Impact Assessment of Ionising Radiation on Wildlife

R&D Publication 128




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The Environment Agency R&D Publication 128: "Impact assessment of ionising radiation in wildlife" was produced in June 2001, under R&D Project P3-085. At the time the emphasis was to look at impacts from nuclear sites discharges. This resulted in the choice of specific radionuclides to derive doses and dose rates to wildlife.

The R&D Publication 128 has already been used widely by Agency inspectors as well as English Nature staff. Furthermore, its methodology has been incorporated into the Environment Agency's functional guidance on applying the Habitats Regulations to Radioactive Substances Authorisations. As part of the R&D Publication 128, the three spreadsheet programmes provided dosimetric and concentration ratios for nine radionuclides:

- estuarine/freshwater ecosystems: ^{3}H, ^{14}C, ^{99}Tc, ^{90}Sr, ^{137}Cs, $^{239+240}$Pu, ^{238}U, ^{129}I, ^{210}Po;
- terrestrial ecosystem: ^{3}H, ^{14}C, ^{35}S, ^{90}Sr, ^{137}Cs, $^{239+240}$Pu, ^{238}U, ^{129}I, ^{226}Ra).

The requirement to extent the list of radionuclides to include ^{60}Co, ^{106}Ru, ^{131}I, ^{125}I, ^{234}Th, ^{234m}Pa, ^{241}Am, and ^{32}P came from Agency inspectors and National Compliance and Assessment Service, who wish to extend the methodology for a range of nuclear and non-nuclear discharges. The objectives for the project were:

- to provide information on concentration factors for additional radionuclides (^{60}Co, ^{106}Ru, ^{131}I, ^{234}Th, ^{234m}Pa, ^{241}Am, ^{32}P);
- to include these additional radionuclides in the three assessment spreadsheets (freshwater, marine, and terrestrial); we have also been asked to provide information and update the spreadsheets for ^{125}I for the freshwater and marine spreadsheets; and
- to provide quality assurance (QA) data to backup the information provided in the spreadsheets and the CF data.

In order to calculate dose rates to organisms for those specific radionuclides, literature searches were carried out to find concentration factors (CF) from the environment to the organisms being modelled. Radionuclide transfer data were provided by ERC, Liverpool University, and dosimetric calculations and spreadsheet programming were sub-contracted to Westlakes Scientific Consulting.

The main problem has been with the identification of suitable data on ^{32}P, as few data have been found. The QA exercise proved successful in identifying a limited numbers of erroneous CF values, and in adding newly found CF values to the original spreadsheets. Queries related to the reference sources can be directed to Dr David Copplestone, at ERC.

The new spreadsheet versions provide extended radionuclide coverage relative to version 1.0, but underlying calculation methods and the user interface are otherwise unchanged. The spreadsheets employ protection to prevent unauthorised modification of code or worksheets. French encryption standards for passwords differ from those of the rest of Europe and the USA. The spreadsheet files on the new CD will work both in France (i.e. with Windows regional settings 'French - standard') and elsewhere. The user is still reminded to use the CD in conjunction with the R&D Publication 128.

Impact Assessment of Ionising Radiation on Wildlife

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Statement of Use

The report is designed as an introduction to the process of impact assessment of ionising radiation on wildlife, reviewing current and proposed approaches resulting from legislation on environmental protection. The report provides recommendations for practitioners who carry out impact assessment of ionising radiation on wildlife in England and Wales, with particular reference to the Habitats Regulations, subject to stated assumptions and constrained to the selected radionuclides. Other parties wishing to carry out such assessments can also make use of the approach outlined.

Keywords

Wildlife; Ionising Radiation; Radioactivity; Effects; Protection; Environment; Radioactive discharges; Radionuclides

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Foreword

There is a requirement to assess the impacts of consents and authorisations affecting Natura 2000 sites under the Habitats Regulations (1994), including the assessment of radiological risk.

The International Commission on Radiological Protection (ICRP) has traditionally presumed that "if man is adequately protected from ionising radiation, then so are other species". This assumption is being increasingly challenged due both to the lack of cited evidence to support the ICRP position, and the inconsistency with situations where the precautionary approach has been adopted to protect the environment from non-radioactive discharges. Increasing public and political pressure to introduce commitments relating to environmental protection is also evident in international conventions. There is now increasing recognition that the environment should be protected in its own right.

This R&D project was commissioned by the Environment Agency and English Nature in January 2001 to provide up-to-date information on ionising radiation impacts on wildlife, upon which a robust assessment approach may be developed. The report provides:

- a review of the latest research on ionising radiation effects on plants, animals and ecosystems since the Nature Conservancy Council report (Kennedy *et al.*, 1990);
- an outline and review of the relevant European and national legislation which has impacts on the requirements for assessments of the impact of ionising radiation on wildlife in the UK;
- a brief review of the role of regulatory bodies in assessing the impact of ionising radiation on wildlife in England and Wales;
- recommendations on the relative biological effectiveness of different types of radiation with respect to different classes of wildlife;
- an assessment of the scale of the impact from the effects of ionising radiation on wildlife; and
- to recommend an approach to assess the impacts to wildlife from ionising radiation from authorised discharges in England and Wales, with spreadsheets to support the methodology.

A European Commission funded project (Framework for ASSessment of Environmental impacT - FASSET) started in November 2000, and is expected to deliver a harmonised framework for adoption within the EU for future radiation assessments by October 2003. However, the need for interim measures to assess the impact of ionising radiation on wildlife has been recognised and this report sets out an approach which may be adopted in England and Wales.

Acknowledgements

This project was funded jointly by the Environment Agency and English Nature to provide guidance on the assessment of the impact of ionising radiation on wildlife under the UK regulations devised under the Habitat and Bird Directives.

The authors would like to acknowledge the input of the project board members: Pam Nolan (National Habitats Directive Project Manager), Andy Mayall (PIR/RSR Inspector), Clive Williams (Radioactive Substances Regulation Policy Development Manager) and Irene Gize (Environment Protection Scientist) from the Environment Agency and Alastair Burn (Senior Pollution Advisor) and Jill Sutcliffe (Botanical Manager) from English Nature.

The authors also acknowledge Dennis Woodhead from CEFAS for the peer review of this report.

Executive Summary

This R&D project was commissioned by the Environment Agency and English Nature in January 2001 to provide up-to-date information on the impacts of ionising radiation on wildlife, upon which a robust assessment approach may be developed. This approach will also feed into the European Commission funded project 'Framework for Assessment of Environmental Impact' (FASSET), due to complete in October 2003.

This report describes the behaviour and transport of radionuclides in the environment, considers the impact of ionising radiation on wildlife, and makes recommendations on an approach for the impact assessment of ionising radiation on wildlife for England and Wales. The assessment approach focuses on three ecosystems representative of those considered potentially most at risk from the impact of authorised radioactive discharges, namely a coastal grassland (terrestrial ecosystem); estuarine and freshwater ecosystems. The likely scale of the impact on wildlife is also assessed in light of a preliminary analysis based on this assessment approach.

The aims of the report are:

- to summarise the latest research on the behaviour, transfer and impact of ionising radiation effects on wildlife;
- an outline and review of the relevant European and national legislation which has impacts on the requirements for assessments of the impact of ionising radiation on wildlife in the UK;
- to consider the role of regulatory bodies in assessing the impact of ionising radiation on wildlife with respect to England and Wales;
- to make recommendations on the relative biological effectiveness of different types of radiation with respect to wildlife; and
- to recommend an approach to assess the impacts to wildlife from ionising radiation from authorised discharges in England and Wales, with spreadsheets to support the methodology.

The report demonstrates the behaviour and transfer of radionuclides in a number of different ecosystem types. Particular emphasis is placed on exposure pathways in those ecosystems most likely to be impacted by the authorised discharges of radioactivity within England and Wales.

