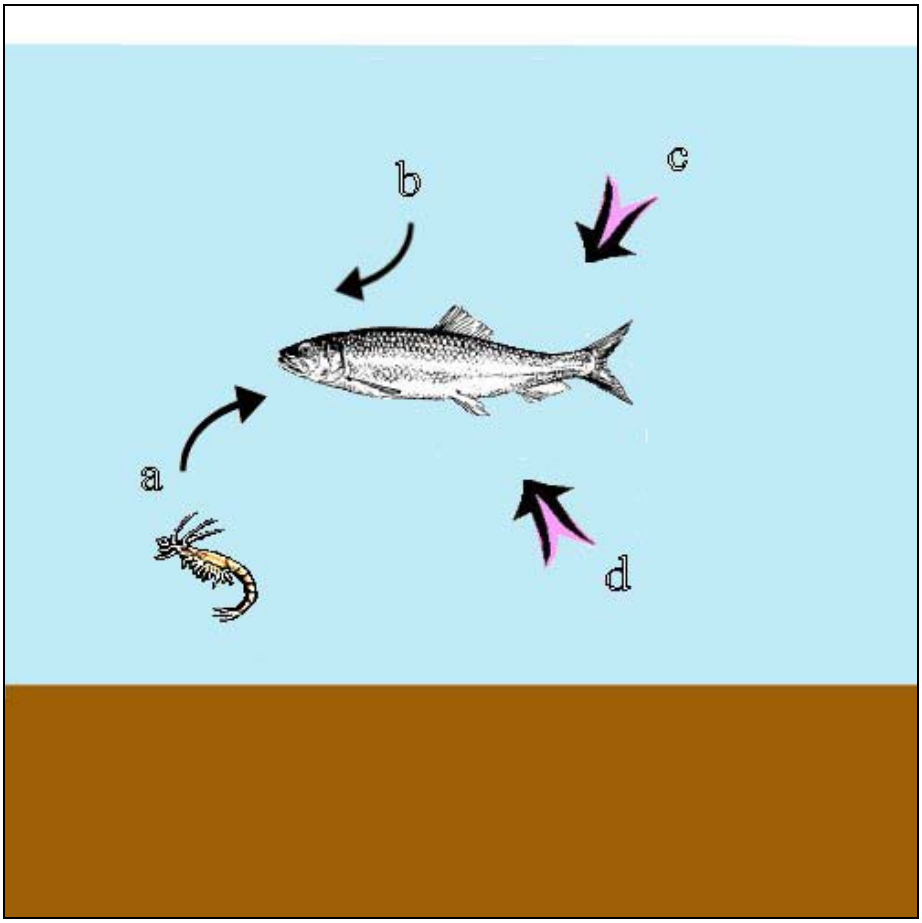

Figure 2-2 Exposure pathways for aquatic organisms as considered by FASSET. (a) Internal exposure via ingestion of contaminated food and assimilation; (b) internal exposure via direct uptake from the water column; (c) external exposure directly from radionuclides in the water column; (d) external exposure from radionuclides in sediments.

2.2 Guidelines for selecting the reference organisms

Within FASSET the number of selection criteria was originally specified to include:

(1) ecological sensitivity, i.e., the potential of the organism, through feeding habits and habitat occupancy, to be exposed to significant dose rates from radionuclides in their environment that derive from a variety of release scenarios,

(2) intrinsic sensitivity of the organism to chronic low-level irradiation for the biological endpoints of significance at the relevant level of biological organisation, and

(3) ecological significance, i.e., the organism's importance to the maintenance of the community or ecosystem. The potential requirement for generic representatives of each trophic level in the marine, freshwater and terrestrial environments will need to be considered.

This selection of reference organisms for FASSET, as described in FASSET Deliverable 1 (Strand *et al*., 2001), was based upon an assessment of ecological sensitivity and, to a more limited extent, an assessment of ecological significance. The lists of candidate reference organisms derived from the study are presented in Tables 2-1 and 2-2.

Table 2-1 Candidate reference organisms for aquatic ecosystems

Bacteria (M,B,F)	Bivalve Mollusc (M,B,F)	Pelagic Fish (M,B,F)
Phytoplankton (M,B,F)	Insect larvae (B,F)	Amphibian (F)
Macroalgae(M,B)	Zooplankton (M,B,F)	Wading bird (M,B,F)
Vascular plant (M,B,F)	Crustacean (M,B,F)	Mammal (M,B,F)
Worm (M,B)	Benthic fish (M,B,F)	

M = Marine; B= Brackish; F = Freshwater

Table 2-2 Candidate reference organisms for terrestrial ecosystems

Microorganism (Fo,S,W)	Shrub (S,A)	Herbivorous mammal (Fo,S,A,W)
Fungi (Fo,S)	Tree (Fo,A)	Burrowing mammal (Fo,S)
Lichen/bryophyte (Fo,S,W)	Worm (Fo,S,W)	Carnivorous mammal (Fo,S,W)
Grass/herb/crop (Fo,S,A,W)	Canopy invertebrate (Fo)	Bird egg (Fo,S)
Plant (Fo,S,A,W)	Detritivorous insect (Fo,S)	

Fo = Forest; S = Semi-natural; A = Agricultural; W = Wetlands

According to the original plans made in the project this list would be refined and reduced, as the qualifying word "candidate" implies, by the application of additional selection criteria, including, for example, intrinsic radiosensitivity of the reference organisms. However, it has not been possible to assess the relative radiosensitivity of the different groups of organisms. Therefore, the entire list has been adopted for consideration within FASSET.

The FASSET approach has not been designed to overly prescriptive. For a given situation the assessor may wish to conduct his/her own exposure pathways assessment, adopting FASSET

methodologies where applicable, in order to refine the reference organism lists presented in Tables 2-1 and 2-2. If detailed information on a particular release of radionuclides to the environment is available, more precise conclusions can be drawn with regards to the most exposed biota types and whether other criteria, for example relative radiosensitivity between groups, modifies the selected organism suite.

2.2.1 Life history information

The generic reference organism list has been used as a basis for:

(1) deriving appropriate environmental transfer data information, and

(2) selecting suitable target geometries/phantoms for dosimetric modelling.

With respect to these points, it became apparent that the identification of actual species (or in some cases families or classes of organisms) representing each of the broadly defined groups would be helpful in some instances. This was true in the case of deriving food chain model parameters where detailed information was often required, beyond a generic consideration, with respect to organism characteristics. It was also true in the case of geometry construction where quantitative information on size, shape and density are required and can be derived, simply and transparently, from a consideration of real flora and fauna. For dosimetric calculations, the dimensions and shape were derived from the adult form of the biota in most cases (see FASSET Deliverable 3 - Pröhl *et al*., 2003).

In the assessment process it is thus recommended that an appropriate list of "representative" reference organisms is specified and that basic ecological information is collated for each of these flora and fauna. The specific organism attributes that should be considered relate directly to the subsequent assessment of exposure. For example, information should be provided on habitat and, where applicable, the fractional occupancy of various organisms in their habitats. This information is important for the weighting of external dose-rates in order to account for the behaviour of the organism (see Section 3.1).

Guidance on the types of ecological information required for reference fauna is provided in Table 2-3.

Examples of suitable (representative) reference organisms have been selected in FASSET through a consideration of their ubiquity, geographical spread in Europe and available data. This information is mainly presented in Appendix 2, Section 1. An overview of the representative organisms selected and where life history information on these organisms can be found in the report is presented in Table 2-4.

Table 2-3 Ecological information required for reference fauna

Information	Comment
(i) Latin and common English name of the selected species.	Simple assessment[1]
(ii) Biota dimensions (mass, dimensions)	Simple assessment[1] Dimension – represent as ellipsoid and defined length, width depth Required for geometry configuration
(iii) Habitat – configuration and occupancy factors	Simple assessment[1] Required for target source configuration – external dose assessment e.g. marine – pelagic, benthic; terrestrial – at soil surface, in soil (depth and orientation), Occupancy factors – fraction of time spent in different habitats – required for average dose-rate calculation
(iv) Habitat (dynamic)	e.g. does animal hibernate (if so where + time) ? Parts of life-cycle in different habitats – meroplanktonic larvae? Advanced assessment – information required in the calculation of integrated doses
(v) Distribution – Home range.	Advanced assessment – information required in the calculation of integrated doses
(vi) Average life expectancy,	Advanced assessment – information required in the calculation of integrated doses
(vii) Feeding habits	e.g. main prey species, Advanced assessment – information required for input to ecological models
(viii) Additional information on lifecycle	e.g. viviparous fish, periods spent in freshwater Advanced assessment – information required in the calculation of integrated doses; sensitive periods in life-cycle

[1]Simple assessment – basic information required for the calculation of dose-rates. An advanced assessment is possibly beyond the scope of initial FASSET aspirations. However, such information may prove useful in the parameterisation of food chain and exposure models.

It should be noted that some of the information specified in Table 2-4 and presented in Appendix 2, Section 1 for selected biota, is redundant for the purpose of conducting the impact assessment described in the next Section of this report. Essentially, only information on the dimensions and habitat of a particular organism are required to allow informed application of appropriate DCCs and occupancy factors. Organism masses have been used in some cases to provide appropriate values for allometric relationships, which have subsequently been implemented within the dynamic radioecological models described in Appendix 2. The additional information, e.g. home range, special life-cycle data etc. may be useful in the application of a more detailed ecological risk assessment (e.g. Sample *et al.*, 1997) or in the parameterisation of models simulating how populations might respond to radiation induced changes in individual attributes (see for example Woodhead, 2003).

In many cases, it may be possible to adopt the FASSET life history information sheets directly for the purposes of an assessment.

Table 2-4 Table showing list of representative organisms and where they can be found in the report

Ecosystem	Representative species	Reference
Forest and semi-natural	Creeping bent, Heather, Reindeer lichen, Cep, Scots pine, Common oak, Earthworm, Woodlouse, Wood ant, Red grouse (egg), Mole, Rabbit, Weasel, Red fox, Moose.	Appendix 2; Section 1.1
Agricultural	Potato, Carrot, Onion, Lettuce, Tomato, Wheat, Grapevine, Orange, Apple, Olive, Cow, Sheep, Pig.	Main Report; Section 4.1.4
Wetlands	Select from freshwater and semi-natural species as appropriate	Appendix 2; Section 1.1 + 1.2
Freshwater	Water millfoil, Freshwater clam, Gastropoda, Freshwater isopod, Burbot, Perch, Common frog, Muskrat, Common gull.	Appendix 2; Section 1.2
Marine	Phytoplankton, Bladder wrack, Northern shrimp, Blow lug, Blue mussel, European lobster, Plaice, Mackerel, Eider duck[3], Harp Seal	Appendix 2; Section 1.3
Brackish	Select from freshwater and marine as appropriate	Appendix 2; Section 1.2 + 1.3
Rivers	Select from freshwater as appropriate	Appendix 2; Section 1.2

2.3 Starting point for the exposure assessment – scenarios covered

The starting point for the exposure assessment, important in terms of the defining the types of information (on transfer factors etc.) to be included in this handbook, has been chosen to be simple and generically applicable. For the aquatic system a unit activity concentration in water (Bq l^{-1}) has been used as a reference point for subsequent derivation of activity concentrations in sediment (Bq kg^{-1} dry mass) and organisms (Bq kg^{-1} fresh mass). For terrestrial systems, the approach is slightly different because of considerations relating to the widely dissimilar nature of foreseeable input terms. In order to simulate the behaviour of radionuclides following both an aerial input of contaminant to the ecosystem as might be observed following a nuclear accident or routine atmospheric stack discharges and an underground input following discharge from, for example, a high level waste repository, two starting points were selected:

(1) unit rate of deposition (in units of Bq m^{-2} y^{-1}) and

(2) unit activity concentration in soil (Bq kg^{-1} dry mass)

[3] Eider Duck (*Somateria mollissima*). This bird is not a wader but the choice of a duck as a representative biota was considered appropriate for numerous reasons, not least the fact that this would be in line with approaches that have been taken elsewhere (e.g. Copplestone *et al*., 2001).

Physical transport models simulating the initial transport of contaminants from the point of release to the point of entry into the selected ecosystems are not incorporated explicitly within the FASSET exposure assessment. For the aquatic environment, information is, however provided on the types of contaminant transport models commonly applied by the European scientific community for the purpose of predicting radionuclide activity concentrations in abiotic environmental media in space and time. A review on atmospheric dispersion models or hydrogeological transport models for the purpose of simulating inputs to terrestrial systems has not been conducted. For both aquatic and terrestrial systems, it is assumed that any assessor seriously wishing to conduct a prospective environmental impact assessment, has access to modelling tools that will allow the generation of appropriate input data to the FASSET assessment.

FASSET does, however, provide tools for the assessment of radionuclide transfers within selected ecosystems from the starting points specified above. Transfer factors have been considered using a number of approaches including reviews of published literature or archived monitoring data and the application of appropriate (food chain) transfer models. For some ecosystems, e.g. freshwater, empirical transfer data have formed the focus of the exercise. For other ecosystems, i.e. agricultural, the derivation of transfer factors has been based primarily on the application of established models. Finally, in cases such as brackish water, semi-natural terrestrial and marine ecosystems, a combination of empirical data review and model simulations has been employed.

In the subsequent derivation of transfer data for inclusion in look-up tables (Appendix 1), consideration was given to transfer factors at equilibrium, unless otherwise specified. Transfer factors derived from experimental and field studies are often expressed as the ratios of the radionuclide concentration in the organism, e.g. plant, animal, to the activity concentration in the surrounding medium, e.g. soil, water. For most situations, equilibrium is tacitly assumed or explicitly stated. In some cases, such as those discussed in more detail in Section 3.2.2, these conditions may not be satisfied and steady state conditions may not be attained even following protracted time periods. In this instance, it may be more appropriate to study the dynamics of radionuclide transfer and uptake in the ecosystem.

The empirical data collation and model runs have drawn heavily on the methods and information derived in human radiological protection studies. Although this approach is sensible for many reasons, not least because the terminology and methodologies will be familiar to individuals working with human radiological impact assessments, it is recognised that the study of radionuclide transfer in non-human food-webs is in its infancy. Much more information is required to form a complete and accurate picture beyond that given by the current data availability.

3 Assessment methodology

The ecosystem type and reference organisms therein have been defined through exposure pathway analyses as considered in Section 2. This facilitated the appropriate choice of transfer factors and DCCs for use in the following guidelines.

For the main FASSET assessment, the basic components of information that are required to derive dose-rates to organisms are the activity concentrations of radionuclides[4] in (selected) reference biota and their habitat. DCCs mapping these activity concentrations onto a dose rate and occupancy factors defining the time spent by biota in various habitats for the parameterisation of external dose calculations.

The whole-body absorbed dose-rate is used as a measure of the reference organism exposure to the ionising radiation, expressed in units of μGy per hour, and is the sum of internal and external absorbed dose rates:

$$\dot{D}^j_{total} = \dot{D}^j_{\text{int}} + \dot{D}^j_{ext} \qquad (3.1)$$

where,

$\dot{D}^j_{total}$ is the total absorbed dose rate received by the organism *j* (μGy h^{-1}),
$\dot{D}^j_{\text{int}}$ is the internal absorbed dose rate received by the organism *j* (μGy h^{-1}),
$\dot{D}^j_{ext}$ is the external absorbed dose rate received by the organism *j* (μGy h^{-1}).

As discussed in Section 2, and in FASSET Deliverable 3 (Pröhl *et al*., 2003) it may be appropriate to introduce radiation weighting factors to take account of the differing biological effectiveness of different types of ionising radiation. At the present time such consideration is recommended for alpha particle radiation, and for beta particle radiation with mean particle energies less than 10 keV. Introduction of these weighting factors leads to the weighted absorbed dose:

$$\begin{aligned}
\dot{D}^j_{total,weighted} &= \dot{D}^j_{\text{int},weighted} + \dot{D}^j_{ext,weighted} \\
\dot{D}^j_{\text{int},weighted} &= w_{low\beta}\dot{D}^j_{\text{int},low\beta} + \dot{D}^j_{\text{int},\beta\gamma} + w_\alpha \dot{D}^j_{\text{int},\alpha} \\
\dot{D}^j_{ext,weighted} &= w_{low\beta}\dot{D}^j_{ext,low\beta} + \dot{D}^j_{ext,\beta\gamma} + w_\alpha \dot{D}^j_{ext,\alpha}
\end{aligned} \qquad (3.2)$$

where $w_{low\beta}$ *and* $w_{low\alpha}$ are the radiation weighting factors for low energy beta radiation, and alpha radiation, respectively and the subscripts *low*β, $\beta\gamma$, and α denote the contributions to absorbed dose rate from low energy beta particles, other beta particles and gamma ray photons, and alpha particles, respectively.

[4] It should be reiterated that the current assessment is restricted to the radioisotopes of the original list of 20 elements considered in Section 1.

Contributions from low energy beta particles and alpha particles to external radiation will usually be negligible, but may need to be considered for organisms whose dimensions are of the same order as the range of these radiation types in tissue - typically, in the sub-millimetre range.

For simplicity of explanation, Sections 3.1 to 3.3 describe the methods for calculation of absorbed dose rates to organisms. Extension of the method to calculate weighted absorbed dose rates is described in Section 3.4.

3.1 Assessment of the external exposure

The external dose rate is calculated in a slightly different way depending on whether the assessment is for an aquatic or terrestrial environment. This reflects the different dosimetric methodologies that are employed for these two types of ecosystems.

Terrestrial ecosystems

For terrestrial ecosystems, the external dose rate, averaged over different habitats, can be determined by the following equation:

$$\dot{D}_{ext}^{j} = \sum_{z} v_z \sum_{i} C_{zi}^{ref} * DCC_{ext,zi}^{j} \qquad (3.3)$$

where,

C_{zi}^{ref} is the average concentration of the radionuclide *i* in the reference media of a given habitat z (Bq/kg dry weight),

$DCC^{j}_{ext,zi}$ is the dose conversion coefficient for external exposure defined as the ratio between the average concentration of the radionuclide *i* in the reference media corresponding to the habitat *z* and the dose rate to the organism *j* (μGy/h per Bq/kg)

v_z is the occupancy factor, i.e. fraction of the time that the organism *j* expends in the habitat *z*. Information about the habitat of reference organisms can be found in Appendix 2, Section 1.