As there is no international consensus on the approach to be taken to assess the impact of ionising radiation on wildlife, some countries have adopted their own legislation. The report evaluates these regulatory frameworks and describes the current UK position. Information reviewed (Woodhead, 1998), indicates that it is unlikely that there will be any significant effects in:

- terrestrial animal populations at chronic dose rates below 40 ì Gyh^{-1};
- terrestrial plant populations at chronic dose rates below 400 ì Gyh^{-1};
- populations of freshwater and coastal organisms at chronic dose rates below 400 ì Gyh^{-1}; and
- populations of organisms in the deep ocean at chronic dose rates below 1,000 ì Gyh^{-1}.

The Environment Agency uses these dose limits to biota when following its current assessment approach to determine the likely impact of exposure to ionising radiation from authorised discharges.

The impact assessment approach described in this report further develops the existing EA approach to provide a generic assessment. It is therefore important to recognise that the assessor must consider site specific features such as the presence of rare species when using generic guidelines given in this report to evaluate the impact of ionising radiation on wildlife. In such instances generic guidelines should be used with caution and possible re-evaluation of the guideline dose limits recommended within this report may be required.

Evidence for effects at low dose rates is reviewed and Tables of experimental and field study data on the effects of ionising radiation are presented, with which to compare any predicted doses to wildlife in order to assist in the impact assessment process.

The use of biomarker techniques is reviewed, and their application to the study of exposure to multiple contaminants is discussed. The application of biomarkers to the study of wildlife is, relatively speaking, still in its infancy but the possible approaches are discussed. Further development of biomarker techniques is required; in particular research is needed into the consequences of any observed biological damage for the health of the exposed individual or population is needed.

Impact Assessment Methodology

The assessment of radiation doses to wildlife is not easy, and there is no equivalent system to that used for humans (based on the ICRP biokinetic model). Although a number of countries have adopted more stringent options or are currently including the implementation of dose limits to the environment (e.g. USA). A simple approach has been adopted in this report based on the latest thinking in the field (e.g. NCRP, 1991; Woodhead, 2000a). The basis of the approach is the calculation of doses to wildlife determined by their size, dietary uptake of radionuclides and external exposure in the environment. The doses may be calculated using literature derived values or from empirical measurements of radionuclide concentrations. This can be used as an interim means of assessing impact until the FASSET recommendations become available in October 2003. The essential steps in these calculations include:

- Each organism is represented as an ellipsoid, so that the fraction of decay energy emitted within the organism can be calculated;
- Selection of organism based on their radioecological significance and radiosensitivity, and endpoints of importance (e.g. morbidity, mortality, reproductive capacity, mutation rate).
- Data from the above are used to evaluate a Dose Per Unit Concentration (DPUC) for each radionuclide;
- The average dose throughout the volume of the organism is calculated, for both internal and external contamination;
- Assessment of dose to each organism is determined using concentration factors (internal dose) and positioning relative to soil/sediment or water (external dose).

Various data are required to enable dose calculations:

- Concentrations of each radionuclide in the soil/sediment, water and air;
- Concentration factors for each radionuclide in each organism to be assessed relative to soil, water or air;
- Organism dimensions in order to determine the size of the ellipsoid;
- The proportion of time the organism spends in different ‘compartments’ of the ecosystem.

Several radionuclides have been selected for the impact assessment in order to investigate the feasibility of the approach. These radionuclides were identified in consultation to determine those with a potentially high radiobiological significance to wildlife.

- Estuarine and freshwater ecosystems: ^{3}H, ^{14}C, ^{99}Tc, ^{90}Sr, ^{137}Cs, $^{239+240}Pu$, ^{238}U, ^{129}I, ^{210}Po
- Terrestrial ecosystem: ^{3}H, ^{14}C, ^{35}S, ^{90}Sr, ^{137}Cs, $^{239+240}Pu$, ^{238}U, ^{129}I, ^{226}Ra.

A wide range of species have been chosen as representative of target species of likely significance:

- For freshwater ecosystem: bacteria, macrophyte, phytoplankton, zooplankton, benthic mollusc, small benthic crustacean, large benthic crustacean, pelagic fish, benthic fish, amphibian, duck, aquatic mammal.

- For estuarine/marine ecosystem: bacteria, macrophyte, phytoplankton, zooplankton, benthic mollusc, small benthic crustacean, large benthic crustacean, pelagic fish, benthic fish, fish egg, seabird, seal, whale.
- For terrestrial ecosystem: bacteria, lichen, tree, shrub, herb, seed, fungus, caterpillar, ant, bee, wood louse, earthworm, herbivorous mammal, carnivorous mammal, rodent, bird, bird egg, reptile.

An extensive literature review was undertaken to provide data for the dose assessment calculations. The review aimed to identify concentration factors for the selected species and radionuclides. As large gaps in the available data on concentration factors were found, particularly for the terrestrial ecosystem, simplifications were required in the assessment process. Differences in the behaviour of ^{3}H, ^{14}C and ^{35}S in the terrestrial environment have also led to additional complications in the dose calculations. These have been overcome either by using an isotopic abundance approach (for ^{3}H and ^{14}C) or by simplifying the approach (e.g. for ^{35}S).

The dose calculations have been programmed into Excel spreadsheets using Visual Basic for Applications. The three spreadsheets (coastal, freshwater and terrestrial) are available on a CD, attached to the report, with instructions on their use.

The spreadsheets can be used to calculate doses to wildlife, which can be used generically or specifically to particular sites. The spreadsheets can be manipulated by the user who can use default values (from the literature review), modelled or measured concentrations in the different species or ecosystem compartments (e.g. soil, water ect.). The doses obtained can be compared with guideline values given above to assess the scale of the impact on wildlife subject to a number of caveats, for example, the need to protect rare or endangered species, and limitations/assumptions in the approach which are detailed in the report. In such cases, comparison of the doses with the effects Tables provided would give an indication of whether biological damage may occur.

Scenarios of radioactive contamination with which to undertake impact assessments have been provided. These are based on measured values in ecosystem components from around UK nuclear sites. The scenarios are not specific to any one site. The scenarios examine the consequences of exposure to ionising radiation that may occur around any one site. In this way, a 'worst case' for wildlife exposure to ionising radiation can be produced.

Using the assessment scenarios and information from the literature, it can be concluded that wildlife in England and Wales are not significantly impacted by exposure to ionising radiation from authorised discharges (subject to the limitations/assumptions detailed in the report), although there are specific areas which need to be investigated further. These include (but are not limited to):

- radiation exposure in long lived animals (particularly marine species);
- site specific features for particular radionuclide discharges;
- other radionuclides not included in this assessment process;
- biological consequences of exposure to multiple contaminants present at a site (interaction between radioactive and non-radioactive contaminants).

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Main Glossary

The following have been adopted or modified from; IAEA Safety Glossary (2000), NRPB (1998), and Warner & Harrison (1993).

Aberration Departure from normal.

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

Actinides A group of 15 elements with atomic number from that of actinum (89) to lawrencium (103) inclusive. All are *radioactive.*

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

Acute exposure Exposure received within a short period of time. Normally used to refer to exposure of sufficiently short duration that the resulting *dose* can be treated as instantaneous (e.g. less than an hour). Usually contrasted with *chronic* and *transitory exposure.*

Advanced Gas Cooled Reactor A development of the *Magnox reactor,* using *enriched uranium* oxide fuel in stainless steel cladding.

Adsorb Usually a solid holding molecules (of a gas or liquid etc.) to its surface, forming a thin film.

Alpha particle A particle consisting of two *protons* plus two *neutrons*. Emitted by a *radionuclide.*

Apoptosis Apoptosis or programmed cell death occurs naturally during the development and maintenance of animal tissues and organs. During these processes more cells are produced than are required for building tissues and organs. The unwanted cells are programmed to die either because the chemical signals that direct them to go on living are suppressed or because they receive a specific signal to die.

Atom The smallest portion of an *element* that can combine chemically with other atoms.