For terrestrial biota, the reference media is invariably soil, owing to the fact that soil constitutes the most relevant external radiation source to biota from a long-term perspective, as argued by Pröhl *et al.* (2003). For the specific case of biota expending a part or all of their time on soil in the terrestrial environment, a reference media, has been defined for use in the derivation of external DCCs. The external DCCs are based on the following source configurations:

(1) A planar source for biota living *on* soil. A surface roughness of 3 mm has been assumed. Essentially radionuclides are uniformly mixed in the top 3 mm of soil.

(2) A volume source for biota living on soil. The DCCs have been derived for a homogeneously contaminated volume source with a thickness of 10 cm and soil density of 1.6 g cm^{-3}.

The assessor should select the external DCC that best represents the field contamination distribution.

For organisms (essentially fauna) expending part, or all their time below the soil surface, the application of only one DCC, based on a volume source, is possible. The animal target is assumed to be at a depth of 25 cm in a 50 cm thick homogeneously contaminated soil layer.

The basic set of information required to estimate occupancy factors relates only to the fractional period spent on the soil surface and below the soil surface. For the case of plants the height of the critical organs, i.e. meristems, and subsequent choice of appropriate external DCCs is defined by whether the plant of interest is an herb, shrub or tree (for more details see Pröhl *et al.* (2003)). External DCCs for reference terrestrial biota are discussed in Section 4.2 and presented in numeric form in Appendix 1 (Section 2.1).

Aquatic ecosystems

For aquatic systems, external DCCs have been derived for a uniformly contaminated isotropic infinite absorbing medium. At the sediment water interface, organisms are exposed from above and below. In this configuration, a semi-infinite absorbing medium is appropriately considered. The following equation reflects this:

$$\dot{D}^{j}_{ext} = \sum_{i} DCC^{j}_{ext,i} * \left[(v_{water} + 0.5v_{watsurf} + 0.5v_{sedsurf}).C_{water,i} . + (0.5v_{sedsurf} + v_{sed}).C_{sed,i} \right] \qquad (3.4)$$

where:

C_{water} is the average concentration of the radionuclide *i* in water (Bq l^{-1}, dissolved phase)

C_{sed} is the average concentration of the radionuclide *i* in sediment (Bq kg^{-1}, fresh weight)

$DCC^{j}_{ext,i}$ is the dose conversion coefficient for external exposure defined as the ratio between the average concentration of the radionuclide *i* in environment (water or sediment) and the dose rate to the organism *j* (μGy h^{-1} per Bq kg^{-1})

v_{water}, $v_{watsurf}$, $v_{sedsurf}$, and v_{sed} are the occupancy factors, i.e. fraction of time that the organism *j* expends in the water column, at the air-water interface, at the sediment surface and buried in the sediment, respectively. Information about the habitat of reference organisms can be found in Appendix 2, Section 1.

External DCCs for reference aquatic biota are discussed in Section 4.2 and presented in numeric form in Appendix 1 (Section 2.2). For the aquatic environment, it has been assumed in the derivation of DCCs that radionuclides are uniformly distributed in an infinite absorbing medium. No refinements can therefore be made to account for more complex subsurface distributions of radionuclides in the process of external dose-rate calculation.

3.1.1 Estimation of concentrations in reference media

In retrospective assessments (see Figure 3-1) the concentrations in the "reference media" (for example activity concentrations in fresh surface sediment or activity concentrations, dry weight (d.w.) in the top 10 cm of soil) may be available and in this case the external dose rates can be estimated, in a straightforward manner, using equations 3.3 or 3.4. If such data are not directly available, they might be derived from other available data sets.

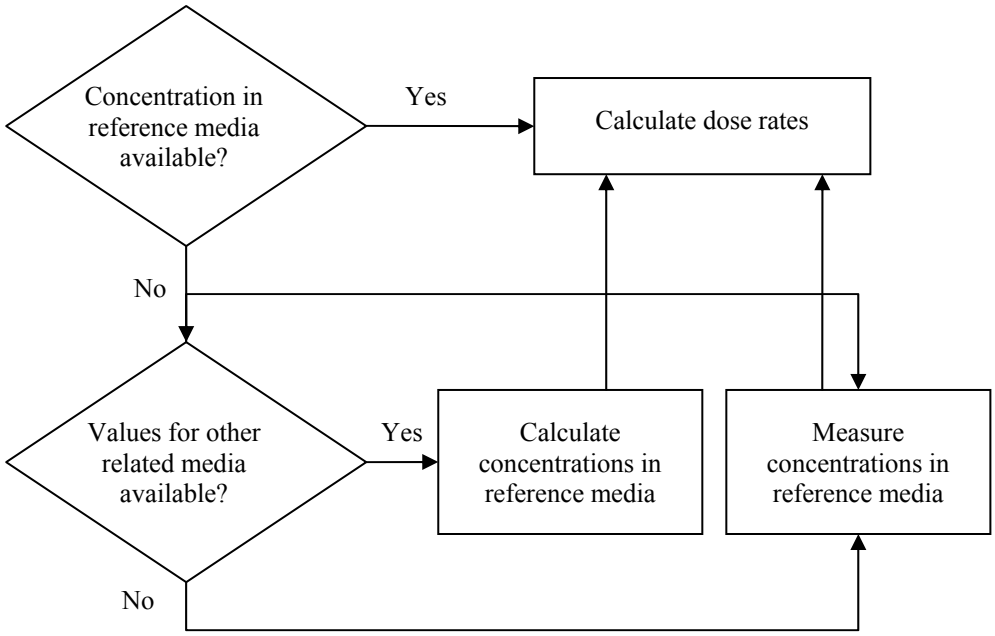

Figure 3-1 Schematic representation of the procedure for estimation of the external exposure of reference organisms. When the concentrations in the reference media are available, the external dose rates can be directly calculated with Equation 3.3 (for terrestrial ecosystems) or Equation 3.4 (for aquatic ecosystems) using the DCC presented in Appendix 1, Section 2. If the concentrations in the reference media are not available, these might be calculated from available values of other related quantities, for example the total deposition in the system. The concentrations in the reference media could also be obtained by measuring environmental samples or by direct measurements in the environment, for example by "in situ" gamma spectrometry. In prospective assessments the concentrations in the reference media are usually calculated with the help of radionuclide transport models.

For instance, if data are not available on activity concentrations in fresh sediment, dry sediment concentrations may be modified using an appropriate correction factor:

$$C_{sed,i,wet} = C_{sed,i,dry}\left(\frac{\rho_{dry}}{\rho_{wet}}\right) \quad (3.5)$$

where:

$C_{sed,i,wet}$ and $C_{sed,i,dry}$ = activity concentrations (Bq kg^{-1}) in wet and dry sediments, respectively.
ρ_{dry} and ρ_{wet} = dry and wet sediment densities (g cm^{-3}), respectively.

Sediment concentrations can be estimated from water concentrations with the help of the distribution coefficients, K_ds :

$$C_{sed,i,dry} = C_{water,i} * K_{d\,i} \quad (3.6)$$

where:

$C_{water,i}$ is the activity concentration (Bq kg^{-1}, Bq l^{-1}) in water,
$K_{d,i}$ is the distribution coefficient between sediments and water, defined as the ratio between the activity concentration in the sediment (Bq kg^{-1}) and in water (Bq kg^{-1} or Bq l^{-1})

Appropriate K_ds for freshwater and marine environments are presented in Sections 4.1.6 and 4.1.7 respectively. For brackish waters K_ds for marine waters may be suitably applied.

Similarly, the activity concentrations in the soil layer 0-10 cm can be estimated from the total inventory (deposition) in the system by the following equation:

$$C_{soil,i} = \frac{Q_i}{d * m} * \varphi_i \quad (3.7)$$

where:

Q_i is the total inventory of the radionuclide in the system (Bq m^{-2})
d is the density of the ten top centimetres layer of the soil (kg m^{-3})
m is the depth of the soil layer and equal to 0.1 (m)
φ_i is the fraction of the radionuclide inventory in the ten top centimetres layer of the soil (r.u)

In prospective assessments the concentrations in reference media are usually calculated with the help of radionuclide transport models.

3.2 Assessment of the internal exposure

The internal dose rate (for biota in both aquatic and terrestrial environments) can be derived from the activity concentration in the selected reference organism using the following equation:

$$\dot{D}^j_{\text{int}} = \sum_i C^j_i * DCC^j_{\text{int},i} \quad (3.8)$$

where:

C_i^j is the average concentration of the radionuclide *i* in the reference organism *j* (Bq kg^{-1} fresh weight),

$DCC_{\text{int},i}^j$ is the radionuclide-specific dose conversion coefficient (DCC) for internal exposure defined as the ratio between the average concentration of the radionuclide *i* in the organism *j* and the dose rate to the organism (μGy h^{-1} per Bq kg^{-1} fresh weight).

Internal DCCs for reference terrestrial and aquatic biota are presented in Appendix 1, Section 2.1 and Appendix 1, Section 2.2 respectively, of this report. Further details, including options on the applications of DCCs, are reported in Pröhl *et al*. (2003).

3.2.1 Deriving activity concentrations in the reference organisms

In retrospective assessments (see Figure 3-2), when the concentrations in the reference organisms are not available, these can be calculated by multiplying the concentrations in the reference media with the appropriated Concentration Ratios (CR).

For the terrestrial ecosystems the CRs are defined as:

$$CR_{b,i} = C_{b,i}/C_{soil,i} \qquad (3.9)$$

Where:

$CR_{b,i}$ = Concentration ratio for reference organism *b* and radionuclide *i* (dimensionless);
$C_{b,i}$ = Activity concentration of radionuclide i in whole body of reference biota (Bq kg^{-1}, fresh weight);
C_{soil} = Activity concentration of radionuclide *i* in surface soil (Bq kg^{-1} dry weight)

Detailed information on the derivation of the CRs for forest, semi-natural pastures and heathlands, agricultural and wetland ecosystems is provided in Section 4.1.2 - 4.1.5. Recommended CRs in tabulated form are presented in Appendix 1; Sections 1.1 to 1.3.

For the aquatic ecosystems the CR, commonly known as Concentration Factors (CF), are defined as:

$$CF_{b,i} = C_{b,i}/C_{aq} \qquad (3.10)$$

Where:
$CF_{b,i}$ = Concentration Factor for reference organism *b* and radionuclide *i* (dimensionless or l kg^{-1});
C_b = Activity concentration of radionuclide *i* in whole body of reference biota (Bq kg^{-1}, fresh weight);
C_{aq} = Activity concentration of radionuclide *i* in aqueous phase (Bq l^{-1} or Bq kg^{-1}) - normally filtered water.

The derivation of appropriate CFs for freshwater, marine and brackish environments are presented in Sections 4.1.6, 4.1.7 and 4.1.8 respectively. Recommended CFs in tabulated form are presented in Appendix 1; Sections 1.4, 1.5 and 1.6 respectively.

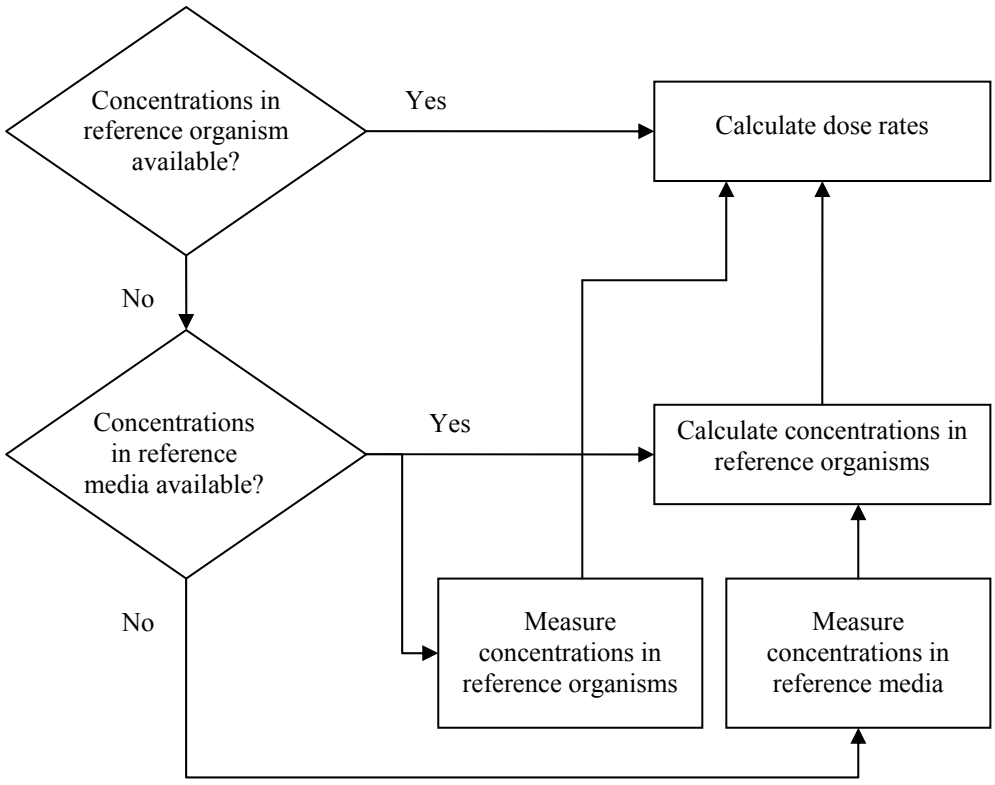

Figure 3-2 Schematic representation of the procedure for estimation of the internal exposure of reference organisms in retrospective assessments. When the concentrations in the reference organisms are available the internal dose rates can be directly calculated with Equation 3.8 using the DCC presented in Appendix 1, Section 2. If the concentrations in the reference organisms are not available, these might be calculated from the concentrations in the reference media (soil for terrestrial ecosystems, water and sediments for aquatic ecosystems) using appropriate transfer factors (Appendix 1, Section 1) or with the help of radioecological models. The concentrations in the reference organisms could also be obtained by measurement of environmental samples.

In prospective assessments the CR (or CF), defined above, can also be used for deriving the activity concentrations in the reference organisms from activity concentrations in the reference media using transport models (see Figure 3-3). This approach is applicable to scenarios where the main radionuclide accumulation pathway in the reference organisms is *via* the reference media. Examples of such scenarios are the long-term phase of an aerial acute contamination, the contamination of terrestrial ecosystems *via* groundwater and most scenarios of contamination of aquatic ecosystems.

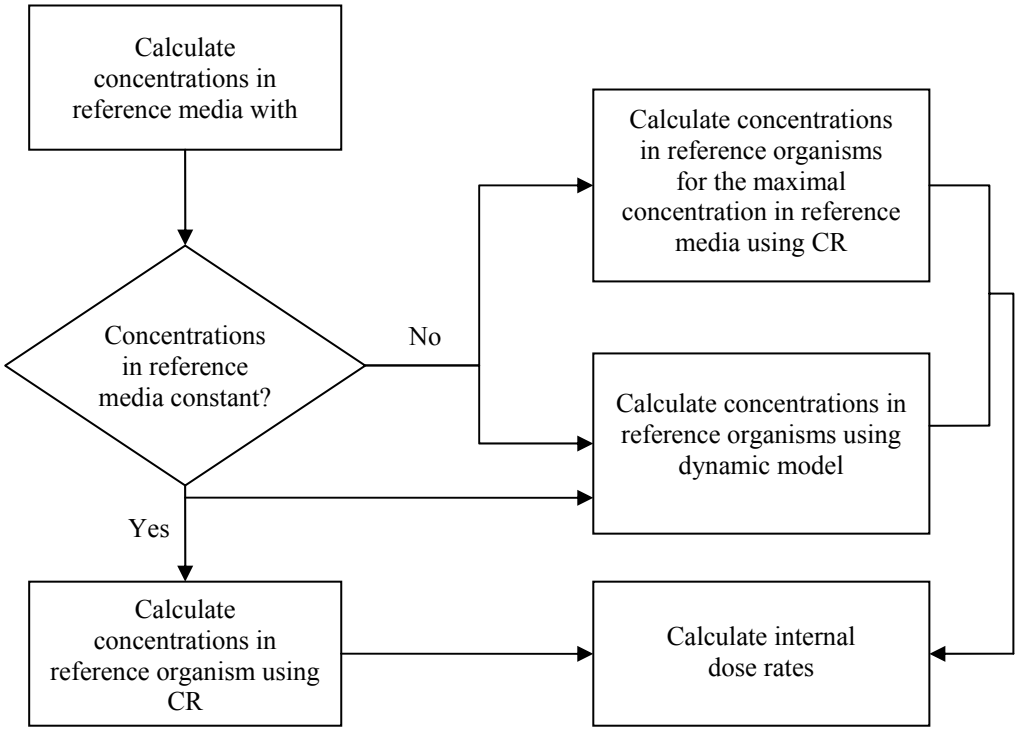

Figure 3-3 Schematic representation of the procedure for estimating the internal exposure of reference organisms in prospective assessments when the accumulation from the reference media is the main contamination pathway. When the calculated concentrations in the reference media (soil for terrestrial ecosystems, water and sediments for aquatic ecosystems) do not vary substantially during the life time of the reference organisms, for example in equilibrium conditions, then the concentration in the reference organisms can be estimated by multiplying the concentrations in the reference media with the appropriated CR (or CF) (Appendix 1, Section 1). If the concentrations in the reference media vary with time, then the concentrations in the reference organisms can be estimated with the help of a dynamic model. A conservative estimate can be obtained by multiplying the maximal concentration in the reference media with the CR (or CF).

In the case of a continuous aerial contamination of terrestrial ecosystems, the use of CR as defined above, may lead to underestimations of the activity concentrations in the reference organisms. This is illustrated in Figure 3-4 which shows predictions, made with the dynamic model "FASTer" (see Appendix 2, Section 5) and with CR, for a chronic aerial deposition of a set of radionuclides above a pasture land. The underestimation of the activity concentrations in mammals is more pronounced for radionuclides with a low soil-to-plant CR, which reflects a larger contribution of direct aerial contamination to the vegetation eaten by mammals.

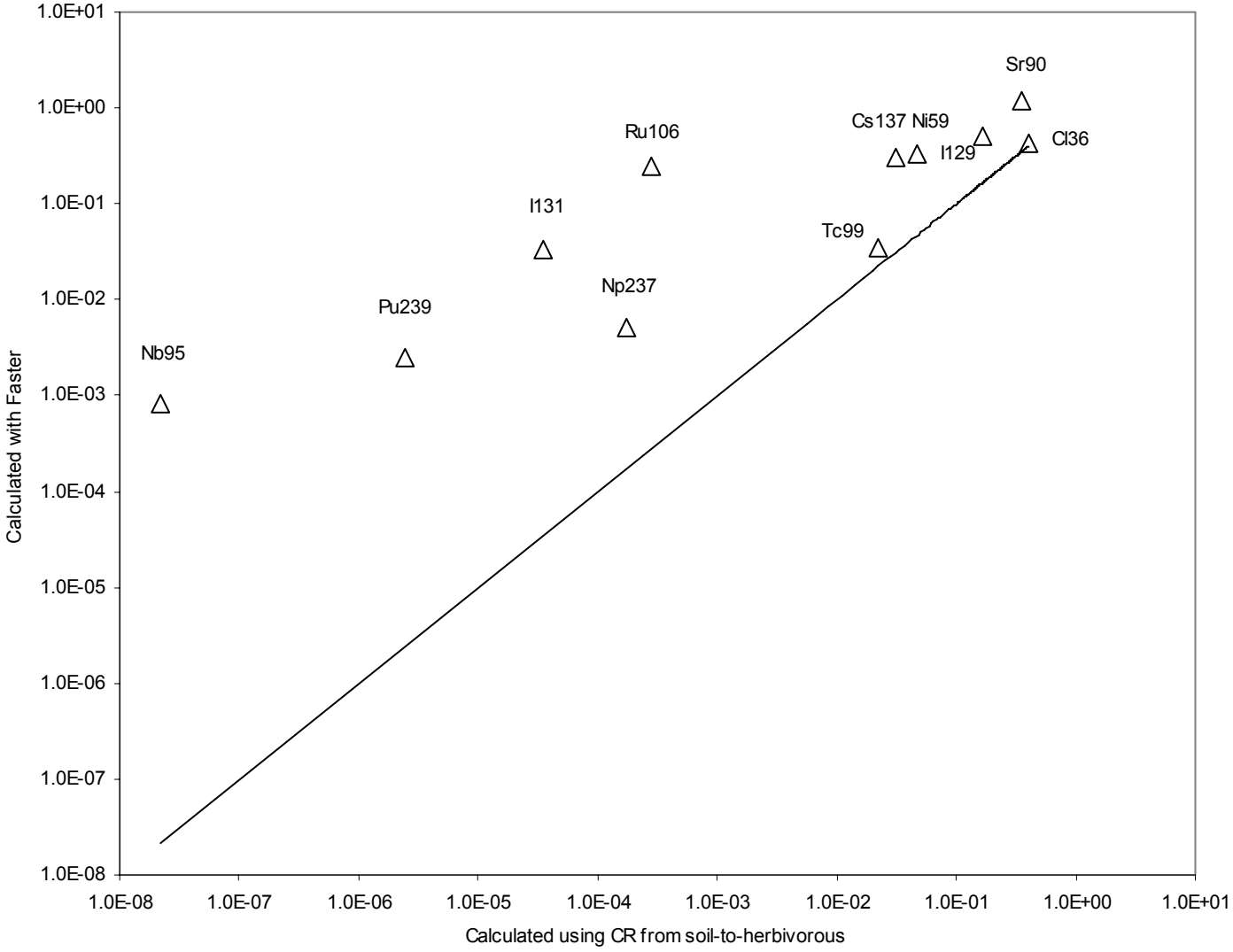

Figure 3-4 Activity concentrations in herbivorous mammals after a continuous constant deposition of 1 Bq/(m^{2}y) above a semi-natural ecosystems calculated with the computer program FASTer (Appendix 2, Section 5) and with the CR in the look-up tables under Section 1.2 in Appendix 1. For radionuclides with low soil-to-plant CR the use of CR leads to underestimation of the activity concentrations in herbivorous mammals.

The methodology recommended for such cases is illustrated in Figure 3-5. In the case of a continuous aerial deposition the activity concentrations in the reference organisms can be obtained by multiplying the deposition rate with a factor "$T_{b,i}$" defined as:

$$T_{b,i} = C_{b,i}/D_i \qquad (3.11)$$

Where:
$T_{b,i}$ = activity concentration relative to annual deposited activity for reference organism b and radionuclide *i* (m^2 y kg^{-1});
$C_{b,i}$ = Activity concentration of radionuclide *i* in whole body of reference biota (Bq kg^{-1}, fresh weight);
D_i = Deposition per year of radionuclide *i* to the ecosystem (Bq m^{-2} y^{-1}).

The derivation of appropriate "$T_{b,i}$" for forest, semi-natural and agricultural environments are presented in Sections 4.1.2, 4.1.3 and 4.1.4 respectively.

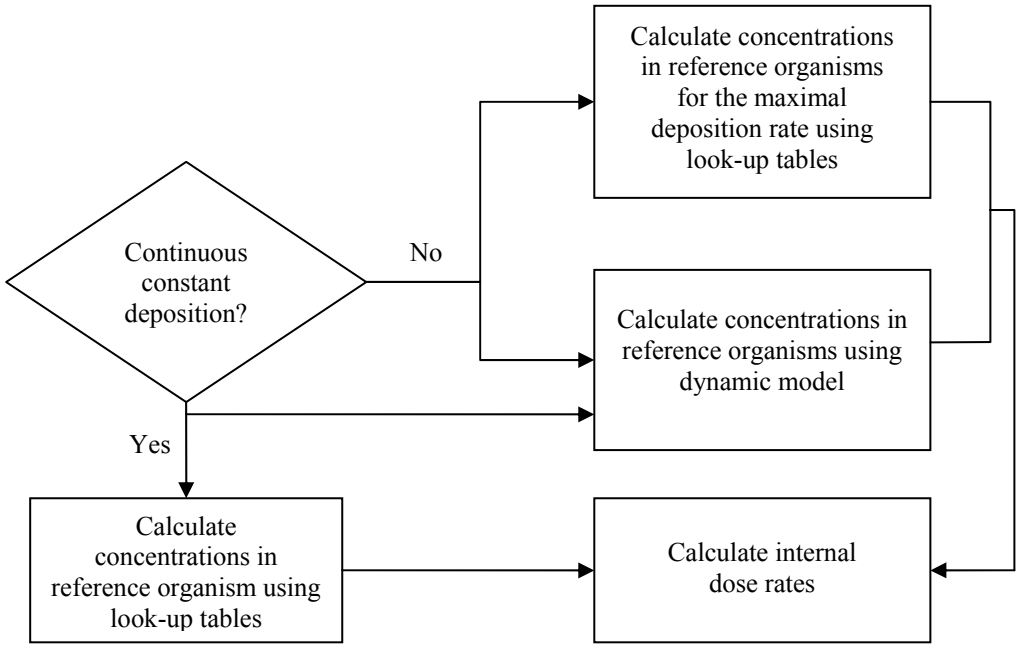

Figure 3-5 Schematic representation of the procedure for estimating the internal exposure of terrestrial reference organisms in prospective assessments in case of an atmospheric deposition. For chronic constant atmospheric deposition the concentrations in the reference organisms can be calculated by multiplying the deposition rate by the values given in the look-up Tables (Appendix 1, Section 1). If the deposition rates vary with time, then the concentrations in the reference organisms can be estimated with the help of a dynamic model. A conservative estimate can be obtained by multiplying the maximal deposition rate with the values given in the look-up Tables (Appendix 1, Section 1).

3.2.2 Limitations in the use of Concentration Ratios

Several procedures, recommended above, for deriving radionuclide concentrations in reference organisms, involve the use of CRs. When applying these procedures it is important to take note of the limitations inherent in using CRs.

One assumption when using a CR is that there is a linear relationship with zero intercept between the concentration in the reference media and the concentration in the reference organism. In the example shown in Figure 3-4, the disagreement in the activity concentrations in herbivorous mammals calculated with the dynamic model and with CR, is precisely due to

deviations from this assumption, which in this case is caused by direct contamination from the air of the vegetation eaten by herbivorous mammals. There can be deviations from linearity even when the contamination is coming mainly from the reference media, because other factors may weight more in the variation of the CR. As a result deviations from linearity will be more pronounced for small intervals of variation of the concentrations in the reference media.

Another assumption is that there is equilibrium between the reference media and the reference organism. Deviations from this condition may lead to erroneous estimations. This can be illustrated with an example relating to lobsters in the Sellafield environment: Shortly after a short-term pulsed release of Tc activity, concentrations of technetium in lobsters sampled along the dispersion path may increase slightly. Lobsters have a biological half-life of elimination for technetium of 60 days or more (66 days for uptake from food and 200-300 days for uptake from seawater (Pentreath, 1981)). However, technetium is soluble in seawater and may clear quickly from the area where lobsters live, aided by tides and currents, so concentration in seawater is likely to decrease sharply within hours-days. If a lobster has been sampled within a few days after the pulse, it may appear to have an anomalous high "concentration factor" because it still retains technetium but the surrounding water does not. It is incorrect to conclude that such a lobster had an abnormal concentration capacity in relation to the water. The anomaly is just conceptual, in trying to capture a snapshot picture of the concentration capacity of the lobster by using a ratio of concentrations that should be taken at equilibrium (as in the laboratory experiments).

The use of concentration factors with regards to assumption of equilibrium can be very limited when investigating short-term environmental releases of radioactivity. Jackson and Vives i Batlle (2000) made the observation that concentrations of ^{99}Tc in seawater exhibit a rapid response phase, whilst seaweed (*Fucus vesiculosus*) and winkles (*Littorina littorea*) do not. This indicates clearly the limitations of using an equilibrium concentration ratio approach to describe the behaviour of ^{99}Tc, when the discharge is of a pulsed nature.

Some of the variability associated with concentration factors and other kinds of partition coefficients can be implicitly incorporated in uncertainty intervals for key model parameters. However, the dynamic picture for environmental compartments that are not under equilibrium can only be adequately described with the help of dynamic models.

3.3 Measurement of activity concentrations in reference organisms (and their habitat)

Provision of detailed Guidelines in relation to the sampling and measurement of radionuclides in the environment are beyond the scope of this handbook. Several publications are of relevance in this respect including Bunzl *et al.* (1999) and IAEA (1989a). However some general observations with respect to environmental impact assessments are necessary. In general, the goals of the assessment guide what information is needed, and subsequently, what kind of monitoring results are important for management. There are plenty of national environmental monitoring programmes for controlling the emissions from nuclear power plants or other nuclear facilities and, for the purpose of environmental impact assessment, it is

desirable that existing programmes are utilised as far as possible, although it is acknowledged that there may be limits to this approach. The monitoring programme should ensure that each different part of the environment is investigated adequately at an optimum cost. Description and documentation of the monitoring should follow the principles of modern quality management.

For terrestrial systems, sampling points are located either in an identified reference area (when monitoring is concerned), or close to a facility emitting radionuclides, taking into account the expected impact point and the prevailing meteorological conditions. If the aim is to monitor the environment surrounding a potential radionuclide source, it is necessary to choose a sampling pattern that covers all dispersion directions. If monitoring a reference area, a sampling pattern should be chosen considering the local variability of the radionuclide distribution, e.g. in forest it should be taken into account that fungal mycelia are able to transport radionuclides. In agricultural areas the soil with deposited radionuclides can be ploughed. Therefore, the sampling depth should exceed the ploughing depth.

Since the variety of aquatic systems is large, it is essential to consider the local circumstances and the characteristics of the receiving water body (e.g. prevailing water currents, a complicated archipelago area or an open sea coast, local topography and water depths) in the design of the monitoring programme. Certain basic rules should be considered:

(1) the monitoring network should cover the main spreading directions of released radionuclides,

(2) the monitoring network should give information about the distribution range and activity concentrations of radionuclides in the environment,

(3) local circumstances should be taken into account in planning of the sampling programme. For instance, the sampling should be focused on the open water period, if the body of water is covered by ice several months a year.

In some cases, it will be necessary to predefine the evaluation area, i.e. site boundary or area of elevated contamination, and then collate data from within these boundaries. The subsequent method of "averaging" data or selecting which data are appropriate for the assessment is currently a point for contention and will depend upon the purpose of the assessment. For example, in the context of an assessment to demonstrate compliance in line with the IAEA criterion of protecting a population of organisms, the *maximally exposed individual* is chosen as the point of reference (IAEA, 1992): the dose rate to this individual must not exceed the defined dose limit. In such a case, it may be necessary to characterise the distribution of values from an empirical data set (e.g. field sampled values of activity concentrations, CFs etc.) and then derive a model generated population from which an appropriate percentile can be selected (see Wilson & Hinton, 2002). Such an approach might be limited by assumptions required about the form of the distribution and the fact that the distribution of absorbed dose rates to individual organisms will not necessarily be the same as the distribution of results for environmental contamination based on samples taken within a particular area - because mobile organisms will receive absorbed dose rates which reflect a spatial average over their home range. Alternatively, the average absorbed dose rates to a relatively small subset of the population (in line with the critical group approach for humans)

would be more tractable, and would be equally valid as an approach to protecting population of wild organisms.

3.4 Weighted absorbed dose-rate calculation

Finally, for the reasons considered at the start of this section, radiation weighting factors may be applied with a resulting modification to the DCCs. Weighting factors of 10 and 3 can be applied for α-radiation and low-energy β radiations respectively, **by way of example**, noting that no single figure weighting factors have been recommended by FASSET at the present time. The DCCs are now modified to become weighted DCCs:

For example, the internal DCCs for a given radionuclide and reference organism

$$\left[DCC^{j}_{\text{int},i,low\beta}\right]_{w} = DCC^{j}_{\text{int},i,low\beta} * w_{low\beta} \quad (3.12)$$

$$\left[DCC^{j}_{\text{int},i,\alpha}\right]_{w} = DCC^{j}_{\text{int},i,\alpha} * w_{\alpha} \quad (3.13)$$

$$\left[DCC^{j}_{\text{int},i,Total}\right]_{w} = \left[DCC^{j}_{\text{int},i,\alpha}\right]_{w} + \left[DCC^{j}_{\text{int},i,low\beta}\right]_{w} + DCC^{j}_{\text{int},i,\beta\gamma} \quad (3.14)$$

where :

$\left[DCC^{j}_{\text{int},i,low\beta}\right]_{w}$; $\left[DCC^{j}_{\text{int},i,\alpha}\right]_{w}$ and $\left[DCC^{j}_{\text{int},i,Total}\right]_{w}$ are "weighted" DCCs for low β, α and all radiation types, respectively. They are specific to radionuclide *i* and reference organism *j*.

$w_{low\beta}$ and w_{α} are radiation weighting factors (default values of 3 and 10 may be used although w_{α} in the range 5-50 can be justified).

$DCC^{j}_{\text{int},i,\beta\gamma}$ is the DCC for β particles (those with mean energies greater than 10 keV) and γ radiation for radionuclide *i* and reference organism *j*.

It should be noted that these weighted DCCs have not been included in the look-up tables presented in Appendix 1. The reader is referred to the Appendix of FASSET Deliverable 3 (Pröhl *et al*., 2003) if further details are required.

3.5 Management of information gaps

There are unfortunately many gaps in our knowledge of the transfer of radionuclides to many wild organisms. Similarly it is only possible to present DCC values for a limited range of geometries. Therefore some recommendations on how to address these deficiencies are required. The following builds upon the approach suggested by Copplestone *et al.* (2003).

Where appropriate values of transfer to reference organisms are unavailable within the look-up tables:

(1) The assessment should clearly state that data are not available.

(2) If data for a specific ecosystem is unavailable consider the suitability of data from other ecosystem types. For instance, transfer values for animals from the semi-natural pastures/heathlands look-up tables could be applied to animals in forest. Indeed, the review of applicable data to derive values for semi-natural pastures/heathlands included data originating from forest (see Appendix 2). Similarly, given the absence of data for wetlands appropriate data for freshwater environments or semi-natural pastures/heathlands could be used.

(3) A transfer (fresh weight activity concentration in organism: fresh weight activity concentration in soil) value of 1 is recommended as being generally conservative for terrestrial environments. There will be exceptions where this assumption is not conservative (e.g. for radiocaesium) but in these case data will generally be available for some organism groups for these radionuclides on which an expert judgement can be based.

(4) For aquatic systems, the highest available concentration factor for a specified radionuclide considering all reference organism types should be compared with the k_d for that radionuclide. The larger number can be selected for the assessment.

(5) Consider if transfer can be justifiably ignored. For some organisms exposed to beta/gamma emitters the total dose is likely to be dominated by external radiation (e.g. a worm inhabiting soil contaminated by gamma-emitters).

(6) For some radionuclides transfer values for radionuclides with a similar biogeochemical behaviour could be employed. For instance, transfer values for Pu could be used to estimate Am activity concentrations.

In instances where the assessor is required to make estimations for an organism not represented by an example reference organism (for which DCC values have been estimated) then a DCC value for an organism of similar geometry and live-weight should be used. For instance, if an assessment for frogs were required suitable DCC may be supplied by the mouse values presented in Appendix 1.

3.6 Guidelines for uncertainty analysis

Uncertainty in the results of an exposure assessment can arise from a number of sources, including conceptual uncertainties in the models applied, uncertainty in the values of the model parameters, uncertainties in the empirical data due to natural variability, measurement errors, biases in the sampling and monitoring. The significance of the overall uncertainty of a dose assessment will depend on how the estimated doses compare with the reference doses used for risk characterisation. If the doses estimated using the values of DCC, TF and models recommended in this handbook, are 2 or more orders of magnitude below the relevant reference doses, then uncertainties are probably not a major issue. We, however, recommend that uncertainty analyses are always conducted as an integral part of the assessment.