Authorisation The granting by a *regulatory body* or other governmental body of written permission for an operator to perform specified activities.

Background The *dose* or *dose rate* (or an observed measure related to the *dose* or *dose rate*), attributable to all sources other than the one(s) specified.

Becquerel (Bq) See *activity.*

Benthic invertebrate Aquatic invertebrate living on or in the sediment.

Benthos Synonym for community of *benthic invertebrate.*

Beta particle An *electron* emitted by the *nucleus* of a *radionuclide.* The electric charge may be positive, in which case the beta particle is called a positron.

Biomarker A biological response to an environmental pollutant which gives a measure of exposure. The response may be molecular, cellular or whole organism.

Chromatid When a *chromosome* becomes shorter and thicker during the first stage of *mitosis* it is seen to become a double thread. Each thread is a *chromatid.*

Chromosome translocation Sporadic and random fusion of part of one *chromosome* onto part of another.

Chromosomes	Rod-shaped bodies found in the nucleus of cells in the body. They contain the *genes*, or hereditary constituents. Each chromosome has a characteristic length and banding pattern.
Chronic exposure	Exposure persisting in time. Normally used to refer to continuous exposures to low concentrations of pollutants. See also t*ransitory* and a*cute exposure*.
Concentration Factor	Ratio of element or nuclide in the consumer (or a specific tissue or organ ect.), to that in what is consumed, or to that in the environmental medium.
Cosmic Rays	High energy *ionising radiations* from outer space.
Critical Group	Sub-group of the public most affected by a given release of radioactivity.
Cytogenetic damage	Damage to chromosomes that can be detected on the microscopic level. Examples of damage include deletions, translocations and micronuclei.
Decay	The process of spontaneous transformation of a *radionuclide*. The decrease in the activity of a *radioactive* substance.
Decay product	A nuclide or *radionuclide* produced by *decay*. It may be formed directly from a *radionuclide* or as a result of a series of successive decays through several *radionuclides.*
Decommissioning	The process of closing down a *nuclear reactor*, removing the spent fuel, dismantling some of the other components, and preparing them for *disposal.* Term may also be applied to other major nuclear facilities.
Deterministic effect	A *radiation* effect for which generally a threshold level of *dose* exists above which the severity of the effect is greater for a higher *dose.*
Disposal	In relation to *radioactive waste*, dispersal or emplacement in any medium without the intention of retrieval.
DNA	*Deoxyribonucleic Acid (DNA).* The compound that controls the structure and function of cells and is the material of inheritance.
Dose	General term for quantity of *ionising radiation.* See *absorbed dose*, *equivalent dose* and *effective dose*. Frequently used for *effective dose*.
Dose assessment	Assessment of the *dose(s)* to an individual or group of people.
Dose rate	Dose released over a specified unit of time.
Effective dose	The quantity obtained by multiplying the *equivalent dose* to various tissues and organs by a weighting factor appropriate to each and summing the products. Unit *sievert*, symbol Sv. Frequently abbreviated to *dose*.
Electron	An elementary particle with low mass, $^{1}/_{1836}$ that of a *proton,* and unit negative electric charge. Positively charged *electron*s, called positrons, also exist. See also *beta particle*.
Electron Volt	Unit of energy employed in *radiation* physics. Equal to the energy gained by an *electron* in passing through a potential difference of 1 volt. Symbol eV. 1eV = 1.6×10^{-19} joule approximately.
Embryo (in animals)	The stage of development between the time that the fertilised egg begins to divide and the developing animal hatches or is born.
Embryo (in plants)	The part of a seed which develops into the root (radicle) and shoot (plumule) of a plant.
Embryogenesis	The processes leading to the development of an *embryo.*

End point	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.
Enriched Uranium	*Uranium* in which the content of the *isotope* uranium-235 has been increased above its natural value of 0.7% by weight.
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 *ionising radiations* in causing harm to tissue. Unit sievert, symbol Sv.
Fallout	The transfer of *radionuclides* produced by nuclear weapons from the atmosphere to earth; the material transferred.
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.
Fission	Nuclear fission. A process in which a *nucleus* splits into two or more nuclei and energy is released. Frequently refers to the splitting of a nucleus of uranium-235 into two approximately equal parts by a thermal neutron with emission of other neutrons.
Fission products	Nuclides or *radionuclides* produced as a result of *fission.*
Foetus	The developing *embryo* is known as a foetus once it can be recognised as a species.
Free radical	A grouping of *atoms* that normally exists in combination with other atoms but can sometimes exist independently. Generally very reactive in a chemical sense.
Gametes	The sex cells which fuse together at fertilisation to form the zygote. In animals the gametes are the sperm in males and the ovum (egg) in females. In plants the gametes are the pollen in the male and the ovules in the female.
Gametogenesis	Process leading to the production of gametes.
Gamma ray	A discrete quantity of electromagnetic energy without mass or charge. Emitted by a *radionuclide.*
Genes	The biological units of heredity. They are arranged along the length of *chromosomes.*
Genotoxicity	Ability to cause damage to genetic material. Such damage may be mutagenic and/or carcinogenic.
Germ cell	Cell specialised to produce *gametes*. The germ cell line is often formed very early in embryonic development.
Gestation	The process of being carried in the womb, from conception to birth.
Gray (Gy)	See absorbed dose.
Half-life	The time taken for the *activity* of a *radionuclide* to lose half its value by *decay.* Symbol *t½.*

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.
Implantation	When an *embryo* passes from the oviduct to the uterus it becomes attached to the uterine wall.
Indicator Species	A species that only thrives under certain environmental conditions and whose presence shows that these conditions are present.
Ion	Electrically charged *atom* or grouping of atoms.
Ionisation	The process by which a neutral *atom* or *molecule* acquires or loses an electric charge. The production of *ions*.
Ionising radiation	*Radiation* that produces *ionisation* in matter. Examples are *alpha particles, gamma rays, X-rays* and *neutrons*. When these radiations pass through the tissues of the body, they have sufficient energy to damage *DNA*.
Isotope	*Nuclides* with the same number of *protons* but different numbers of *neutrons*. Not a synonym for nuclide.
Karyotype	The complete set of *chromosomes* of a cell or organism.
LD_{50}	The dose that causes mortality in 50% of the organisms tested.
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.
Low and intermediate level waste (LLW & ILW)	*Radioactive waste* with radiological characteristics between those of exempt waste and high level waste. These may be long-lived waste (LILW-LL) or short-lived waste (LILW-SL).
Magnox reactor	A thermal reactor named after the magnesium alloy in which the uranium metal fuel is contained. The *moderator* is graphite and the coolant is carbon dioxide gas.
Meiosis	A form of nuclear division in which each daughter cell receives only one of each homologous chromosome pair. Meiosis occurs during the formation of gametes.
Mitosis	A type of cell division by which two daughter cells are produced from one parent cell, with no change in the number of chromosomes.
Moderator	A material used in *nuclear reactors* to reduce the energy and speed of the *neutrons* produced as a result of *fission*.
Molecule	The smallest portion of a substance that can exist by itself and retain the properties of the substance.
Morbidity	The state of being diseased.
Morphogenesis	The process of "shape formation": the processes that are responsible for producing the complex shapes of adults from the simple ball of cells that derives from division of the fertilised egg.

Mutation	A change in the genetic material of an organism. This can be spontaneous or induced by chemicals or radiation.
Naturally occurring radionuclides	*Radionuclides* that occur naturally in significant quantities on Earth.
Neutron	An elementary particle with unit atomic mass approximately and no electric charge.
Non-ionising radiation	*Radiation* that does not produce ionisation in matter. Examples are ultraviolet radiation, light, infrared radiation and radiofrequency radiation. When these radiations pass through the tissues of the body they do not have sufficient energy to damage *DNA* directly.
Non-nuclear licensed site	A non-nuclear licensed site (or non nuclear site) is where the handling, use and discharge of radioactive substances may occur but not as the main activity. This includes research institutions, hospitals, defence establishments etc.
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.
Nuclear Licensed site	A nuclear licensed site (or nuclear site) holds an operating licence under the Nuclear Installations Act (1965) where the handling or use of radioactive materials is the main activity
Nuclear Power	Power obtained from the operation of a *nuclear reactor*.
Nuclear Reactor	A device in which nuclear *fission* can be sustained in a self supporting chain reaction involving *neutrons*. In thermal reactors, fission is brought about by thermal neutrons.
Nuclear Weapon	Explosive device deriving its power from *fission* or fusion of nuclei or from both.
Nucleus (of atom)	The core of an *atom*, occupying little of the volume, containing most of the mass, and bearing positive electric charge.
Nucleus (of cell)	The central part of a cell containing *chromosomes* and the genetic information bound in *DNA*.
Oocyte	The developing female gamete before maturation and release.
Organogenesis	The process of formation of specific organs in a plant or animal involving morphogenesis and differentiation.
Pelagic biota	Aquatic organisms living in the water column of a body of water, rather than along the shore or in the bottom sediments.
Pressurised Water Reactor (PWR)	A thermal reactor using water as both a *moderator* and coolant. Uses *enriched uranium* oxide fuel.
Proton	An elementary particle with a mass of $1.672\ 614 \times 10^{-27}$ kg and unit positive electric charge.
Radiation	The process of emitting energy as waves or particles. The energy thus radiated. Frequently used for *ionising radiation* except when it is necessary to avoid confusion with *non-ionising radiation.*
Radiation Weighting Factor (w_r)	w_r values (radiation weighting factors) represent the relative biological effectiveness of the different radiation types, relative to X- or ã-rays, in producing endpoints of ecological significance

Radioactive Waste	Useless material containing *radionuclides.* Frequently categorised in the nuclear power industry according to activity and other criteria, as *low level, intermediate level,* and *high level waste.*
Radiobiology	The study of the effects of *ionising radiation* on living things.
Radiological protection	The science and practice of limiting the harm to human beings from *radiation.*
Radionuclide	An unstable nuclide that emits *ionising radiation.*
Regulatory Body	An authority or a system of authorities designated by the government of a State as having legal authority for conducting the regulatory process, including issuing authorisations and thereby regulating nuclear, *radiation*, *radioactive waste* and transport safety.
Relative Biological Effectiveness (RBE)	A relative measure of the effectiveness of different *radiation* types at inducing a specified health effect, expressed as the inverse ratio of the *absorbed doses* of two different *radiation* types that would produce the same degree of a defined biological endpoint.
Reprocessing	A process or operation, the purpose of which is to extract radioactive isotopes from *spent fuel* for further use.
Risk	A measure of the probability and extent of harm.
Sievert	See *effective dose.*
Spent fuel	Nuclear fuel removed from a reactor following irradiation, which is no longer useable in its present form because of depletion of fissile material, poison build-up or *radiation* damage.
Spermatocytes	Cells of the male reproductive system.
Stem cell	A cell that upon division, produces dissimilar daughters, one replacing the original stem cell, the other differentiating further (e.g. meristems of plants).
Stochastic effect	A *radiation*-induced health effect, the probability of occurrence of which is greater for a higher *radiation dose* and the severity of, which (if it occurs) is independent of *dose*.
Telomere	The end of a *chromosome*.
Transitory Exposure	Exposure that is too protracted to be described as *acute exposure*, but does not persist for many years, is sometimes described as *transitory exposure.*
X-ray	A discrete quantity of electromagnetic energy without mass or charge. Emitted by an X-ray machine.

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1. Introduction

1.1 Aims and scope of the report

The procedures for the protection of humans from ionising radiation are well developed, with a system in place to limit the effects on individuals based on recommendations from the International Commission on Radiological Protection (ICRP). At present, an internationally accepted method for assessing the environmental impact of ionising radiation does not exist and up to now the approach taken has relied on recommendations from the ICRP first made in 1977, and modified in 1990. The ICRP states that the standard of environmental control needed to protect humans will ensure that other species will not be put at risk (ICRP, 1991).

This statement is being increasingly challenged, in part due to:

- the lack of cited evidence to support the ICRP position (Thompson, 1988);
- because the approach does not demonstrate adequate protection for habitats with little or no human habitation;
- lack of protection in habitats where biota could be exposed to harmful doses whilst human exposure is below the recommended dose limits (Pentreath, 1998).

An example of the latter would be deep-sea disposal of radioactive waste, which, although no longer practised, has been shown to potentially give rise to very high doses to benthic fauna (International Atomic Energy Agency, 1988a). The ICRP approach to protection of the environment is also inconsistent with the more precautionary approach adopted to protect the environment from non-radioactive discharges. Ideally, an integrated approach that enables assessment of the total environmental impact of a site discharging both radioactive and non-radioactive discharges is required, including considerations on the interactions between different pollutants.

Increasingly, there is now a general recognition that the environment should be protected in its own right from the effects of pollution. The Rio Declaration (UN, 1993) addressed environmental protection stating that:

> ''*in order to achieve sustainable development environmental protection shall constitute an integral part of the development process and cannot be considered in isolation from it*''.

The commitment towards sustainable development that arose from United Nations Conference on Environment and Development (UN, 1993) and increased public awareness of environmental issues has led to pressure to define more fully the impact of human activities on the environment. The 'environment' being defined as all biota (including humans), and the interactions with their physical surroundings (IAEA, 2000). It has now been agreed that the objectives of environmental protection from the effects of ionising radiation should be to minimise unnecessary impacts and maintain biodiversity and ecosystem functioning (IAEA, 2000) with 'harm' defined as any response which contradicts these aims.

The development of a framework to provide criteria and an approach for the protection of the environment from ionising radiation has been proposed (Pentreath, 1999; Strand *et al.*, 2000). This has culminated in a European Commission funded project 'Framework for Assessment of Environmental Impact' (FASSET) which started in November 2000 and is due to complete in October 2003. The Environment Agency[1] (EA) and English Nature[2] (EN) have recognised the need for an

[1] The Environment Agency have a statutory duty to protect the environment, including authorising radioactive discharges.

[2] EN – English Nature is responsible for designating and monitoring the conservation status of SSSIs and is a statutory consultee for radioactive substances regulation within the UK. Through a Joint Nature Conservation Committee (JNCC) lead agency arrangement, EN also represents the interests of the Countryside Council of Wales in respect of the latter.

interim approach to the assessment of the impact of ionising radiation on the environment pending the outcome of the FASSET project.

This report has been commissioned to review both the current knowledge on the exposure and the effects of ionising radiation on wildlife and the approaches to the protection of the environment from ionising radiation being adopted internationally as a baseline upon which to develop recommendations for assessments in England and Wales. The aims of this report are to:

- review the latest relevant research on the behaviour of radionuclides in the environment, with particular emphasis on the transfer pathways to wildlife;
- review the latest relevant research on ionising radiation effects on plants, animals and ecosystems;
- an outline and review of the relevant European and national legislation which has impacts on the requirements for assessments of the impact of ionising radiation on wildlife in the UK;
- review the role of regulatory bodies in assessing the impact of ionising radiation on wildlife with respect to England and Wales;
- review the international approaches being adopted to assess the impact of ionising radiation on wildlife;
- make recommendations on the relative biological effectiveness of different types of radiation with respect to different classes of organisms;
- recommend an approach to assess the impacts to wildlife from ionising radiation from authorised discharges in England and Wales, with spreadsheets to support the methodology; and
- make recommendations for an approach to the protection of the environment from ionising radiation from authorised discharges (prior to the FASSET recommendations in October 2003).