This section provides some general guidelines on how to perform uncertainty analyses in exposure assessments. The guidelines are not meant to be comprehensive, but aim at providing an overview of methods that can be applied in such analyses. The methods are generally straightforward and readily accessible, through publications – see, for example, IAEA (1989b), Morgan & Henrion (1990).

3.6.1 Approaches for treatment of uncertainties

An approach frequently used to sort out the uncertainties is to carry out conservative deterministic assessments. The main advantage of this approach is its simplicity, which allows screening to be achieved with a minimum of information. However, problems arise when the reference values, or dose limits, are exceeded or might be exceeded and when the costs for realizing the reference values are high. In such cases, lack of knowledge on the degree of conservatism involved impairs a rational comparison of possible radiological risk to biota against other interests. A common way to deal with this problem is to carry out deterministic assessments in tiers. This means that if a conservative assessment yield values above the reference values, then more detailed, realistic assessments will be carried out. This approach can be seen as a simplified version of a probabilistic approach, where probability distributions assigned to all model parameters are propagated to obtain a probability distribution for the endpoint, which can be used as a measure of the overall uncertainty of the assessment (Figure 3-6).

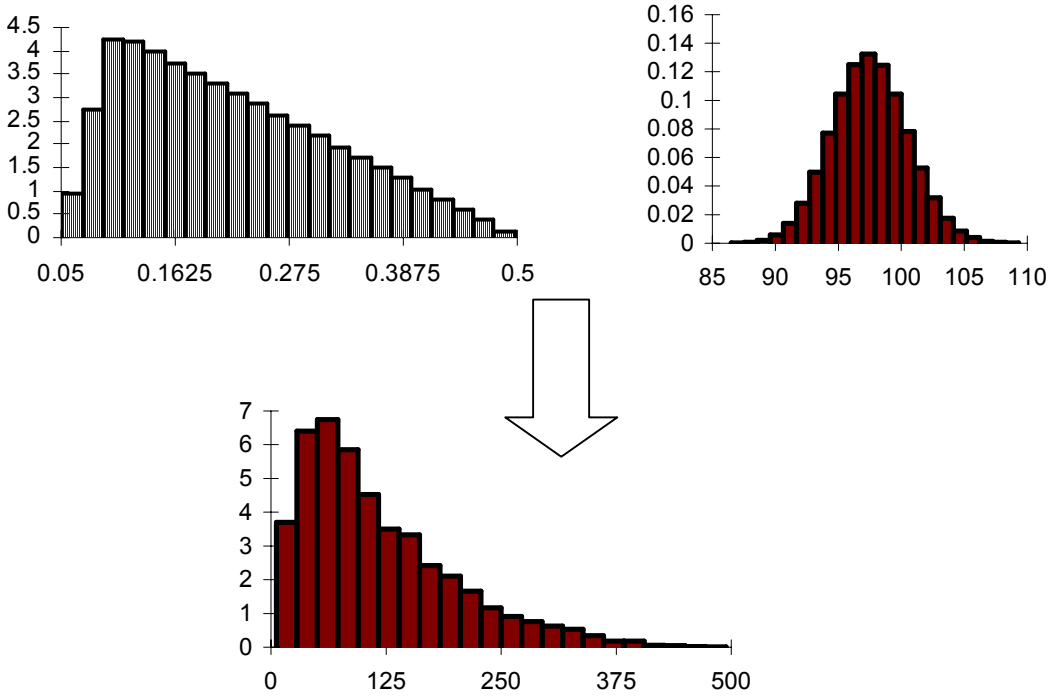

Figure 3-6 Graphic illustration of the probabilistic approach, where probability distributions assigned to the input parameters are propagated to obtain a probability distribution for the assessment endpoint.

In the remaining part of this chapter, we will provide guidelines on how to conduct an uncertainty analysis in order to provide a quantitative estimate of the range expressed by assessment endpoints resulting from uncertainties in the input parameters. Furthermore, the sources that dominate the overall uncertainty will be identified, facilitating the setting of priorities for work aimed at reducing the uncertainty.

3.6.2 Obtaining probability distributions

The first step in an uncertainty analysis is to obtain probability distributions for the parameters and inputs used in the assessment. The probability distributions for uncertain parameters must be carefully constructed, if the uncertainty estimates for the assessments are to be meaningful. When sufficient empirical data are available, the probability distribution of the parameters or inputs can be directly estimated using standard statistical techniques (Taylor, 1993). This would be the case, for example, when activity concentrations in the reference organisms or reference media are obtained by means of environmental monitoring.

The data available for a given parameter are, however, often limited both in quality and quantity. Moreover, some of the existing data may not be consistent with the assessment context. The process of deriving probability distributions is, therefore, largely subjective, and requires specialised knowledge and judgement of each parameter. Principles for the formal

collection and use of expert opinion have received considerable attention in recent years (Hofer 1986). Formal techniques have been, for example, developed to study risks of reactor operation (Hora & Iman 1989). It should be, however, noted that formal expert elicitation is an expensive and time-consuming procedure. Some general guidance on the type of distribution to select for a given amount of available data is given in Bäverstam *et al*. (1994).

3.6.3 Propagating parameter uncertainties

To find the uncertainties in the endpoints of an assessment, the uncertainties in the inputs and parameters must be propagated though the model. A good discussion on this subject is found in IAEA (1989b). When simple analytical expressions for the probability distributions are available, variance propagation can be sometimes applied for propagating the uncertainties (Morgan & Henrion, 1990, Hoffman & Hammonds, 1994). When analytical methods cannot be applied, the uncertainties can be propagated using *Monte Carlo* analysis. The bases of this method are straightforward (see Vose, 1996): point estimates in a model equation are replaced with probability distributions, samples are randomly taken from each distribution, and the results tallied usually in the form of a probability density function or cumulative distribution. Several variations of the Monte Carlo technique for sampling from input distributions are available. One popular version is *Latin hypercube* sampling, which divides the input distributions into intervals of equal probability. Latin hypercube sampling is more precise than conventional Monte Carlo sampling, because the entire range of the input distributions is sampled in a more even, consistent manner (Iman & Helton, 1988).

3.6.4 Methods for ranking uncertain parameters

One of the applications of an uncertainty analysis is to rank the parameters according to their contribution to the assessment endpoint and its uncertainty. The ranking may then provide a criterion to efficiently allocate further research efforts aimed at reducing the overall uncertainty. A large number of methods for ranking uncertain parameters, commonly known as sensitivity analysis, are reported in the literature (for reviews see Rose & Swarzman, 1981, Iman & Helton, 1985 and Mariovet *et al*., 1987). The objective of the sensitivity analysis is to study how changes in input parameters affect the assessment endpoints. A parameter may have an important effect on the results because it is strongly correlated with the endpoints or/and because it has a large uncertainty due to lack of data or it's large "natural" variability.

In its simplest form, sensitivity analysis consists of varying selected input parameters, one at a time, over a specified range and recording the corresponding changes in the model predictions. Those parameters causing the largest relative changes in the predictions are defined as the important model parameters. Correlation and regression coefficients can also be used as a measure of sensitivity (see Vose, 1996). The results are often presented in *Tornado charts* (Figure 3-7). The longer the bar in a Tornado chart, the grater the effect that parameter is having on the models output. It should be noted that rank order correlation makes no assumption about the relationship between inputs and outputs. Least square regression, on the other hand, assumes that there would be a linear relationship between the input and the output variables.

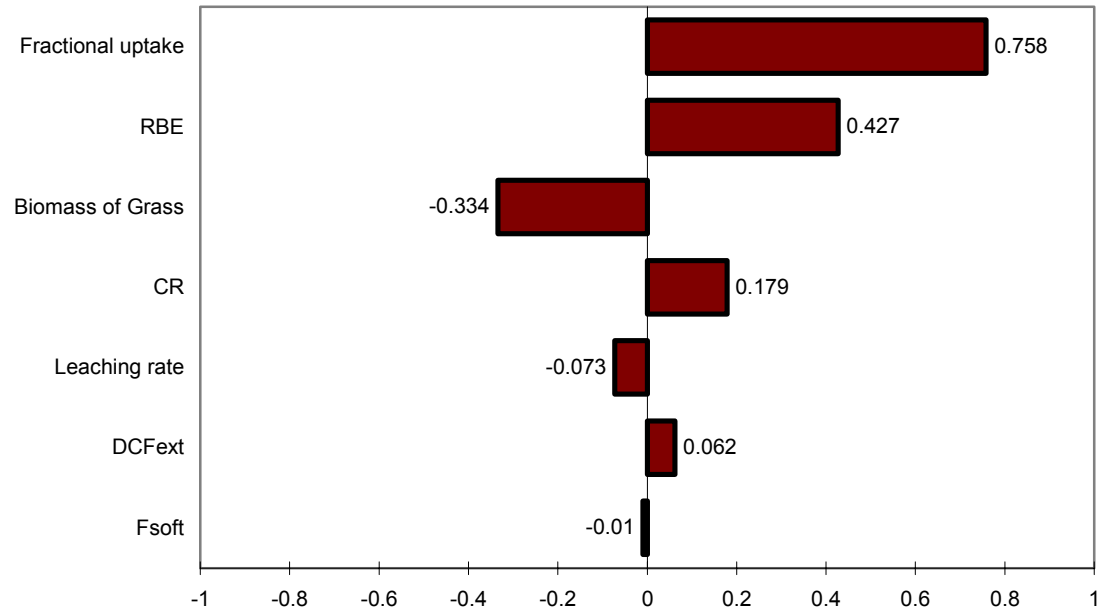

Figure 3-7 Presentation of the results of a Sensitivity Analysis as a Tornado chart showing the Spearman rank order correlation coefficients between the input parameters and the endpoint of the assessment. The longer the bar in the Tornado chart, the greater the effect that this parameter has on the assessment endpoint. This figure pertains to the hypothetical example of ^{239}Pu transfer to a generic rabbit following a unit chronic deposition after 50 years. The simulations were conducted using the FASTer model

The above sensitivity measures will provide a ranking of the influence of each variable on the assessment endpoint, but give no quantitative feel for the contribution that each input is making to the outputs uncertainty. If the model is a simple additive one (with uncorrelated variables), the relative contribution to the outputs uncertainty of each input variable can be fairly accurately estimated by dividing each rank order correlation coefficient by the sum of all coefficients. Figure 3-7 represents the approximate fraction of the total output uncertainty that is being provided by the input in question.

Most assessment models are neither purely additive nor free from correlations, so the above technique will not usually be appropriate. A more generally applicable alternative is to run a number of simulations where, in each simulation, the uncertainty of one variable is removed and replaced by its expected (BE) value (Vose, 1996). After each simulation the standard deviation is recorded as the measure of the uncertainty of each simulation result. The reduction in the output uncertainty is calculated for each of the simulations where the uncertainty of a variable was removed. These figures are then normalised, by dividing each by the sum of all the reductions, to give estimates of the percentage contribution of each variable to the outputs uncertainty. An example of the type of results obtained, in such analyses, is shown in Figure 3-8.

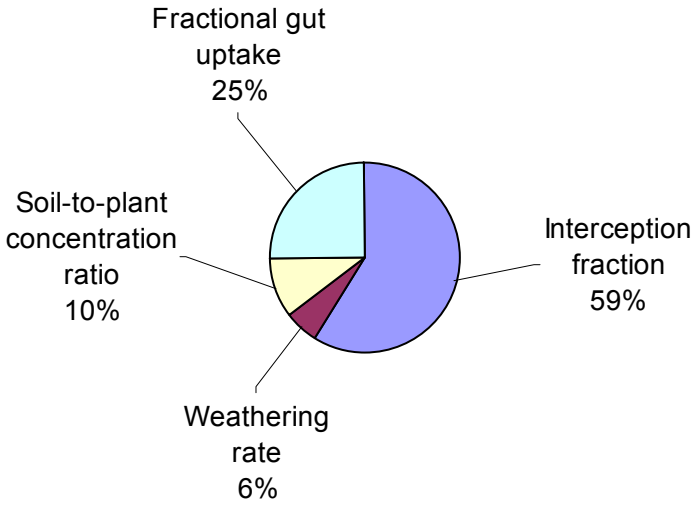

Figure 3-8 Relative contribution of the uncertainty of different inputs on the uncertainty of the assessment endpoint. The results show that a few parameters explain most of the uncertainty of the assessment endpoint, which is a common feature of environmental assessments. This figure pertains to the hypothetical example of ^{137}Cs transfer to a generic mammal following a unit chronic deposition after 50 years. The simulations were conducted using the FASTer model.

4 Transfer Factors and Dose Conversion Coefficient look-up tables: background and derivation

4.1 Transfer Factors

4.1.1 Introduction

The concentrations of radionuclides in biota can be estimated using transfer factors as described in Section 3. The intention with the application of such transfer factors is to provide an aggregated relationship (accumulation factor) between the concentration in specific environmental media and the concentrations in the reference organisms in conditions of equilibrium.

Transfer factors are provided for forest, semi-natural pasture/heathland and agricultural systems in the case of terrestrial systems and freshwater, marine and brackish water for aquatic systems. In addition, methods that can be adopted in the derivation of relevant transfer information are discussed for wetlands and for rivers.

For each ecosystem, an overview of the methods employed in the derivation of look-up table values is presented in this Section.

In view of the large volumes of transfer data generated for each of the ecosystems considered, all look-up table values pertaining to biological transfer have been placed in a separate Appendix (Appendix 1: Tabulated transfer factors). An overview of the tabulated values is given below:

- Forest Ecosystem : Appendix 1, Section 1.1
- Semi-natural pasture and heathland: Appendix 1, Section 1.2
- Agricultural : Appendix 1, Section 1.3
- Freshwater : Appendix 1, Section 1.4
- Marine : Appendix 1, Section 1.5
- Brackish water : Appendix 1, Section 1.6

The initial ambition was to provide values for all cases, i.e. all radionuclide biota combinations, but in numerous instances this ambition was not realised due to lack of information. A column showing confidence levels is also present using the following categories (unless otherwise stated):

(1) **High** - for example, value derived from a large empirical data set or model parameters are all well defined.

(2) **Medium** - for example, derived from a smaller empirical data set or models with more poorly defined parameters,

(3) **Low** - estimate or based on very few data or models with assumed parameters etc.

In line with the recommendations made earlier in the FASSET project whereby "FASSET will provide 'realistic assessments' and will not be overly conservative" (FASSET Deliverable 2 - Larsson *et al*., 2002b) the information presented in the look-up tables are based on **best estimates** in most cases. An exception to this approach is presented for the agricultural system which is based on a conservative screening methodology. However, recognition that the system may require application to cases of compliance where conservative values may be more appropriate, has stimulated the inclusion of additional descriptive statistics (e.g. standard deviations, number of data used to derive the value), in some cases. A column with comments on the nature and sources of the values is also provided.

Further details, concerning the derivations of look-up table values, whether through the collation of empirical data or through the application of appropriate models is provided in Appendix 2 (Underpinning scientific information).

4.1.2 Forests

The Transfer Factors (TF) for forest reference organisms, provided in Appendix 1.1, are expressed in units of Bq/kg per Bq/m^2, either on a dry weight (for plants and fungi) or a wet weight (for animals) basis. This type of Transfer Factors, known as Aggregated Transfer Factors (Tag), is usually preferred for forest systems and most available empirical forest TF are expressed in such units. The main reason for this, is the pronounced vertical heterogeneity exhibited by forest soils, which makes it more difficult to use the traditional CR (expressed in units of Bq/kg per Bq/kg), because of their higher variability. Some of the tabulated values (as indicated in the comments to the tables) were, however, obtained from reported CR, by dividing the latter by a density of 1400 kg/m^3 and a soil thickness of 0.1 m, where most of the radionuclides are usually present.

Most of the reference organisms listed in the FASSET Deliverible1 have been covered with the following exceptions:

- soil fauna
- bird eggs
- detrivorous organisms

The reason for excluding these organisms from the tables is that empirical data or models that could be used for derivation of TF were not available. For the same reason all plants from the herbaceous layer were combined into a single group, which is referred to as "understory vegetation".

The radionuclides covered in the tables are: ^{137}Cs, ^{90}Sr, ^{239}Pu, ^{36}Cl, ^{99}Tc and ^{59}Ni. In the case of ^{137}Cs and ^{90}Sr most of the values provided, with a few exceptions, are based on empirical data. The values for the other nuclides are a combination of empirical data with results of model simulations. Details on how these values were obtained are provided in Appendix 2

(Section 2). Other relevant radionuclides could not be considered because there were almost no data available and suitable models were either non-existent or not accessible.

It should be noted that ranges of values, instead of best estimate (BE), are provided in the tables. This is motivated by the high variability of species, and the very large range of variation expressed by TFs in forest ecosystems. For instance, the variability of TF in a site may cover the same range of variability observed for agricultural plants over many sites. Under such circumstances BE values can only be selected for each specific assessment context. The ranges provided in the tables are considered to cover 50 % or more of the whole (real) interval of variation. However, for those TFs with a low degree of confidence (see tables in Appendix 1) the "real" interval of variation and uncertainty might be much higher.

To obtain a range of variation in the reference organisms, multiply the radionuclide inventory in the system (in Bq/m^2) by the corresponding TF provided in Appendix 1. The range of variation of the TF can be also combined with ranges of variation of the inventory, if available, using interval mathematics. Another possibility is to derive a probability density function from the range, by assuming that the values follow a certain type of distribution, for instance a lognormal distribution. The resulting distributions can be used in probabilistic assessments.

It should be noted that the tabulated TFs are applicable to the "long-term" phase of the contamination. In the case of an aerial deposition one could consider that this phase starts 1-4 years after the deposition event.