To achieve this, the report considers the research carried out since the report for Nature Conservancy Council "Radioactivity and Wildlife" (Kennedy *et al.*, 1990) and incorporates the following major reviews on environmental protection from ionising radiation:

- NCRP (1991) *Effects of Ionising Radiation on Aquatic Organisms*. NCRP Report No. 109. NCRP, Bethesda.
- IAEA (1992) *Effects of Ionising Radiation on Plants and Animals at Levels implied by Current Radiation Protection Standards.* Technical Report Series No. 332. IAEA, Vienna.
- UNSCEAR (1996) Effects of radiation on the environment. In: *Sources and Effects of Ionising Radiation.* United Nations Scientific Committee on the Effects of Atomic Radiation. UNSCEAR 1996 Report to the General Assembly, with Scientific Annex. United Nations, New York.
- Woodhead D (1998) *The Impact of Radioactive Discharges on Native British Wildlife and the Implications for Environmental Protection.* Environment Agency R&D Technical Report P135.
- Woodhead D (2000a) Environmental Dosimetry: the Current Position and the Implications for Developing a Framework for Environmental Protection. Environment Agency R&D Technical Report P350.

1.2 Report structure

This report provides a basic review of dosimetry and considers the sources of ionising radiation in the environment (Chapter 1) before reviewing current knowledge of the routes of exposure of wildlife to ionising radiation (Chapter 2). Chapter 3 reviews current literature on the effects of ionising radiation

on wildlife. Chapter 4 puts the effects in the context of legislation and considers the approach taken to environmental protection from ionising radiation by different countries. Chapter 5 describes the dosimetric method for calculating doses to wildlife based on best available information and Chapter 6 describes the impact assessment approach. The flow chart (Figure 1.1) illustrates how these Sections inter-relate in the impact assessment approach.

1.3 Units in radiation protection

Unstable forms of naturally occurring and anthropogenic elements are known as radioisotopes. To reach stability these radioisotopes release energy mainly in the form of α particles, β particles and γ rays, during a process known as radioactive decay. Each type of radiation has differing capability to penetrate biological tissues and other substances. The radioactive characteristics of each radionuclide are dependent upon the type of radiation emitted, the energy of that radiation, and the radionuclide's half-life. Tables 1.1 and 1.2 describe the units commonly used when dealing with radioactivity and some of the characteristics of different radiation types. A more detailed review on the properties of ionising radiation is provided by the NRPB (1998) and Martin and Harbison (1996).

Radiation dosimetry is the process of determining the quantity of energy absorbed by a defined target from the ambient radiation field. There are two fundamental quantities used in radiation dosimetry: the '**absorbed dose**' and the '**dose equivalent**' (Table 1.1). The amount of radiation absorbed by the body is expressed in terms of the energy deposited in the tissues - the absorbed dose - and is measured in Grays (Gy). In this report, total doses will be expressed in Gy and dose rates in $\mu Gy\ h^{-1}$.

Ionising radiations differ in the way in which they interact with biological tissues, so that equal absorbed doses (meaning equal amounts of energy deposited) do not necessarily have equal biological effects. For example, 1 Gy to tissue from á radiation is more harmful than 1 Gy from â or ã radiation. This is because an á particle, being slower and more heavily charged, loses its energy over a much shorter distance along its path (NRPB, 1998). This loss of energy over a path is termed linear energy transfer (LET) and the á particles are said to have a high LET.

Another quantity must be used to assess the biological effects of ionising radiations from different sources on an equal basis. This is the equivalent dose, expressed in a unit called the sievert (Sv) and calculated by the application of a radiation-weighting factor. At present no internationally agreed radiation weighting factors have been determined for wildlife species. Chapters 5 and 6 discuss the significance of this in more detail.

Table 1.1 Units commonly used when dealing with radioactivity (Kennedy et al., 1990)

Unit	*Symbol*	*Measure of*	*Characteristics*
Becquerel	Bq	Radioactivity	1 disintegration per second
	TBq		10^{12} disintegrations per second
	PBq		10^{15} disintegrations per second
Gray	Gy	Absorbed dose	A dose of 1 Gy deposits 1 Joule of energy per kilogram
Sievert	Sv	Dose equivalent	The absorbed dose in Grays multiplied by the radiation weighting factor.

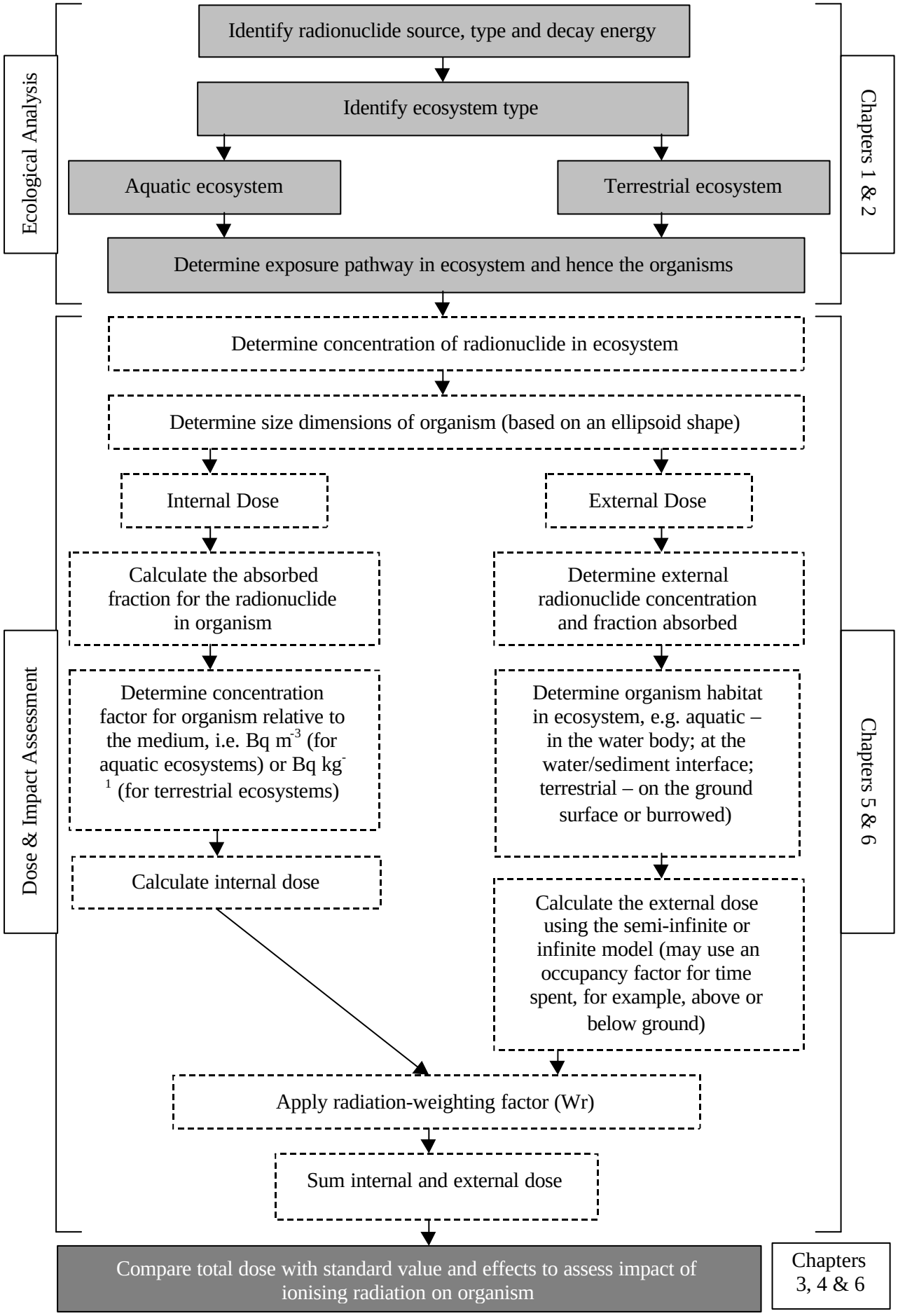

Figure 1.1 Flow diagram of the impact assessment

Table 1.2 Radiation and dosimetry units (Kennedy et al., 1990)

Radiation	*Symbol*	*Typical Energy (Mev)*	*Linear Energy Transfer (LET)*	*Characteristics*
alpha particle	α	4-8	High	Energy dissipated in a few centimetres of air or 0.04mm of tissue
beta particle	β	1-4	Low	Energy dissipated by a thin sheet of metal or 5mm of tissue
gamma radiation	γ	High energy, short wavelength	Low	ã radiation is surplus energy released after radioactive decay. ã rays are extremely penetrating and may completely penetrate tissue. ã rays are absorbed by dense materials such as concrete

Radiation doses received by wildlife or humans can be from external or internal sources:

- **External exposure**

  Radiation emitted from radionuclides in the air, in water or on/in the ground can interact with organisms leading to an external dose.