4.1.3 Semi-natural pastures and heathlands

4.1.3.1 Introduction

Semi-natural pastures and heathlands incorporate a broad range of ecosystems including mountain (e.g. Alpine pastures) and upland grasslands (e.g. those characteristic of many upland areas of the UK), heath- and shrublands (e.g. Mediterranean garigue), salt marshes and some Arctic ecosystems. Many of these ecosystems, most especially those in the Mediterranean, are species rich with area of noted biodiversity and therefore of conservation status (e.g. EEA, 1998). These ecosystems are termed ”semi-natural“ because, whilst they comprise natural species not introduced by man, they have been influenced by human use, for instance by the grazing of livestock. Indeed many natural semi-grasslands would revert to scrub and woodlands if it were not for their utilisation by man. An overview of these ecosystems within Europe is given within Beresford *et al*. (2001).

Candidate reference organisms for this ecosystem group previously selected for consideration within FASSET (Strand *et al*. 2001) are: soil micro-organisms; soil invertebrates (represented by ‘worms’); detritivorous insects; lichens and bryophytes; grasses and herbs; shrubs; burrowing mammals; herbivorous mammals; carnivorous mammals; eggs of ground nesting birds.

4.1.3.2 Empirical data collations and review

A database including transfer of the FASSET radionuclides (as selected within Strand *et al*. (2001)) from soil to reference organisms was generated predominantly from the following sources:

(1) Literature review (using Web of Science[5]) of English language refereed publications and cited works within these;

(2) Data supplied by Institute of Radiation Hygiene (IRH) for areas with elevated natural radionuclides within the Komi Autonomous Republic of the Russian Federation (Litver *et al*. 1976; Pokarzhhevskii & Krivolutzkii 1997; RCSI 1974-1998; Troitskaya 1981; Verhovskaya 1972[6]) for the EPIC project (see Beresford *et al*. 2003);

(3) Data supplied by IRH (from published Russian language sources and in-house databases) on the transfer of a range of radionuclides to wildlife species from throughout European Russia (with an emphasis on Arctic regions and post Chernobyl studies in the Bryansk Oblast) for the EPIC project (see Beresford *et al*. 2003);

(4) Data for wildlife species within the Chernobyl Exclusion Zone (Gaschak *et al*. in press);

(5) The Arctic Monitoring and Assessment Programme database (see AMAP 1998).

[5] http://wos.mimas.ac.uk/
[6]Original references mostly in Russian – see also Maslov *et al*. (1966) for site description in English.

More than 300 publications (refereed literature, books, institute reports and conference proceedings) were reviewed. Because of the scarcity of suitable data all appropriate terrestrial wild species were considered with no differentiation between habitats (i.e. the database contains some values for wild species inhabiting forests and agricultural land). A considerable number of data were rejected from the review as the level of detail within the original publications was insufficient to enable its use with any degree of confidence (e.g. all collated Th data for grasses and herbs were rejected). Transfer to soil micro-organisms was not included because the absorbed doses for bacteria will be predominately defined by the activity concentrations in the surrounding medium (Pröhl 2003)[7]. A detailed description of this data collation exercise is provided in Appendix 2, Section 3.

The transfer of ^{3}H and ^{14}C from soil to biota was not considered. A detailed description of an approach for predicting the activity concentrations of these two radionuclides in reference organisms is presented in Appendix 2, Section 4.

4.1.3.3 Dynamic Modelling for Semi-natural Ecosystems

There are no bespoke models to enable the dynamic prediction of the wide range of radionuclides to reference organisms as considered here within semi-natural ecosystems. In addition, appropriate observed data are lacking for many radionuclides - reference organism combinations. Furthermore, a requirement to be able to predict exposure of wild organisms in circumstances of chronic release, accidental release and release to ground water (i.e. from deep repositories) was defined within the objectives of FASSET. This can only realistically be achieved by developing a dynamic modelling approach. This can be further justified by the observation that the assumption of 'equilibrium' soil – reference organism CR values will not be conservative in some instances. This is demonstrated in Figure 4-1 which presents a comparison of ^{99}Tc and ^{137}Cs activity concentrations in a small herbivorous mammal assuming equilibrium soil-herbivore concentration ratios with predictions using a dynamic model for a simulation of a constant annual deposition over 50 years.

[7] Readers interested in the uptake of radionuclides by micro-organisms should refer to Keith-Roach & F.R. Livens (2002).

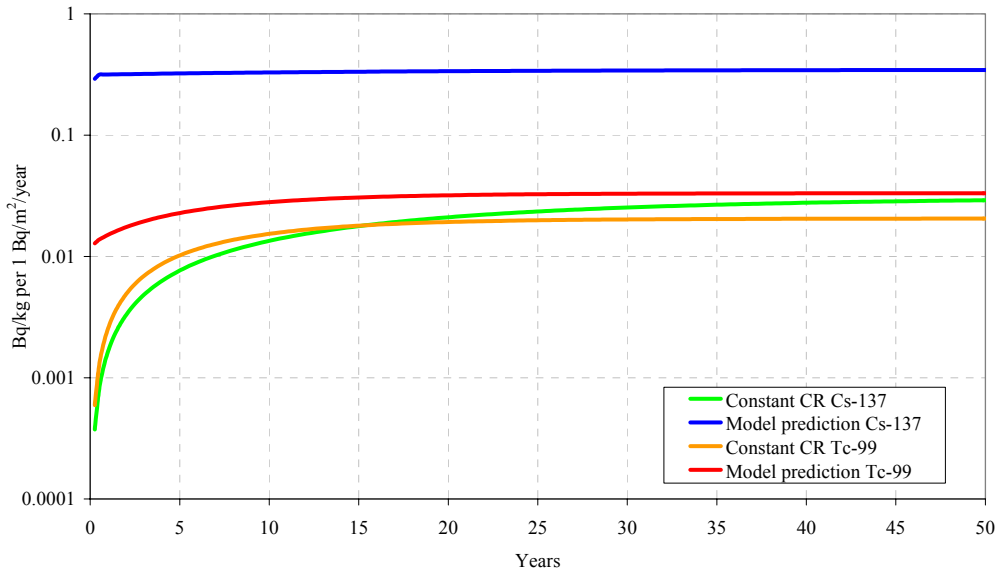

Figure 4-1 Predicted ^{137}Cs and ^{99}Tc activity concentrations in a 1.75 kg herbivorous mammal assuming a constant annual deposition over 50 years. Predictions assuming an equilibrium soil - reference organism CR over the simulation period are compared with predictions using the dynamic model described in Appendix 2 (Section 5).

The development of a dynamic modelling approach for FASSET semi-natural terrestrial ecosystems (FASTer) to enable the prediction of the activity concentrations of radionuclides in a selected number of reference organisms is described in Appendix 2 (Section 5).

4.1.3.4 Look-up tables

For use in assessments of doses to biota 'look-up' tables of transfer factors for two scenarios have been derived on the basis of the data reviews and model development discussed in this section. These are presented in Appendix 1, Section 1.2.

The scenarios considered, for the derivation of look-up table values, are (i) equilibrium transfer from soil to reference organism; (ii) chronic deposition. In the case of equilibrium transfer, concentration ratios (reference organism activity concentration relative to soil activity concentration) the tables present the mean estimate derived from the available empirical data (Appendix 2, Section 3) and also the best estimate FASTer model prediction (i.e. from Appendix 2, Section 5, Table 5-8). A comparison of mean and maximum observed data within Appendix 2, Section 3 suggests that the upper range in CR values is likely to be one order of magnitude higher than the estimated mean. For chronic deposition, values predicted from Appendix 2, Section 5, Table 5-9 (FASTer model), expressing reference organism activity concentration relative to annual deposit are used; observed data are not available for this scenario. Values for ^{3}H and ^{14}C are presented as predicted activity concentrations relative to constant air concentrations; the most conservative estimate from the different climate types modelled are presented within the look-up tables.

4.1.4 Agricultural

4.1.4.1 Conceptual Model

The model used in the derivation of transfer factors for (i) concentration ratio and (ii) activity concentration relative to annual deposited activity, is based on Safety Report Series No. 19 "Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment" (IAEA, 2001). It has been developed for the purpose of screening proposed radioactive discharges; that is for determining through simplified, but conservative assessment, the likely magnitude of the impact.

The modelling approaches described are applicable to continuous or prolonged releases into the environment when it is reasonable to assume that an equilibrium, or quasi-equilibrium, has been established with respect to the released radionuclides and the relevant components of the environment.

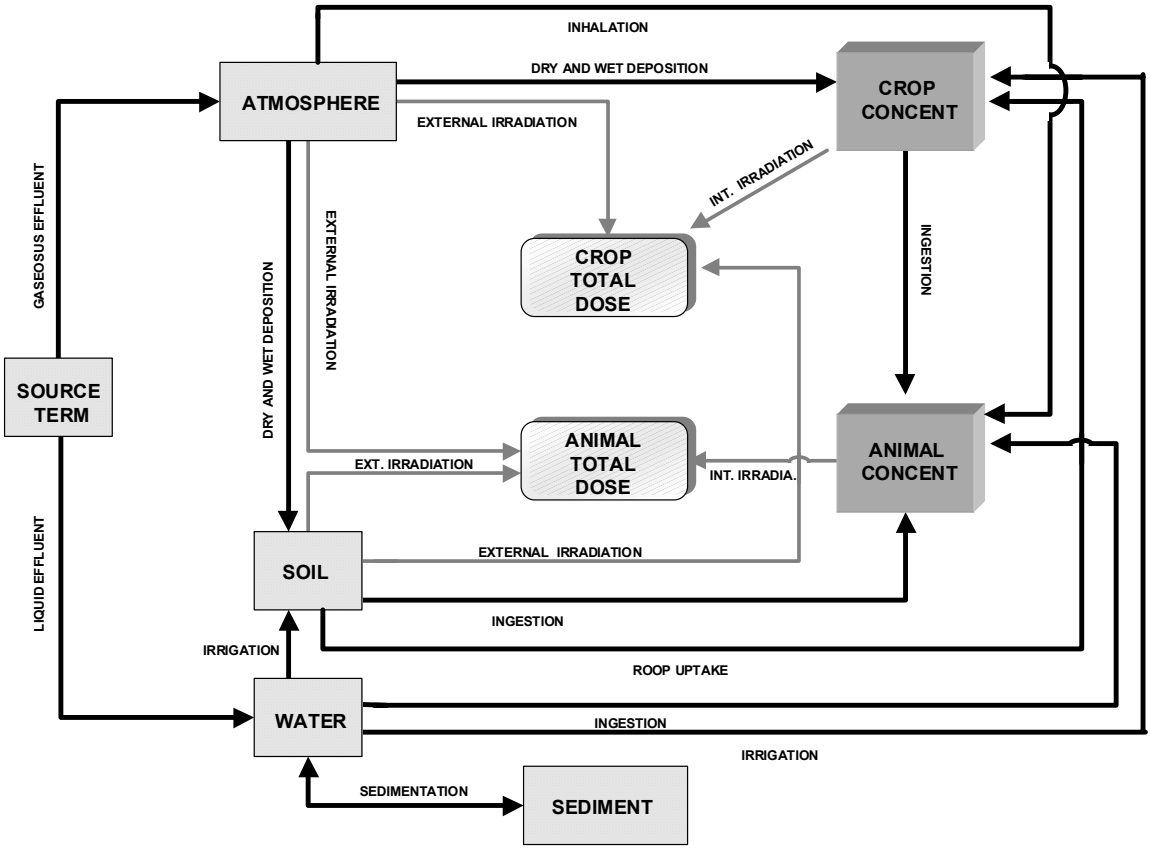

Figure 4-2 Conceptual Model developed for Agricultural Ecosystem

The conceptual model developed for an agricultural ecosystem is shown in Figure 4-2. This includes four compartments representing environmental media (Atmosphere, Soil, Water and Sediment), two compartments representing concentrations in biota (crop activity

concentration and animal activity concentration) and two biota final receptors receiving doses (crop total dose and animal total dose).

4.1.4.2 Activity concentration in vegetation

Radionuclides intercepted by and retained on vegetation may result from fallout, washout, rainout and irrigation with contaminated water or deposition of re-suspended matter from contaminated soil. External deposits can be taken up into plants by foliar absorption. Radionuclides may also be incorporated by uptake from the soil *via* plant roots, followed by internal redistribution of radionuclides within the plant. In the model, processes that can lead to the reduction of radionuclide activity concentration in vegetation include radioactive decay, growth dilution, wash-off of externally deposited radionuclides, leaching and soil fixation. Further removal of radioactive material from vegetation can occur due to grazing, harvesting, etc. Starting from the estimated or measured activity concentrations in air, water and soil, the assessment concentration for each radionuclide is calculated using a common mathematical expression for every vegetable type. The specific parameter values for each vegetable, determine the radionuclide concentration of different crop types (root vegetables, fruit vegetables, leafy vegetables, cereals and fruits) The general expression is:

$$C_{i,veg,hum} = \left[\frac{\dot{d}_i \, \alpha_1 (1 - \exp(-\lambda_{ief}^{v} \, t_e))}{\lambda_{ief}^{v}} + \frac{\dot{d}_i \, (1 - \exp(-\lambda_{ief}^{s} \, t_b)) \, F_{1vi}}{\rho \, \lambda_{ief}^{s}} \right] \exp(-\lambda_i \, t_h) \qquad (4\text{-}1)$$

Where:

$$\lambda_{ief}^{v} = \lambda_i + \lambda_w ; \qquad \lambda_{ief}^{s} = \lambda_i + \lambda_s ; \qquad \dot{d}_i = d_i + C_{i,w} \, I_w$$

Table 4-1 Vegetables - Parameters and variables description

Symbol	Unit	Description
$C_{i,veg,hum}$	Bq kg^{-1}	Radionuclide activity concentration in vegetable
$C_{i,w}$	Bq m^{-3}	Radionuclide activity concentration in water
d_i	Bq m^{-2} d^{-1}	Atmospheric deposition rate
$\dot{d}_i$	Bq m^{-2} d^{-1}	Deposition rate modified due to irrigation (N/A)*
F_{1vi}	---	Soil to plant transfer factor
I_w	m^3 m^{-2} d^{-1}	Irrigation rate
t_b	d	Duration of the discharge of radioactive material
t_e	d	Time period that crops are exposed during the growing season
t_h	d	Delay time between harvest and food consumption (N/A)
α_1	m^2 kg^{-1}	Interception factor
λ_i	s^{-1}	Constant for radioactive decay
λ_{ief}^{s}	s^{-1}	Effective rate constant for reduction of the activity concentration in the root zone of soils
λ_{ief}^{v}	s^{-1}	Effective rate constant for reduction of the activity concentration from crops
λ_{si}	s^{-1}	Rate constant for reduction of the concentration of material deposited in the root zone of soils due to processes other than radioactive decay
λ_{wi}	s^{-1}	Rate constant for reduction of the concentration of material deposited on the plant surface due to processes other than radioactive decay
ρ	kg m^{-2}	Surface density for the effective root zone in soils

* (N/A) Non applicable in this case.

4.1.4.3 Activity concentration in animals

The intake of radionuclides by animals depends on animal species, mass, age and growth rate of the animal, the digestibility of feed, and, in the case of lactating animals, the milk yield. The radionuclide concentrations in meat, milk and eggs are calculated using the following expressions respectively:

$$C_{i,anim} = \left[\sum (F_{i,veg,anim}\ C_{i,veg,anim}\ M_{veg,anim}) + F_{i,water,anim}\ C_{i,w}\ V_{w,anim} \right] \exp(-\lambda_i\ t_{anim}) \quad (4.2)$$

Table 4-2 Animals - Parameters and variables description

Symbol	Unit	Description
$C_{i,anim}$	Bq kg^{-1}	Radionuclide activity concentration in animal
$C_{i,veg,anim}$	Bq kg^{-1}	Radionuclide activity concentration in vegetables
C_{iw}	Bq m^{-3}	Radionuclide activity concentration in water
$F_{i,water,anim}$	d kg^{-1}	Fraction of the animal's water intake of a radionuclide that appear in animal at equilibrium
$F_{i,veg,\ anim}$	d kg^{-1}	Fraction of the animal's daily intake of a radionuclide that appear in meat at equilibrium
F_{2vi}	---	Concentration ratio for uptake of the radionuclide from soil by edible parts of crops
$M_{veg,anim}$	kg d^{-1}	Amount of feed consumed by the animal per day
$V_{w,anim}$		Amount of water consumed by the animal per day (N/A)
t_{anim}	d	Average time between slaughter and consumption
λ_i	s^{-1}	Constant for radioactive decay

4.1.4.4 Reference Organisms

We have identified a set of candidate reference organisms, which have been suggested primarily on radioecological criteria (IUR, 2000; Strand & Larsson, 2001). Based upon the knowledge of the distribution of radionuclides within the environment, a simplified compartmentalisation of the ecosystems has been used: soil, herbaceous layer and canopy for terrestrial ecosystems (Strand *et al.,* 2001). Representative species for each reference organism has been selected and their physical and ecological characteristics defined (Tables 4-3 and 4-4). These data have been used to obtain the look-up table's values.

Table 4-3 Physical and ecological characteristics for vegetable species

FLORA	Latin/common English name	Habitat	Distribution	Average life expectancy
Soil associated plants	*Solanum tuberosum* potatoe	Agricultural, in soil (-20 to 40 cm)	generalised	3-6 months, 1-2 harvest per year, annual crop
	Daucus carota carrot	Agricultural, in soil (-20 to 20 cm)	generalised	1-2 months, continued in the year annual crop
	Allium cepa/onion	Agricultural, in soil (-10 to 50 cm)	generalised	1-2 months, continued in the year annual crop
Herbaceus layer	*Lactuca sativa* lettuce	Agricultural/green house´s agriculture, soil surface	generalised	1-2 months, continued in the year in temperate zones, annual crop
	Lycopersicum esculentum tomato	Agricultural/green house´s agriculture, 5-130 cm	generalised	3-4 months, continued in the year in temperate zones, annual crop
	Triticum sativum wheat	Agricultural, 60 cm	dry and temperated zones	4-6 months, 1 harvest per year, annual crop
Shrubs	*Vitis vinifera* grapevine	Agricultural, 5-100 cm	temperated zones	1 harvest per year, ligneous crop
Trees	*Citrus sinensis* orange	Agricultural, canopy layer	temperated zones, no freeze	1 harvest per year
	Pyrus malus apple	Agricultural, canopy layer	no high temperatures	1 harvest per year
	Olea europaea olive	Agricultural, canopy layer	temperated zones (-10, 35 °C)	1 harvest per year

Table 4-4 Physical and ecological characteristics for animal species

FAUNA.- Herbivorous mammals			
Latin/common English name	Habitat dinamic	Average life expectancy	Feeding habits (kg/d, L/d)
Box taurus cow	12 h Indoor, 12 h outdoor	14-16 y. Slaughtering age: <10 months 10-18 months, 18-36 months	meat cow: water 40, fodder 8, pasture 30 milk cow: water 60, fooder 10, pasture 30
Ovix sp Sheep	12 h Indoor, 12 h outdoor	8-10 y. Slaughtering age: <1.5 months 1.5-3 months, 3-12 months	water 6, pasture 8
Sus sp pig	Indoor, outdoor	10-12 y. Slaughtering age: <1 months 6-8 months	water 5, grain 3, root 2, milk 3

4.1.4.5 Look-up tables for agricultural systems

To build the look-up tables, two different release scenarios have been considered. In the first one, a continuous atmospheric deposition of 1 Bq m^{-2} is considered and in the second one, a homogeneous soil activity concentration of 1 Bq kg^{-1} is considered. Calculations were performed for 34 radioisotopes of the 20 radionuclides selected for FASSET (see Strand *et al*., 2001 and Section 1.1) and for each reference organism. The concentration in crops and farm animals have been calculated using the CROM code (Suáñez & Robles, 1998), developed by CIEMAT following the IAEA methodology (IAEA, 2001).

The 'soil-to-plant' and 'feed-to-animal' transfer factors used in the calculation have been selected based on the literature review and are reported in FASSET Deliverable 1 (Strand *et al*., 2001)

From the values obtained, it is possible to point out that, ^{36}Cl and ^{99}Tc are the most important radionuclides due to their high impact on flora and fauna, followed by ^{90}Sr, ^{129}I, ^{137}Cs, ^{135}Cs, ^{59}Ni, ^{63}Ni, ^{237}Np and ^{210}Pb.

Comparatively, the contamination due to direct deposition on vegetation is more important than that due to uniform concentrations in soil. Depending on the radionuclide selected, the difference in the values obtained ranges from 3 to 9 orders of magnitude.

Transfer factor look-up tables for agricultural systems are presented in Appendix 1, Section 1.3.

4.1.5 Wetlands

Although wetlands are unique, widespread and very important ecosystems, they have received very little attention in radiological assessments, and very scarce information is available on radionuclide accumulation and transfer in wetlands. More research is unquestionably needed in this area, particularly in light of the fact that wetlands are often important conservation areas.

Wetlands are situated at the interface between land and water, both in marine, as well as in freshwater areas. They are areas of high biodiversity and biological productivity, comparable to tropical rain forests and coral reefs. Wetlands are considered critical resources due to their numerous services and functions such as protecting and improving water quality, flood control, river regulation, erosion control, sediment retention, supply of fuel, and providing wildlife habitats. They are fairly widespread over Europe, but vary widely because of regional and local differences in soils, topography, climate, hydrology, water chemistry, vegetation, and other factors including human disturbance.

Wetlands act as natural water filters and it has been shown that many pollutants accumulate in these areas. The accumulation of many radionuclides in wetland ecosystems is a product of the extreme chemical gradients and large amounts of organic matter to which these radionuclides are exposed. The chemical conditions and the presence of organic matter promote adsorption of radionuclides onto organic particulates. Many of these contaminated particulates are then retained in the wetland system due the high degree of sedimentation that occurs. Wetlands can therefore be important sinks for radionuclides. Radiological assessment for these environments has however been limited, probably due to the lack of easily identifiable direct pathways to humans.

Levels of some radionuclides in peat have been assessed, however, mainly due to the urban uses of peat. The mean radionuclide activity concentrations in peat were found to be in the range 2.5 – 500 Bq/kg dry weight peat for a radionuclide suite consisting of radioisotopes of U, Ra, Th, Pb, Po and Cs (see Table 5 -1, in FASSET Deliverable 1, Appendix 1, Strand *et al*., 2001).

4.1.5.1 Transfer of radionuclides to reference organisms

Very little information is available on accumulation of radionuclides in wetland species and ecosystems. Limited data are available on caesium accumulation in the frog *Rana arvalis* from a wetland area North of Gävle in Sweden where ^{137}Cs accumulated compared to nearby areas, after the Chernobyl accident in 1986. The concentration ratio of ^{137}Cs for *Rana arvalis* in this area was found to be 0.8 Bq kg^{-1} $_{\text{organism dry weight (d.w.).}}$ / $Bqkg^{-1}$ $_{\text{soil d.w.}}$, and 19144 $Bqkg^{-1}$ $_{\text{organism d.w.}}$ / Bql^{-1} $_{\text{water}}$ (Stark, 2001). The concentration ratio of iodine for aquatic amphibian species (a mean of 17 species including tadpoles) is 130 $Bqkg^{-1}$ $_{\text{fresh weight (f.w.)}}$ / $Bqkg^{-1}$.

In the absence of a comprehensive data-set pertaining to transfer of radionuclides in wetland systems, it may be necessary to employ surrogate transfer factors derived for semi-natural and freshwater systems.

4.1.6 Freshwater (lakes)

4.1.6.1 Sediment water distribution coefficients

Values of K_d are affected by numerous factors including sediment type and water quality. They can vary by several orders of magnitude for a given radionuclide. It is therefore important to carefully select a value that is appropriate for the specific site under consideration. Usually, the finer the sediment, the higher the K_d value for a given radionuclide under the same water quality conditions. The suspended sediment concentration varies greatly depending on the characteristics of the lake. If a site-specific K_d value is not available, default values from Table 4-5 can be used. The IAEA has derived recommended screening values for K_ds from stable element data.

Table 4-5 Recommended screening values for K_d (L/kg) for FASSET radionuclides in natural freshwater environments (IAEA, 1994 and IAEA, 2001).

Radionuclide	K_d (L/kg) Expected	K_d (L/kg) Range
Am	5000	90 - 40000
C	5	n.a.
Cm	5000	10 - 70000
Cs	1000	50 - 80000
I	10	0 - 80
Np	10	0,2 - 100
Pu	100000	100 - 10000000
Ra	500	100 - 1000
Ru	500	n.a.
Sr	1000	8 - 4000
Tc	5	0 - 100
Th	10000	1000 - 1000000
U	50	20 - 1000

n.a. = not available

4.1.6.2 Derivation of concentration factors

The derivation for concentration factors for the reference organisms (defined in Section 2.2) and the suite of FASSET radionuclides (defined in Section 1.1) was empirically-based. The reference organisms for freshwater, selected within FASSET, are: phytoplankton, zooplankton, vascular plants, gastropods, bivalve molluscs, crustaceans, insect larvae, benthic fish, pelagic fish, amphibians, birds and mammals. Open literature was collated for the transfer data and a database of concentration factors for freshwater organisms was developed. The database was developed in ACCESS format and contains approximately 700 data entries on concentration factors of radionuclides between organisms and water. In addition to data on CFs between water and whole organisms, CFs pertaining to various tissues or organs of organisms were also collated.

Transfer data derived from the database were used to form the look-up tables for various radionuclides.

The look-up tables and further details concerning references used are presented in (Appendix 1, Section 1.4).

Activity concentrations of radionuclides in freshwater biota vary widely. The levels are influenced by (1) the water chemistry, which can affect the uptake of radionuclides in biota, and (2) by the hydro-geochemistry of the catchment area, which can affect the influx of radionuclides into the lake. Therefore, it is difficult to interpret the existing data in terms of mean values and ranges for European freshwaters, and even more difficult to obtain the missing values by extrapolation or by some other means.

Macrophytes were considered to represent the group of vascular plants. However, in spite of the problems associated with the degree to which concentration factors vary, mean values with the standard deviation (SD) were calculated if several values for one organism group and radionuclide were available. If only one value was available, uncertainty or variation of the concentration factor was not estimated. The data used in the look-up tables was predominantly derived from studies in N. America, while many data on natural radionuclides were from India.

Many data gaps on concentration factors of freshwater biota have been identified. No data on H, C, K, Ni, Nb, Tc, Ru and Pb were found for freshwater environment. The most studied artificial radionuclide is ^{137}Cs, but even for this radionuclide, data coverage does not extend to all the reference organisms considered within FASSET (Table 4-6).It should, in most cases, be possible to estimate whole-body concentrations of selected faunal groups through the application of biokinetic and allometric relationships. Such methods have been employed with some success for both marine and terrestrial (forest and semi-natural) ecosystems. Further work in this direction is envisaged.

Table 4-6 Data gaps of concentration factors in freshwater environment (x = data exist).

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

4.1.7 Marine

4.1.7.1 Transport models

Following release to the marine system, radionuclides will be prone to (i) dispersion and advection via hydrodynamic processes, (ii) interaction with suspended particulate material that may be deposited to the sea-bed sediment and be subject to both sediment mixing (e.g. bioturbation) and sediment transport processes and (iii) interaction with marine organisms leading to uptake and transfer within marine food chains. These processes can be simulated using appropriate models. In some cases, it is difficult to separate food chain models from physical transport (dispersion and sediment interaction) models because there are instances in which the two model types interact in ways other than the obvious case of the advection/dispersion model acting as an input source to the food chain model (Coughtrey & Thorne, 1983). Nonetheless, for the purpose of this assessment a distinction has been made between these two model types on presentational grounds.

A comprehensive description of models applied in simulating the physical transport governed by the processes of hydrodynamics, advection-dispersion and sediment transport is presented in Appendix 2, Section 6. The focus for this marine section of the report, in line with the approach taken for other ecosystems, has been weighted towards the derivation of data pertaining to biological uptake and transfer. Nevertheless, a cursory consideration of solid phase: water phase distribution coefficients, K_ds, will be made initially in acknowledgement of the fact that numerous marine assessment models employ such parameters in the derivation of sediment activity concentrations (see for example IMO, 1983; Iosjpe *et al*., 2002)

4.1.7.2 Sediment water distribution coefficients

An updated collation of data pertaining to K_ds has recently been undertaken by the IAEA (IAEA, in press) for deep ocean and ocean margins. The data set provides generic values applicable to all world oceans. In view of the fact that (i) releases, from a European maritime perspective, are most likely to occur to coastal waters (as discussed above) and (ii) there is no reason to expect that distribution coefficients for European marine environments will differ significantly from those based on world average values, the IAEA K_d values for ocean margins have been adopted for this report (Table 4-7).

The IAEA have, in most cases, derived "recommended" ocean margin K_ds from stable element data. Dissolved element concentrations are derived from direct measurements made in open oceans and coastal water data, where possible. Concentrations of elements have been derived for sediments based on direct measurement, mean shale compositions or mean crustal abundances. In view of the facts that K_ds relate to the fraction of the sediment-bound element/radionuclide in equilibrium with the aqueous phase an estimate of the exchangeable proportion of the element associated with particulate material is required. The IAEA assume that, for all elements except C, 20 % of the total concentration of the element in silts and clays represents the exchangeable component of the element. This was considered to account for the varying proportion of coarse sediment (essentially not involved in exchange processes) observed in coastal sediments and empirical information pertaining to the proportion of

elements associated with fine sediments (i.e. geochemical phase association data). The limitations associated with the various assumptions made in this derivation should be acknowledged and site specific information should be utilized where possible.

The IAEA no longer quote a range for recommended K_ds but instead provide advice on how an appropriate range might be derived. For example, in the case of conducting a sensitivity analysis, an arbitrary range can be derived by, for example assuming maximum and minimum values that fall a factor of 10 higher and lower than the recommended value.

Table 4-7 Distribution coefficients for "FASSET" radionuclides in ocean margins (IAEA in press)

Radionuclide	K_d	Radionuclide	K_d
Cs	4 000	Pb	100 000
Tc	100	Po	20 000 000
Sr	8	C	1 000
U	1 000	H	1
Th	3 000 000	Nb	800 000
Pu	100 000	Ni	20 000
Am	2 000 000	Ru	40 000
Cm	2 000 000	I	70
Np	1 000	Cl	0.03
Ra	2 000		

It should be noted that although K_ds may provide a reasonable estimate of activity concentrations in deposited surface sediments, in cases where the reference biota is submerged within the sediment, e.g. polychaete worm, this value may be of limited use. In such cases, models simulating early diagenetic processes (e.g. post-depositional mobilisation-desorption, sedimentation, consolidation etc.) may need to be invoked (see for example, Smith *et al*., 1995).

4.1.7.3 Biological uptake models - Concentration factors

Concentrations factors[8] have been widely used in modelling the transfer of radionuclides from the water column to biota and numerous reviews and summaries of the available literature have been made (Coughtrey & Thorne, 1983; Harrison, 1986; Gomez *et al*. 1991). Probably the most widely–used concentration factor values, in the fulfilment of human dose assessments, are those reported by the International Atomic Energy Agency in Technical Report Series 247 (IAEA, 1985) and the updated version of this document. The CF approach has the advantage of being simple and provides the assessor with a large and easily-accessible data-base. It therefore provides a useful starting point for an assessment of transfer and uptake of radionuclides within marine ecosystems. However, although the generic organism groups considered in IAEA (1985) and IAEA (in press) are similar, and in some cases identical, to

[8] The concentration factor (CF) is usually defined as the ratio of the concentration of the radionuclide in the organism or tissue (normally fresh weight) to that in (normally filtered) seawater.

those selected as reference organisms within FASSET (Table 4-8), the applicability of these data to the present work is partly limited.

Table 4-8 (Candidate) reference organism categories selected in FASSET (see Strand *et al*., 2001)

Bacteria	Crustacean	Mammal
Worm	Bivalve Mollusc	Wading bird
Vascular plant	Benthic fish	Phytoplankton
Macroalgae	Pelagic Fish	Zooplankton

In view of the fact that the intended use of CF data would be in human dose assessments, the approach adopted in IAEA (1985) and IAEA (in press) involved the collation of data for organism forming parts of food chains leading to man, i.e. edible plants and animals. Furthermore, the information was, where possible, reported for the edible body parts of these organisms. Clearly, a question of data compatibility exists here. Within marine environmental impact assessments, organisms forming parts of food chains that have no connection with man should be given equal consideration to those dealt with in human dose-assessment. It is also of importance to consider not only those parts of an organism eaten by man but also those body parts that might be of interest from a dosimetric or dose-effects perspective for the organism, *per se*. Such organs/body parts might include, where relevant, the hepatic system (where high accumulation of heavy metal contaminants can occur) and gonads (important from the perspective of a fertility endpoint). Indeed the IAEA (IAEA, in press) acknowledge these points with the words:

"*The biological data compiled in the present study are likely to be of limited value for predicting radiological effects on biota. The focus of this report…would allow assessment of the potential risks associated with human consumption of edible fractions. The distribution of radionuclides in specific organs will be more critical for assessing harm to the organism, and is beyond the scope of this report*".

In view of these limitations, a data collation exercise was conducted in order to derive information that would be of use in an environmental impact assessment. The full review forms part of the report Appendices (Appendix 2, Section 7). Where appropriate, information has been extracted from this review for inclusion in the look-up tables (Appendix 1, Section 1.5).

Table 4-9 illustrates the coverage of empirical data derived from the present review and IAEA (in press). The grey boxes represent the presence of 1 or more data entries and the blank boxes the absence of data for the given radionuclide-biota intersect. During the data collation exercise conducted within FASSET (Appendix 2, Section 7), little information was found for radionuclides such as I, Ru, Ra, Np, Cm and U; no data were found for Th, C, H, Nb, Ni and Cl. No data for vascular plants were found. Marine birds, (polychaete) worms and mammals are particularly poorly characterised using CF datasets.

Table 4-9 CF Data coverage in the present study and IAEA (in press)

Elem.	Macro Algae	Vascular plant	Mollusc	Crusta-cean	Fish[a]	Zoo-plankton	Phyto-plankton	Worm	Mammal	Bird
Cs										
Tc										
Sr										
U										
Th										
Pu										
Am										
Cm										
Np										
Ra										
Pb										
Po										
C										
H										
Nb										
Ni										
Ru										
I										
Cl										

[a] Includes both benthic and pelagic fish as reported in the main body of text.

4.1.7.4 Limitations with the use of concentration factors in the marine environment

The CF approach is open to criticism because:

(1) it provides no information concerning the types of processes/mechanisms in operation during biological uptake,

(2) the relationship between the radionuclide concentration in water and within (the organs or whole body of) a high trophic-level organism, deriving most of its contaminant load from ingested food, may not be a simple, linear one,

(3) the assumption that the system is under equilibrium, a requirement for CFs to be truly applicable, is often invalid,

(4) Even if the generic data for the world oceans are employed (from IAEA, 1985), with the limitations on use considered in Section 4.1.7.3 having been accepted, the uptake of many radionuclides to certain reference organism types are poorly, if at all, described. A good example can be presented for sea mammals and birds for which data coverage extends only to a handful of radionuclides and where the great preponderance of data exists for ^{137}Cs.

It was concluded by Coughtrey & Thorne (1983) in a comprehensive review on this theme that "*the use of one concentration factor for either marine organisms in general, for the same organism in different sites, for studies involving chronic compared to acute contamination, for short-lived compare to long-lived nuclides, ...for open-ocean compared to coastal sites and for specific animal tissues compared to whole animals is highly unsatisfactory*".

Other approaches to modelling the transfer of radionuclides in ecological systems will, therefore, be explored in this section. Biokinetic models may allow more realistic prognoses concerning the dynamic response of an ecological system to be made and allow tentative estimates to be derived concerning equilibrium CFs. Where data are lacking on some of the parameters required for simulation, allometric[9] relationships may provide surrogate values.