- **Internal exposure**

  Organisms may receive an internal dose following the uptake of a radionuclide via inhalation, ingestion or absorption of a radionuclide, which then continues its radioactive decay process inside the organism.

The range of an α particle in soft tissue is around 50 ìm; thus internal tissues and organs will not, in general, receive significant doses from external exposure. A localised high dose may be received at the point of contact of the α emitting source with the outer surface of the organism.

In contrast, α particles arising from an internal source of radiation will result in localised irradiation at the site, or in the tissue/organ, of deposition. γ rays produced internally may have less impact on an organism than α particles due to their higher penetrating ability; so smaller organisms will generally receive a lower dose from an internal γ source than a larger organism. Internal radiation from less penetrating sources i.e. α and β particles are generally of more concern than γ rays because their energy is more likely to be deposited within an organism.

The extent of internal exposure is dependent upon the type of radiation, uptake rate and elimination rate of the radionuclide from the body, and exposure duration (NRPB, 1998). For example, tritium (^{3}H) is readily absorbed by, and distributed through, the body. It emits low energy β particles and so the organism will receive a uniform low dose. In contrast, ^{241}Am is not readily absorbed by the body although it can be inhaled and accumulate in the lung. ^{241}Am emits both α particles and a low energy γ ray, resulting in localised radiation exposure to the tissues around the deposition site.

1.4 Exposure to ionising radiation

All living organisms are exposed to ionising radiation. Exposure to ionising radiation can arise from both natural and anthropogenic sources of radionuclides (Table 1.3), and these need to be considered together in order to evaluate the effects of anthropogenic releases of radioactivity, particularly as exposure to radiation from natural sources accounts for approximately 85% of the dose to the UK human population.

Table 1.3 Sources of annual average ionising radiation dose to the UK population (NRPB, 1998)

Source	*% Contribution*
Natural Radon	50
Medical	14
Natural ã	14
Cosmic	10
Internal	11.5
Fallout	0.2
Occupational	0.3
Discharges	<0.1
Products	<0.1

Situations exist where the contribution of anthropogenic radionuclides to the radiation dose to wildlife outweigh that derived from natural sources, for example around point sources of radioactive discharges. Exposure to natural sources of radiation varies greatly, with background radiation differing between geographical regions as a result of the local geology. It is important to consider both the radiation dose received by organisms due to their geographical location as well as through exposure as a result of their habits and diet (Chapter 2) and the types of radiation involved.

1.5 Sources of ionising radiation in the environment

1.5.1 Natural sources

Natural radionuclides are present either because they are primordial, with half-lives comparable to the age of the earth, or because they are continually generated by the decay of these long-lived precursors, or because they are continuously generated by cosmic radiation. The natural radionuclides can be divided into two groups:

- series radionuclides, such as ^{238}U, ^{232}Th and ^{235}U, which themselves decay to give rise to radioactive progeny;
- non-series radionuclides that are produced through the interaction of cosmic radiation with elements in the atmosphere (e.g. ^{14}C from ^{14}N).

Table 1.4 lists the most significant primordial series radionuclides and their half-lives. The underlying geology of an area can affect the concentration of natural series radionuclides present, for example higher concentrations typically found in granite compared with sandstone or limestone areas. Technological advances have led to releases of these natural radionuclides, which would otherwise have remained trapped in the Earth's crust. This is known as technologically enhanced natural radiation (TENR) and can be the result of a wide range of human activities including the burning of fossil fuels, mining and smelting of natural ores (including the production and subsequent application of fertilisers).

Table 1.5 lists the most significant non-series radionuclides produced through the interaction of cosmic radiation and elements in the atmosphere. The continuous cosmic ray bombardment of the atmosphere replenishes the earth's supply of these radionuclides.

Table 1.4 Significant primordial radionuclides (Hughes and Shaw, 1996)

Radionuclide	*Half Life (years)*
Potassium-40 (^{40}K)	1.28 x10^{9}
Rubidium-87 (^{87}Ru)	4.80 x10^{10}
Thorium-232 (^{232}Th)	1.41 x10^{10}
Uranium-235 (^{235}U)	7.04 x10^{8}
Uranium-238 (^{238}U)	4.47 x10^{9}
Uranium-238 (^{238}U)	4.47 x10^{9}

Table 1.5 Significant cosmic ray produced radionuclides (Shapiro et al., 1993)

Radionuclide	*Half Life*
Tritium (^{3}H)	12.3 years
Beryllium (^{7}Be)	53.7 days
Carbon-14 (^{14}C)	5,370 years
Phosphorus-32 (^{32}P)	14.3 days
Phosphorus-33 (^{33}P)	25.3 days
Sulphur-33 (^{33}S)	87.2 days

1.5.2 Anthropogenic sources

Radionuclides are released into the environment from a variety of anthropogenic sources and processes, which include amongst others, the nuclear fuel cycle and nuclear weapons testing. The fuel cycle includes mining, milling, fuel enrichment, fabrication, reactors, spent fuel storage, reprocessing, waste storage and decommissioning. Figure 1.2 shows the locations of principal anthropogenic sources of radioactive discharges in the UK. Tables 1.6 to 1.8 compare the releases of radioactivity to the environment from anthropogenic sources. Tables 1.7 and 1.8 highlight more specifically typical discharges from nuclear power stations.

Disposal of radioactive waste is authorised by the Environment Agency in England and Wales, and the Scottish Environment Protection Agency (SEPA) in Scotland. Disposal may occur from nuclear licensed sites or from non-nuclear licensed sites. The Food Standards Agency (FSA) and SEPA have a regulatory responsibility to ensure the safety of the food chain and so monitor food products for radioactivity. Until April 2000, this was the responsibility of Ministry of Agriculture, Food and Fisheries (MAFF). The FSA and SEPA have the power to ban the sale of food if required.

Figure 1.2 Location of principal sources of radioactive discharges in the UK (DETR, 2000)

Table 1.6 Comparison of releases of selected radionuclides to the environment (see also Tables 1.7 and 1.8)

Source Term	*Radioactivity released (PBq)*				
	^{137}Cs	*^{131}I*	*Pu isotopes*	*Total Release*	*Naturally occurring radionuclides*
Intentional Releases					
Nuclear fuel reprocessing plant	0.0025 [3]	0.000017 [3]			
Nuclear weapons testing	948 [4]	675,000 [4]	153 [4]	2,566,000 [4]	
Conventional power generation					0.084 [1]
Mineral Processing Industries					8.1 [6]
Manufacture of radioactive products	0.00002 [2]	0.0005 [2]			
Accidental Releases					
(refer to Chapter 1 and Appendix 1)					
Windscale	0.044 [5]	0.59 [5]			
SNAP-9a (satellites)			0.629 [5]		
Chernobyl	37 [5]	670 [5]		2,000 [5]	
Kysthym				74 [4]	
Three Mile Island		0.0011 [5]		100 [5]	

References: **1** includes isotopes of uranium, thorium, lead, potassium and polonium as TBq released annually and is based on a new coal fired power station in the UK which retains 99% of the fly ash produced, Shapiro *et al.*, 1993; **2** EA Annual Report for 1999 - Radioactivity in the Environment values in TBq released annually [EA, 2000]; **3** UNSCEAR, 2000 - Cap de la Hague annual discharges for 1997; **4** UNSCEAR, 2000; **5** Appleby and Luttrell, 1993; **6** UNSCEAR, 2000 based on uranium, thorium, radium, radon, lead, polonium and potassium annual releases. Releases from phosphorus ore processing, oil and gas extraction account for around 85% of the release.