4.1.7.5 Dynamic radioecological models

There are a number of factors that should be taken into account when performing dynamic radioecological modelling. These include:

- The time required for equilibrium to be attained depends on the physical half-life of the radionuclide and the biological half-life of the element in the organism (Till and Meyer, 1983).
- The physicochemical form of the element and its route of entry into the organism are among other factors that affect the CF value.
- Radionuclides may exist in different physicochemical forms with a distribution that varies according to the radionuclide and the features of the ecosystem under consideration.
- Environmental factors, including temperature, light (in the case of algae), salinity and pH affect the growth and metabolism of organisms, and consequently the uptake of radionuclides (Meinhold & Hamilton, 1991).

Coughtrey & Thorne (1983) noted that one of the factors that most affects the active uptake in aquatic organisms is the chemical composition of the medium and in many cases the level of dissolved organic matter is the most important factor. The uptake routes for radionuclides (related in part to the trophic level of the organism), falling into categories including, for example:- direct adsorption, ingestion of organic particles or biota, ingestion of inorganic particles, intake of seawater during osmotic regulation etc. will also affect the dynamics of body activity concentrations and actual levels observed for particular organisms. For example, bivalve molluscs might be expected to attain relatively high body loadings of particle-reactive radionuclides because their food source consists of suspended particulate matter. The actual whole-body concentrations will of course depend upon the degree to which the radionuclide is assimilated by the mollusc. It should be noted that from an environmental impact perspective, the activity associated with contaminated particles in transit through the gut might, depending on the degree of contamination and the radionuclide, be a significant source of exposure. This is particularly the case for radionuclides that are not assimilated to a great degree.

The balance between intake and excretion will define the activity concentration of radionuclides with time within the body of an animal. It has long been recognized that metabolic rate and thus food energy requirement is closely related to body weight. Furthermore, animals exhibit reasonably consistent relationships between the long-component retention half-time and body weight (Whicker & Shultz, 1982). In other words, both the uptake and retention of radionuclides is driven to some extent by the size of the animal. Metabolic rate will in turn be influenced by ambient temperature because organisms may need

[9] The allometric approach is based on the observation that many metabolic parameters, including basal metabolic rates, ingestion rates, biological half times etc., are related (as power functions) to the masses of organisms.

to expend more energy to maintain body temperatures in cold water environments. In some cases, large differences in uptake between taxonomically similar groups of organism are observed but a deep mechanistic understanding has not been developed. Uptake of Tc, for example, may be related to physiological specialisation, for certain classes of biota, where phylogenetically more primitive forms exhibit higher Technetium CFs than more advanced forms (Swift 1989), but more profound elucidations are not currently possible.

In theory all of these, and also other food chain, processes can be simulated but model parameterisation is often a major stumbling block. Within FASSET several models have been used in the consideration of dynamic food chain transfer. Further details are provided in FASSET Deliverable 1 (Strand *et al*., 2001) and selected publications (see, for example, Vives I Batlle *et al*. (2002) and Olsen & Vives I Batlle (2003)).

As discussed above, there are instances where CF data are not available for a particular radionuclide-reference organism combination. In this case dynamic radioecological models may provide some insight into the working of the biological system and allow estimates to be made of steady state CFs for simple scenarios e.g. unit activity concentration in the water column. Such an approach has been adopted for the derivation of CFs for numerous radionuclides in sea mammals and seabirds (Appendix 2, Section 8).

Of special note, in this context, is the definition of sea mammal and seabird for use in the dynamic models. In the parameterisation process, it was considered necessary to identify a particular animal type at the family or species level. More specifically, information was required in relation to prey type, ingestion rate and excretion rates, all of which will vary according to the animal selected. An adult cetacean, for example, would be characterized by a quite different set of parameters compared to an adult pinniped. In this study, representatives of the reference organism sea mammal and sea bird were selected to be a Harp seal (*Phoca groenlandica*) and a Common eider (*Somateria mollissima*) respectively. Details relating to life history can be found in (Appendix 2, Section 1.3). In particular the weight of the adult was important in deriving allometrically based parameters for simulation purposes.

Finally, a further problem in relation to the parameterisation for dynamic models should be noted. In some cases allometrically derived elimination rates were not available. In this case retention factors for man (see ICRP-30, parts 1-4) were employed (through the use of a multi-compartmental box model) in order to simulate excretion. The appropriateness of such factors is clearly of some concern and leads to large uncertainty within some CF values.

Concentration factor look-up tables for marine ecosystems (derived from both empirical data review and dynamic modelling) are presented in Appendix 1, Section 1.5.

4.1.8 Brackish waters

The Baltic Sea was chosen as a target area to represent European brackish water ecosystems. Thus, the reference organisms are typical for this marine area. The salinity of surface water of the Baltic Sea gradually changes as one moves from the Danish Straits towards the northern regions, and this affects the uptake of many radionuclides (especially that of caesium) by organisms. The data given in the look-up table represent conditions in the Northern Baltic Sea with a salinity of 4-6 ‰ (i.e. the areas of the Bothnian Sea and the Gulf of Finland). Besides salinity, other environmental factors, and in particular the amount of Chernobyl fallout, also affect the intensity of uptake. The Baltic Sea was the marine area most affected by the Chernobyl accident (Povinec *et al*., 1996), and the fallout from Chernobyl was very unevenly dispersed in the drainage area of the Baltic Sea. The areas of the Bothnian Sea and the eastern part of the Gulf of Finland received most of the deposition.

4.1.8.1 Concentration factors for ^{14}C for coastal areas of the Baltic Sea

As a complement to the look-up tables for Cs, Sr and Pu for brackish water environments, concentration factors for ^{14}C have been derived from an ecosystem model for the environmental transport of ^{14}C for a coastal area of the Baltic Sea. The model, which is described in detail in Kumblad *et al*. (in press), was developed for a safety assessment to predict the fate of a hypothetical discharge of radioactive carbon from the underground Swedish final repository for radioactive operational waste (located in Öregrundsgrepen, Baltic Sea). The development of the model involved identification, quantification and dynamic modelling of the main flows and storages of carbon both in the physical environment and in the food web of the bay above the repository. In the model, ^{14}C was introduced into the food web *via* photosynthesising organisms and then its transfer through the food web was modelled as being proportional to non-radioactive carbon. The ecosystem structure and the metabolic rates of the organisms in the area were also taken into account.

The ^{14}C contamination of the modelled ecosystem was assessed assuming a release of 51.3 MBq/year for 1000 years. The discharge rate was the best estimate of the average annual discharge from the repository in case of a leakage (Lindgren *et al*., 2001). In the modelling study the implications of changes to the route of ^{14}C entry into the food web and water exchange were examined in three different simulations, A, B and C (Table 4-10.). In simulation A, ^{14}C was assumed to enter the system in the aphotic zone and thus be taken up homogenously by all plants in proportion to their primary production rate and the ^{14}C content in the dissolved fraction of the water. In simulation B, ^{14}C was assumed to enter the system in the photic zone and, therefore, primarily taken up by benthic plants directly from the discharge (not diluted in the water). In simulation C, the ^{14}C assimilation was modelled as in simulation A, but the water exchange rate was reduced by a factor of ten.

Table 4-10 Description of modelling simulations (A, B and C) for which concentration factors have been derived

Simulation	A	B	C
Discharge zone	Aphotic zone	Photic zone	Aphotic zone
Uptake pathway for C-14	By all plants from the dissolved fraction of the water	By benthic plants directly from the discharge	By all plants from the dissolved fraction of the water
Water exchange rate	Normal	Normal	Reduced by ten

Concentration factor look-up tables for brackish water ecosystems (derived from both data review and dynamic modelling) are presented in Appendix 1, Section 1.6.

4.1.9 Rivers

Biological transfer factors have not been derived explicitly for river ecosystems. It can be assumed that the CFs recommended for freshwater ecosystems may be appropriately applied in most cases. Work on river systems has focussed on the physical transfer of radionuclides and the subsequent derivation of a methodology to calculate activity concentrations in water, suspended load and deposited sediments under specified conditions. Full details of this methodology are provided in Appendix 2, Section 9.

4.2 Dose Conversion Coefficients

The methods employed in the derivation of dose conversion coefficients (DCCs) for terrestrial and aquatic ecosystems have been considered in some detail by Pröhl *et al*., (2003). It is therefore unnecessary to repeat this information here. However, the tabulated DCCs themselves have been extracted from Pröhl *et al*., (2003), for easy access by the assessor within this handbook. These data are provided in Appendix 1 of this report under the following sections (Table 4-11):

Table 4-11 Where to find relevant DCCs for the impact assessment

Category	Source target configuration	Where in Appendix 1 ?
Terrestrial	Unweighted DCCs for external exposure for organism that live *on soil* for a planar source with a surface roughness of 3 mm	Section 2.1; Table 2.1.1
Terrestrial	Unweighted DCCs for external exposure of organisms that live *on soil* for a homogeneously contaminated volume source; the thickness of the contaminated soil layer is 10 cm, the soil density is 1.6 g/cm³.	Section 2.1; Table 2.1.2
Terrestrial	Unweighted DCCs for external exposure of organisms that live *in soil* for a homogeneously volume source; the thickness of the contaminated soil layer is 50 cm, the soil density is 1.6 g/cm³, the organisms live at a depth of 25 cm.	Section 2.1; Table 2.1.3
Terrestrial	Unweighted DCCs for external exposure of the critical organs of plants. The values are given for meristem of grass and for buds of a shrub and a tree for a planar source with a surface roughness of 3 mm and volume source with a depth of 10 cm.	Section 2.1; Table 2.1.4
Terrestrial	Unweighted DCCs for internal exposure for terrestrial organisms, the activity is homogeneously distributed in the organisms.	Section 2.1; Table 2.1.5
Aquatic - freshwater	Unweighted DCCs for external exposure of freshwater-estuarine organisms. The DCC is applicable for sediment or water.	Section 2.2; Table 2.2.1
Aquatic - freshwater	Unweighted DCCs for internal exposure of freshwater-estuarine organisms, the activity is homogeneously distributed in the organisms.	Section 2.2; Table 2.2.2
Aquatic - marine	Unweighted DCCs for external exposure of coastal-estuarine organisms. The DCC is applicable for sediment or water.	Section 2.2; Table 2.2.3
Aquatic - marine	Unweighted DCCs for internal exposure of coastal-estuarine organisms, the activity is homogeneously distributed in the organisms.	Section 2.2; Table 2.2.4

It should be noted that all data pertain to unweighted DCCs (essentially an absorbed dose in units of μGy h^{-1} per unit activity) for terrestrial and aquatic environment. In many cases, it may be necessary to apply a radiation weighting factor to account for the relative biological effectiveness of low β and α radiation. Under these conditions, it will be necessary to refer to the original reference, i.e. Pröhl *et al*., (2003), wherein DCCs have been split into their component parts.

5 Examples of application

In order to facilitate the application of the exposure assessment methodology described in the preceding Sections of this report, three examples of application are provided.

5.1 Marine system

5.1.1 Model description

An exposure assessment for marine flora and fauna has to cover whole processes such as dispersion of radionuclides in oceanic space, transfer of radioactivity between sea water and sediments, uptake of radionuclides by biota and, finally, dose calculations. Here, the modelling approach for environmental impact assessment described in Iosjpe *et al*., (2002) and Iosjpe *et al*., (2003) is applied to a generic marine box, the latter having been described by IAEA (2003).

The system of differential equations which describes the present application of the model, is of the form:

$$\frac{dA_i}{dt} = \sum_{j=1}^{n} k_{ji} A_j - \sum_{j=1}^{n} k_{ij} A_i - k_i A_i + Q_i, \qquad (5.1)$$

where $k_{ii}=0$ for all i, A_i and A_j are activities (Bq) at time t in boxes i and j; k_{ij} and k_{ji} are rates of transfer (y^{-1}) between boxes i and j; k_i is an effective rate of transfer of activity (y^{-1}) from box i taking into account loss of material from the compartment without transfer to another, for example radioactive decay; Q_i is a source of input into box i (Bq y^{-1}); n is the number of boxes in the system.

Another assumption is that, at any given time, the activity in the water column is partitioned between the water phase and the suspended sediment material. The fraction of the activity (F_W) in the water column, which is in solution, is given by:

$$F_W = \frac{1}{1 + K_d SSL}, \qquad (5.2)$$

where K_d is the sediment-water distribution coefficient and SSL the suspended sediment load.

Activity on suspended sediments is lost to the underlying boxes when particles settle by gravitation. The fractional transfer from the water column (box i) to the sediments (box j) due to sedimentation is given by:

$$k_{ij} \equiv \frac{K_d SR_i}{d_i (1 + K_d SSL_i)}, \qquad (5.3)$$

where d_i is the mean water depth and SR_i the mass sedimentation rate.

The model also includes the processes of diffusivity of radioactivity through the pore water, resuspension, mixing due to bioturbation and process leading to the burial of activity in deep sediment. Radioactive decay is included in all compartments.

The model can provide information concerning the dispersion of radionuclides in water, sediment and biota phases of marine environment using site-specific data. The generic marine box is characterized as follows: water column with volume of $2 \cdot 10^9$ m^3 and depth of 70 m, flux of water of $4 \cdot 10^{10}$ m^3 y^{-1}, thickness of sediment layer of 0.1 m, suspended sediment load in water column of $3 \cdot 10^{-3}$ kg m^{-3}, sedimentation rate of 5 kg m^{-2} y^{-1}. The generic marine box is totally immersed within a "world ocean box" with volume of $1 \cdot 10^{18}$ m^3 and depth of $3.8 \cdot 10^3$ m.

5.1.2 Exposure assessment methodology

The generic list of reference organism, as specified in Table 2-1, Section 2.2 of this report has been adopted for further analyses. Simulations have been subsequently run for phytoplankton, zooplankton, pelagic fish, sea bird and mammal, mollusc, crustaceans and benthic fish. Further consideration of "representative" reference organisms was deemed unnecessary for this particular analysis because of the highly generic, desk-top-based, nature of this work. For example, simplifying assumptions have been made about occupancy factors (see below) and no particular species required special attention.

The model applied here allows predictions to be made for activity concentrations in surface sediment. It is assumed that these activity concentrations are representative for all depths in sediments, i.e. there is an implicit assumption that contamination is mixed homogeneously to infinite depth.

The activity concentrations associated with the reference organisms has been calculated by applying appropriate generic CFs, from FASSET Deliverable 5 look-up tables, to radionuclide concentrations in filtered sea water derived from model prediction. By way of example, ^{137}Cs and ^{239}Pu have been selected for further analyses. The relevant information can be found in Appendix 1, Section 1.5 wherein CFs for Cs and Pu are presented in Tables 1.5.10 and Tables 1.5.16, respectively.

A decision was made to present the data in terms of weighted absorbed dose rates using a weighting factor of 3 for low β and a weighting factor of 10 for α radiations. Simplifying assumptions have been made with respect to occupancy factors, *v*. For all benthic biota (mollusc, crustaceans and benthic fish) it has been assumed that the organism is continually present at the sediment-water interface at all times. For pelagic biota (phytoplankton, zooplankton, pelagic fish sea-bird and mammal), it has been assumed that the organism is totally immersed in water at all times.

Dose conversion coefficients have been extracted from the relevant tables in FASSET Deliverable 3 (Pröhl *et al*., 2003) – see Tables in Appendix of Pröhl *et al*., (2003) – Section 9.2, DCCs for aquatic reference organisms, Section 9.2.1.

Having collated all necessary information the formulae presented in Section 3 (Equations 3.4 and 3.8 for external and internal dose-rate estimation respectively) have been incorporated into the main computer code. In view of the assumptions used with respect to occupancy factors the

external absorbed dose-rate equations can be simplified for pelagic organisms (Equation 5.4) and benthic organisms (Equation 5.5) respectively:

$$\dot{D}_{ext}^{j} = \sum_{i} DCC_{ext,i}^{j} * C_{water,i} \quad (5.4)$$

$$\dot{D}_{ext}^{j} = \sum_{i} DCC_{ext,i}^{j} * \left[0.5C_{water,i} + 0.5C_{sed,i}\right] \quad (5.5)$$

where:

C_{water} is the average concentration of the radionuclide *i* in water (Bq l^{-1}, dissolved phase)

C_{sed} is the average concentration of the radionuclide *i* in sediment (Bq kg^{-1}, fresh weight)

$DCC^{j}_{ext,i}$ is the dose conversion coefficient for external exposure defined as the ratio between the average concentration of the radionuclide *i* in environment (water or sediment) and the dose rate to the organism *j* (μGy h^{-1} per Bq kg^{-1})

All simulations have been made for a 1 TBq continuous release of radionuclides into the generic marine box.

5.1.3 Results from model runs

Outputs from simulations using the generic box model are illustrated in Figures 5-1 to 5-3 and Table 5-1. Results indicate that for ^{137}Cs, reference organisms can be divided into two groups defined by habitat, i.e. pelagic and benthic organisms. Organisms from the same group exhibit similar dose-rate dynamics. The concentration dynamics of ^{137}Cs in the water column and sediment, as illustrated in Figure 5-1, determine the dynamics of dose rates for all marine organisms. Typical results for dose rates in benthic and pelagic fish, split into components of internal and external dose rates, are illustrated in Figures 5-2 and 5-3 respectively.

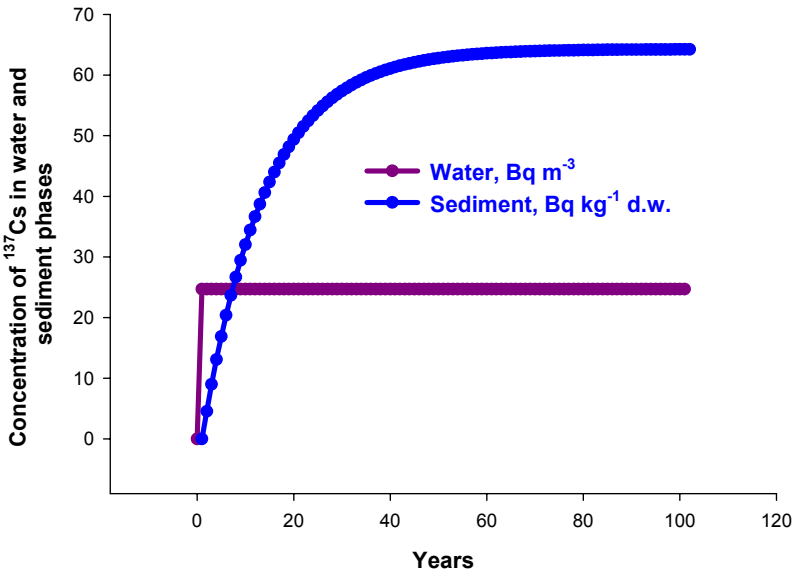

Figure 5-1 ^{137}Cs dynamic concentrations in the marine environment following a 1TBq release to a generic compartment.

Figures 5-2 and 5-3 indicate that for ^{137}Cs, external doses, which vary as a function of sediment-water distribution coefficients, dominate for benthic organisms, whereas internal doses, which vary as a function of bioaccumulation as defined by a CF, dominate for pelagic organisms.

Dose rates to marine organisms for ^{239}Pu are determined solely by internal exposure. Dose rates will therefore be strongly influenced by CF and internal dose conversion coefficients. Results of calculations for ^{239}Pu are shown in Table 5-1.

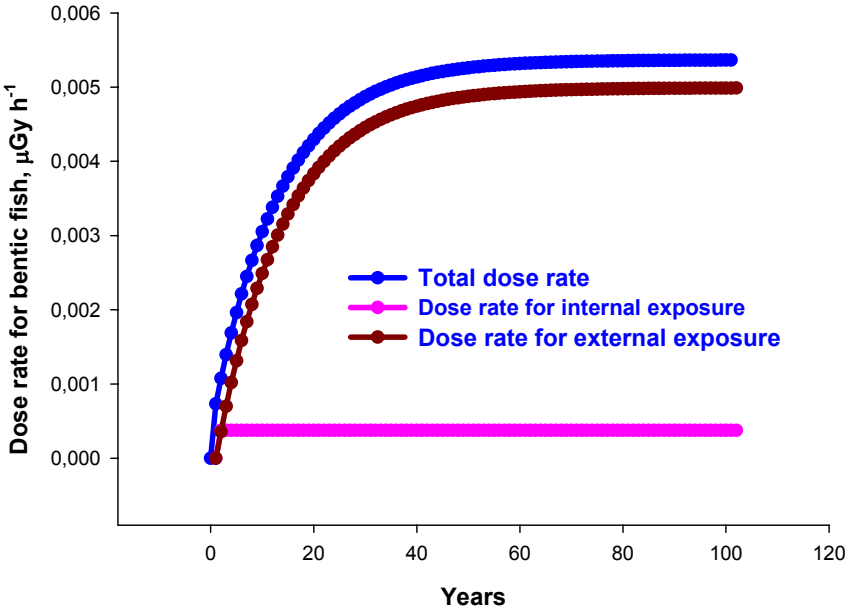

Figure 5-2 Dose rate dynamic ^{137}Cs for benthic fish.

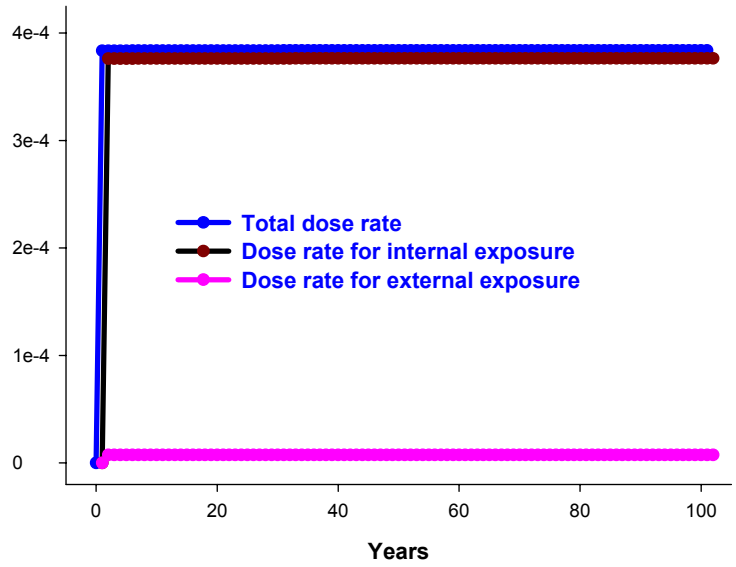

Figure 5-3 Dose rate dynamic ^{137}Cs for pelagic fish.

Table 5-1 Dose rates for marine organisms from ^{239}Pu exposure, µGy h^{-1}.

Phyto-plankton	Macro-algae	Zoo-plankton	Mollusc	Crusta-cean	Bentic fish	Pelagic fish	Sea bird	Mammal
9	2	2	0.6	0.09	0.05	0.05	0.2	0.004

5.2 Terrestrial ecosystems – example 1

For the purpose of demonstrating the exposure assessment procedure for terrestrial ecosystem we will assume a semi-natural pasture/heathland with activity concentrations in soil of 10 kBq ^{137}Cs kg^{-1} (dry weight) and 1 kBq ^{239}Pu kg^{-1} (dry weight). To demonstrate all stages of the assessment we will assume that no measurements of activity concentrations in biota are available.

5.2.1 Selection of concentration ratios

The first step in calculating exposure is the selection of concentration ratios (CR) for reference organisms. For ^{137}Cs Table 1.2.10 (Appendix 1) contains observed values of the transfer from soil to reference organisms (summarised in Table 5-2). In the case of ^{239}Pu no observed data are available for soil invertebrates or carnivorous mammals (Table 1.2.16; Appendix 1). However, for carnivorous mammals we have an estimated value from the

FASTer model which can be employed (although the assessor should note that discussion of the FASTer predictions in Appendix 2 notes that those for actinide elements may be low). For soil invertebrates we can assume the value for detritivores as this is the highest observed concentration ratio (i.e. following the procedure suggested in Section 3.5 to address data gaps). Concentration ratios for plants reference organisms are not required as we are estimating doses to the meristem of grass and buds of shrubs.

Estimated activity concentrations (see Table 5-2) can then be estimated by:

Activity concentration in biota (Bq kg^{-1} f.w.) = CR × Soil activity concentration (Bq kg^{-1} d.w.)

Table 5-2 CR values and predicted whole-body ^{137}Cs and ^{239}Pu activity concentrations for the terrestrial assessment example.

Reference organism	^{137}Cs CR	Estimated Bq ^{137}Cs kg^{-1} (f.w.)	^{239}Pu CR	Estimated Bq kg^{-1} ^{239}Pu (f.w.)
Soil invertebrate[a]	5.66x10^{-2}	570	2.16x10^{-1}	220
Grass	-	-	-	-
Shrub	-	-	-	-
Detritivorous invertebrate	8.49x10^{-2}	850	2.16x10^{-1}	220
Carnivorous mammal[b]	4.96	49600	1.60x10^{-7}	1.6x10^{-4}
Herbivorous mammal	1.84	18400	1.82x10^{-3}	2

[a]Pu-239 CR assumed to be the same as for detritivorous invertebrate (i.e. the highest observed value)
[b]Pu-239 CR taken from FASTer prediction

5.2.2 Estimation of dose rate

Here we will assume that the radionuclides are homogeneously distributed in soil. We therefore need to select dose conversion coefficients (DCCs) from the following tables of Appendix 1: Tables 2.1.2 (external dose organisms on soil; DCC_{ext_on}), 2.1.3 (external dose organisms in soil; DCC_{ext_in}), 2.1.4 (external dose plants) and 2.1.5 (internal dose; DCC_{int}) (summarized in Table 5-3). The DCC tables present alternative geometries for some reference organism types. Here we will assume the values for rabbit and fox, for herbivorous and carnivorous animals respectively. These species are burrowing and, with reference to the appropriate life-history data sheets (Appendix 2, Sections 1.1.13 and 1.1.15), rabbits spend 50 % of their time underground and foxes 10 %. Soil and detritivorous invertebrates are assumed to live solely under and above ground respectively.

Internal (D_{int}) and external (D_{ext}) dose rates (Table 5-3) for each radionuclide can then be estimated by:

D_{int} = *Activity concentration in biota (Bq kg^{-1} f.w.)* × DCC_{int}

D_{ext} = *Activity concentration in soil (Bq kg^{-1} d.w.)* ×([DCC_{ext_on} × f_{on}] + [DCC_{ext_in} × f_{in}])

where f_{on} and f_{in} are the fractions of time spent above and below ground

The total dose rate from each radionuclide (D_{tot}) can then be estimated by summing the internal and external dose rate, and the total dose rate by summing the dose rates for each radionuclide. Dose rates presented in Table 5-3 are unweighted, refer to Pröhl *et al*. (2003) for details of how to weight DCC values for RBE.

Table 5-3 DCCs and estimated unweighted dose rates for terrestrial assessment example.

	Soil invertebrate	Grass	Shrub	Detritivorous invertebrate	Carnivorous mammal	Herbivorous mammal
^{137}Cs DCC_{ext_on} (μGy h^{-1}/Bq kg^{-1})	-	-	-	$1.2x10^{-4}$	$9.5x10^{-5}$	$1.0x10^{-4}$
^{137}Cs DCC_{ext_in} (μGy h^{-1}/Bq kg^{-1})	$1.5x10^{-4}$	$1.1x10^{-4}$	$1.1x10^{-4}$	-	$5.3x10^{-5}$	$7.9x10^{-5}$
^{137}Cs DCC_{int} (μGy h^{-1}/Bq kg^{-1})	$1.4x10^{-4}$	-	-	$1.2x10^{-4}$	$2.2x10^{-4}$	$2.0x10^{-4}$
^{137}Cs D_{ext} (μGy h^{-1})	1.5	1.1	1.1	1.2	0.9	0.9
^{137}Cs D_{int} (μGy h^{-1})	$7.9x10^{-2}$	-	-	$1.0x10^{-1}$	11	3.7
^{137}Cs D_{tot} (μGy h^{-1})	1.6	1.1	1.1	1.3	12	4.6
^{239}Pu DCC_{ext_on} (μGy h^{-1}/Bq kg^{-1})	-	-	-	$5.2x10^{-8}$	$4.0x10^{-8}$	$4.5x10^{-8}$
^{239}Pu DCC_{ext_in} (μGy h^{-1}/Bq kg^{-1})	$2.0x10^{-8}$	$5.5x10^{-8}$	$2.9x10^{-8}$	-	$3.5x10^{-9}$	$5.4x10^{-9}$
^{239}Pu DCC_{int} (μGy h^{-1}/Bq kg^{-1})	$3.0x10^{-3}$	-	-	$3.0x10^{-3}$	$3.0x10^{-3}$	$3.0x10^{-3}$
^{239}Pu D_{ext} (μGy h^{-1})	$2.0x10^{-5}$	$5.5x10^{-5}$	$2.9x10^{-5}$	$5.2x10^{-5}$	$3.6x10^{-5}$	$2.5x10^{-5}$
^{239}Pu D_{int} (μGy h^{-1})	$6.5x10^{-1}$	-	-	$6.5x10^{-1}$	$4.8x10^{-7}$	$5.5x10^{-3}$
^{239}Pu D_{tot} (μGy h^{-1})	$6.5x10^{-1}$	$5.5x10^{-5}$	$2.9x10^{-5}$	$6.5x10^{-1}$	$3.7x10^{-5}$	$5.5x10^{-3}$
D_{tot} (μGy h^{-1})	2.2	1.1	1.1	2	12	4.6

5.3 Terrestrial ecosystems – example 2

In this example we conduct an assessment of the exposure of moor frogs (*R. arvalis*) living in a wetland area in the middle-east of Sweden, where the deposition of Cs-137 following the Chernobyl accident, in April 1986, was more than 100 kBq/m^2. The wetland area consists of an alder forest swampland situated in a depression mainly surrounded by coniferous forests.

Experimental data of Cs-137 (Table 5-4) reported by Stark *et al.* (2003) were used to calculate internal and external doses to frogs (Table 5-5). The average inventory of Cs-137 in the forest was about 1000 kBq/m^2. Between 86 and 99 % of the inventory was found in the top 12 cm of the soil profile and the activity concentrations followed a lognormal distribution (Figure 5-4). The values oscillated between 12.5 (5th percentile) and 65.7(95th percentile) with a geometric mean of 25.6 kBq/kg d.w. The total activity concentration in unfiltered water was 0.59 Bq/l. The average activity concentration in frogs was 1.7 kBq/kg f.w. The highest activity concentrations were measured in the smallest frogs.

Table 5-4 Experimental data of Cs-137 in different environmental media of the contaminated area.

Soil Concentration kBq/kg d.w.	Water concentrations Bq/l	Concentration in Frogs kBq/kg f.w.
12.2 (5%)	0.59 (mean)	0.5 (min)
25.6 (50%)		1.7 (mean)
65.7 (95%)		3.5 (max)

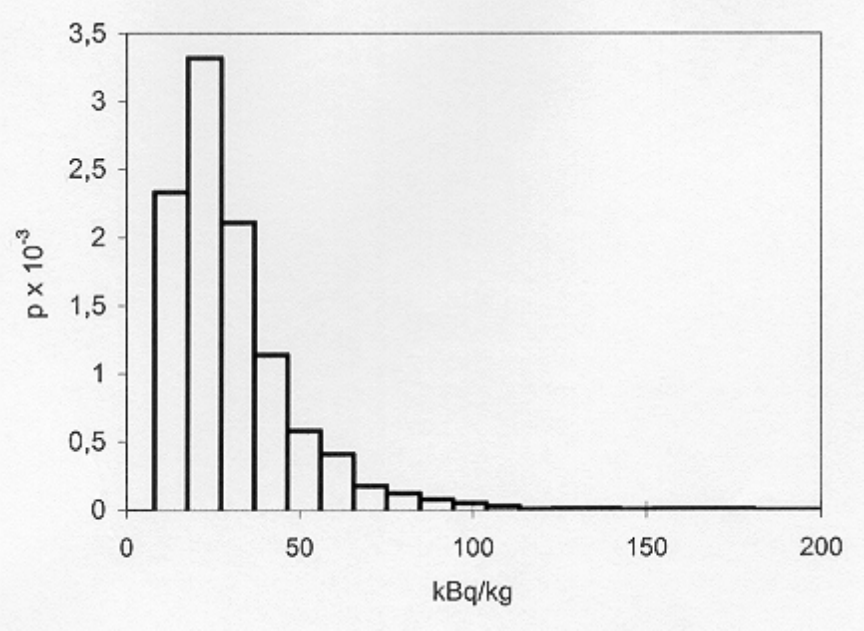

Figure 5-4 Probability distribution of Cs-137 activity concentrations in the top 12 cm of the soil in the study area.

To estimate the external dose rates it was assumed that the frogs spend 50 % of the year on the soil surface and 50 % buried in the soil. The activity concentrations in Table 5-4 were multiplied with the DCC for mouse provided in Tables 2.1.2 and 2.1.3 of Appendix 1 (the same value of 1.2E-4 µGy/h per Bq/kg is given in both tables). The internal dose rates were calculated using the DCC for mouse (1.6E-4) given in Table 2.1.5. The DCCs for mouse were chosen; because this was the reference organism which weight (35 g) was the closest to the average weight (30 g) of frogs living in the study area.

The results of the calculations are shown in Table 5-5. The total doses are dominated by the external exposure from the soil and are similar in the period when the frogs are on top of the soil (in summer) and immersed in the soil (in winter).

Table 5-5 External and internal dose rates to frogs calculated from the values of activity concentration shown in Table 5-4 using the DCCs recommended in Appendix 1.

External dose rate mGy/year	Internal dose rates mGy/year
1.3E+01 (5%)	7.0E-01 (min)
2.7E+01 (50%)	2.4E+00 (mean)
6.9E+01 (95%)	4.9E+00 (max)

6 Concluding remarks

The methodology presented within this report allows an assessor to derive weighted or unweighted absorbed dose rates to a suite of reference organisms following a release of radionuclides to the environment. At this stage, the approach has been developed for radioisotopes of 20 elements and 7 ecosystems starting from a point where data pertaining to activity concentrations (in units of Bq kg^{-1} and Bq l^{-1} for terrestrial systems and aquatic, respectively) or depositions (in units of Bq m^{-2} for terrestrial systems only) are available.

The methodology is based on the application of transfer factors, derived from literature review and modelling work, and dose conversion coefficients, derived earlier in FASSET Deliverable 3 (Pröhl *et al*., 2003). The coverage of transfer factor data for the radionuclides considered varies greatly between ecosystems. Whereas comprehensive coverage has been attained for agricultural ecosystem and marine systems, albeit with low confidence in some cases, more limited coverage has been reported for freshwater and semi-natural systems. For yet other ecosystems, as exemplified by wetlands, very little information is available on transfer of radionuclides to reference flora and fauna. Some methods have been explored for the purpose of filling gaps related to transfer, notably the implementation of biokinetic models parameterised using allometrically-derived values. Furthermore, the validity of applying transfer factors to non-equilibrium situations has been questioned leading to the view that dynamic models, formulated from a mechanistic understanding of the processes involved, would be preferable. Life history data have been collated for specific examples of reference organisms with a view to provision of information that may be useful in the process of conducting detailed impact assessments. For the generic methodology presented in FASSET, however, only information in relation to (i) the biota's body sizes (to construct typical ellipsoid gemotries in the derivation of DCCs and in some cases in the process of model parameterisation using allometric relationships) and (ii) occupancy in selected habitats has been used. Further work may be required to establish whether the simplifying assumptions adopted provide a reasonable estimate of dose rate compared to more detailed analyses considering exposure assessments for multiple life stages, differential uptake of radionuclides within the body of the organism etc. In order to address uncertainties in a preliminary way, some guidance is given in this report. The application of such methods may allow the identification of components in the assessment where uncertainty is greatest and facilitate the allocation of resources to areas of study (though experiment, further modelling etc.) that will reduce overall uncertainty in the most effective manner.

The examples of application provided in this report have shown, as expected, that the absolute value of the dose rate and relative importance of internal and external irradiation is influenced to a large degree by radionuclide-specific parameters and habitat. Although the exposure assessment methodology has been tested in a fairly simple way in Section 5, further work is required to explore the limitations of the approach, possibly through its application within comprehensive case studies.

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