- **Radioactive waste arising from the nuclear fuel cycle**

  The main sources of radionuclide release from the nuclear fuel cycle result from nuclear power and fuel reprocessing plants.

  Nuclear reactors harnessed for power generation use and produce copious quantities of radionuclides. The actual inventory of radionuclides present within a reactor core at any time is dependent upon the type of reactor and its operating history. Under normal operating conditions gaseous, liquid and solid wastes are produced by the fission and neutron activation processes, which cause contamination of the materials used in the reactor or its housing. Some of these radionuclides may also be discharged to the environment under authorisation (Table 1.7).

  Spent nuclear fuel is also periodically replaced during, and removed at the end of, the reactors operational life. In the UK, the spent nuclear fuel is removed from the reactor and, after a period of storage, sent for reprocessing.

  Within the spent fuel of nuclear reactors around 3% of the original uranium is used. The majority of the spent fuel waste therefore comprises unused uranium and generated plutonium, typically around 96% and 1% respectively. The aim of nuclear fuel reprocessing is to reclaim the unused uranium by separating it from the waste material (e.g. fission products). In the UK, this reprocessing is carried out at the British Nuclear Fuels Ltd. (BNFL) Sellafield site in Cumbria.

The Sellafield complex has been involved in reprocessing of spent nuclear fuel since 1952, with the existing Magnox fuel reprocessing plant being constructed in 1964. The thermal oxide reprocessing plant (THORP) was commissioned in the early 1990's to reprocess uranium oxide fuel from more modern nuclear power stations in Britain and overseas. Table 1.8 illustrates the principal discharges from the nuclear fuel reprocessing plant at Sellafield.

Table 1.7 Typical discharges from nuclear power plants in 1999 (TBq) (EA, 2001a)

Reactor Type and Establishment	*Radionuclide*	*Discharge Route*	*Discharges (TBq)*
Magnox	^{3}H	Liquid	0.84
(Hinkley Point A)	^{137}Cs	"	0.44
	^{3}H	Gaseous	3.30
	^{14}C	"	1.58
	^{35}S	"	0.05
	^{41}Ar	"	1,140
Advanced Gas Cooled Reactor (AGR)	^{3}H	Liquid	356
(Hinkley Point B)	^{35}S	"	0.59
	^{60}Co	"	0.0004
	^{3}H	Gaseous	2.18
	^{14}C	"	1.08
	^{35}S	"	0.12
	^{41}Ar	"	34.1
	^{131}I	"	0.00001
Pressurised Water Cooled Reactor (PWR)	^{3}H	Liquid	56
(Sizewell B)	Noble gases	Gaseous	7.30
	Halogens	"	0.0003
	^{3}H	"	0.69
	^{14}C	"	0.25

Table 1.8 Principal discharges from the licensed site at Sellafield, Cumbria, UK in 1999 (EA, 2001)

Sea Pipeline Discharge Route		***Gaseous Discharges***	
Radionuclide	***TBq***	***Radionuclide***	***GBq***
Á	0.13	Á	0.017
Â	110	Â	2.15
^{3}H	2,520	^{3}H	250 000
^{14}C	5.76	^{14}C	2650
^{60}Co	0.89	^{35}S	99.6
^{90}Sr	31.2	^{60}Co	0.004
^{95}Zr+^{95}Nb	0.18	^{85}Kr	94 900 000
^{99}Tc	68.8	^{90}Sr	0.006
^{106}Ru	2.67	^{106}Ru	0.960
^{129}I	0.49	^{125}Sb	0.253
^{134}Cs	0.34	^{129}I	25.3
^{137}Cs	9.11	^{131}I	4.02
^{144}Ce	0.60	^{137}Cs	0.583
Pu á	0.12	Pu á	0.107
^{241}Pu	2.87	^{241}Pu	0.830
^{241}Am	0.03	^{241}Am+ ^{242}Cm	76.6
U (discharge in kg)	536		

The radioactive wastes produced during reprocessing are physically separated into three categories (Table 1.9):

- High level waste (HLW) consists of the fission products arising from the chemical separation process. HLW is currently vitrified and stored on site at Sellafield, pending a Government decision on the final management route.
- Intermediate level waste (ILW), which consists of fuel cladding and contaminated equipment, is encapsulated, often with a grout or cement in drums and is currently stored on site. The final management route for storage location of the ILW is currently under debate; one option is disposal in a deep underground repository.
- Low-level solid radioactive waste (LLW) is disposed of in concrete vaults at the Drigg low-level waste repository in Cumbria and consists of a variety of materials which have been contaminated with radioactivity, for example, disposable gloves and paper waste. Drigg receives LLW from a range of sources.

Table 1.9 Definition of high-, intermediate- and low level radioactive waste (NRPB, 1998)

Waste Category	*Abbreviation*	*Composition and Storage*
High Level Waste	HLW	Most of the fission products and actinides from the fuel cycle, high heat creation, low bulk – usually stored as a liquid or vitrified into glass blocks. Both are stored in special cooling facilities awaiting eventual disposal
Intermediate Level Waste	ILW	Larger quantities of fission products and actinides with long half lives, low heat creation, high bulk – usually encapsulated in a concrete, bitumen or resin. Currently stored at various nuclear licensed sites awaiting decision on final disposal route
Low Level Waste	LLW	Contains various radionuclides in general refuse and rubble, tends to be low activity, high bulk – not usually processed except for compaction, direct disposal to authorised burial site

- **Solid radioactive waste repositories**

  Disposal method and location should ensure that the emissions of radionuclides are very small when compared to the discharges from routine operations of nuclear licensed sites and with background radiation from naturally occurring radionuclides (Patton *et al.*, 2001). Guidance on standards applied by UK regulation are published in "Disposal facilities on land for low and intermediate level radioactive wastes: guidance on requirements for authorisation" (Environment Agency, 1997).

- **Low level radioactive atmospheric discharges**

  Radionuclides are discharged with gaseous effluents into the atmosphere from nuclear licensed sites (Tables 1.7 and 1.8).

- **Low level radioactive liquid discharges**

  Liquid effluent containing radionuclides is discharged from nuclear (Tables 1.7 and 1.8) and non-nuclear (Table 1.10) sites. The effluent is usually discharged into sewerage, rivers, lakes or the sea beyond the low tide level. Liquid effluent discharge data are published annually by the site operators and regulatory bodies (e.g. Tables 1.7 and 1.8). The extent of liquid discharges from

nuclear power plants is dependent on the reactor type (Table 1.7). Liquid radioactive discharges from non-nuclear sites (which are generally made to sewer systems) are released into freshwater systems via sewage works, and may include a wide variety of radionuclides (Table 1.10).

Table 1.10 Typical monthly radionuclide releases from Beckton sewage works (London, UK), from non-nuclear sites (Titley et al., 2000)

Nuclide	*Estimated activity* $Bq\ m^{-3}$	*Nuclide*	*Estimated activity* $Bq\ m^{-3}$
^{3}H	84,000	^{84}Rb	48
^{99m}Tc	12,000	^{111}In	42
^{89}Sr	42	^{51}Cr	33
^{125}I	2,400	^{33}P	18
^{14}C	1,600	^{90}Y	8.9
^{131}I	1,000	^{133}Xe	1.8
^{35}S	350	^{75}Se	1
^{32}P	210	^{57}Co	0.1
^{201}Tl	210	^{58}Co	0.06
^{123}I	200	Other β/γ	2,000
^{67}Ga	110	α-emitters	0.005

- **Fallout from nuclear weapons testing**

  Radionuclides of concern from the fallout of atmospheric nuclear weapons testing are similar to those arising from operations of nuclear power plants. A range of different radionuclides may be present in nuclear weapons testing fallout. The radionuclides produced are, however, dependent upon the type and composition of the nuclear device.

  Radionuclides are released as a result of the nuclear reaction within the core of the weapon, or from ejected unused fragments of the uranium/plutonium core. Depending upon the type of weapon's test (atmospheric, ground burst or underground) these released radionuclides may be injected into the planet's atmosphere, where they are dispersed around the globe. Nuclear weapons testing began in the early 1950s and peaked in 1961-1962, with the greatest deposition of radionuclides observed in 1963. Total emission to the environment following testing has been estimated as 910,000 TBq for ^{137}Cs (Cambray *et al.*, 1989) and 13,300 TBq for $^{239+240}Pu$ (Perkins and Thomas, 1980).

  Following the cessation of atmospheric weapons testing (treaty signed in 1968), the annual deposition of nuclear weapons-derived radionuclides across the UK has been declining steadily (Cambray *et al.*, 1989). For example ^{137}Cs deposition in the Northern Hemisphere in 1990 was <1 PBq compared to 150 PBq in 1963 (Playford *et al.*, 1992).

- **Accidental or unplanned releases of radioactive material**

  Unplanned releases of radionuclides into the environment can result from unauthorised discharges, leaks, explosions or fires. Accidents have arisen during nuclear weapons production as exemplified in these major incidents (see Table 1.6 for levels released).

  - In 1957 at Kyshtym (Soviet Union) a chemical explosion in a storage tank containing $250m^3$ of HLW resulted in the release of 7.4×10^{17} Bq of activity to the atmosphere (Appleby and Luttrell, 1993; Nikipelov, 1989). The resulting doses to the environment

were dominated in the first year by exposure to ^{144}Ce and ^{144}Pr, whilst ^{90}Sr was the principal contributor to long-term exposure.

- The Windscale accident of 1957 in Cumbria resulted in the uncontrolled release of 2.2 x 10^{13} Bq of ^{137}Cs and 1.6 x 10^{9} Bq of $^{239+240}$Pu from a reactor into the atmosphere over a 24-hour period (Crick and Linsley, 1982).
- The accident at Three Mile Island (USA, 1979) was the first nuclear power plant accident culminating in approximately 50% of the fuel melting, releasing fission products to the reactor vessel (Gerusky, 1988). Most of the radioactivity remained contained within the reactor vessel and its containment structure. It is estimated that the amount of radioactivity released to the environment was in the order of 10^{17} Bq and consisted mainly of the noble gases ^{133}Xe, ^{133m}Xe and ^{135}Xe (Gerusky, 1988).
- The accident at Chernobyl (Ukraine, 1986) is considered to be the most serious accident involving a nuclear reactor. Radioactive material containing spent fuel, noble gases and volatile radionuclides was ejected into the atmosphere and emissions continued for several days as a result of the subsequent fire in the graphite moderator. Appendix 1 discusses the latest research on the impact of the Chernobyl accident on wildlife in more detail.

Another source of unplanned release comes from satellites (see Table 1.6). Many satellites use nuclear auxiliary power units (SNAPs), which utilise the heat released from radionuclide decay to generate electricity for satellite equipment. If, as with SNAP-9A in 1964, a satellite fails to attain orbit the satellite can re-enter the earth's atmosphere. During re-entry the nuclear power source will be volatilised resulting in the release of radionuclides. These devices often use plutonium as a fuel source (SNAP-9A contained 629 TBq of ^{238}Pu), which will be dispersed globally and thus contribute to the exposure of wildlife and humans from ionising radiation particularly via inhalation routes (Appleby and Luttrell, 1993).

1.6 Summary

There is a wide variety of sources of radionuclides in the environment; natural, technologically enhanced naturals, anthropogenic and accidental release. An understanding of the amount of radioactivity is not sufficient for the assessment of possible environmental impact, knowledge of how the radioactivity can be transferred in the environment and how wildlife can be exposed is also required, as discussed in Chapter 2.

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2. Pathways of Exposure for Plants and Animals to Ionising Radiation

All organisms are exposed to sources of naturally occurring ionising radiation, but anthropogenic activities can give rise to increased exposure, as outlined in Chapter 1. This Chapter aims to review the different routes of exposure, consider the different ecosystem compartments and then provide an overview of the pathways through which biota can be exposed to ionising radiation.

The Chapter also reviews the latest research on the behaviour and transfer of radionuclides in each ecosystem, i.e. since the report for Nature Conservancy Council "Radioactivity and Wildlife" (Kennedy *et al.*, 1990).

Chapter 6 Section 6.2.3 provides additional information on the radionuclides selected for the assessment within this report, including a review of their environmental source, behaviour and chemical properties. Furthermore, Section 6.4 describes the derivation of, and provides in Tables 6.7 to 6.9 values, for concentration factors derived from the literature reviewed in the following Sections for use in the assessment. These concentration factors provide an indication on the likely bioaccumulation of radionuclides into a number of different ecosystem components considered in this Chapter.

2.1 Exposure pathways

Wildlife can be exposed to ionising radiation though a number of different routes including:

- External irradiation;
- Plant root uptake from soil;
- Foliar absorption;
- Inhalation of:
  - resuspended material;
  - gaseous radionuclides;
- Ingestion of:
  - plant material;
  - animal material;
  - microbial material;
  - soil;
  - water.

There are many interactions between biota and their surroundings which may influence the uptake and transfer of radionuclides. Figure 2.1 provides a simplified food chain diagram demonstrating these relationships. Radionuclides may be transferred through the food chains from the soil or sediment compartment through different trophic levels, e.g. plant uptake, into herbivores, carnivores and higher predators.

The transfer rates of different radionuclides will be affected by their chemical form and their bioavailability within the ecosystem. Furthermore, some radionuclides are considered to be more biological mobile because they are analogues to essential elements which the plant or animal requires and can be absorbed into organisms more readily. For example, caesium (e.g. ^{137}Cs) and strontium (e.g. ^{90}Sr) are considered to be natural analogues of potassium and calcium respectively (Shaw, 1993). Both potassium and calcium have essential functions in biological organisms and therefore uptake routes exist, consequently both ^{137}Cs and ^{90}Sr tend to accumulate in biological organisms. This is reflected in the concentration factors described in Tables 6.7 to 6.9.

Radionuclides will usually follow the energy flow as indicated in Figure 2.1, including excretion and recycling of material after death of the biological material.

Unusual pathways of radionuclide transport off nuclear sites have been considered recently due to the observed high radionuclide concentrations in pigeons roosting on nuclear sites in the UK (Copeland Borough Council *et al.*, 1999) Identification of similar unusual pathways involving biota and their role in transferring radioactivity from nuclear sites to the environment has been recently assessed (EA, 2001b) and will not be discussed further in this report.

It will be seen in the following Sections that the pathways of exposure to ionising radiation are similar for both humans and biota. It is the magnitude of the exposure through increased occupancy by biota of radioactively contaminated areas compared to humans or through differences in the uptake and accumulation of the radionuclides into the biota which determines the level of impact from ionising radiation. The only significant 'unusual' pathway identified to date which may lead to high levels of exposure to wildlife but not humans is that of deep sea disposal of radioactive waste. In this specific case, biota may potentially be exposed to very high levels of ionising radiation but the pathway for the radionuclides to return to, and cause exposure in, humans is so long that only long-lived radionuclides will be involved. In this case, the resulting exposure to humans is low and consequently under the existing radiological protection provided by the ICRP, biota could be exposed at levels greater than that permissible for humans. This pathway of exposure to ionising radiation has been described in detail elsewhere (Pentreath and Woodhead, 1988; IAEA, 1988a).

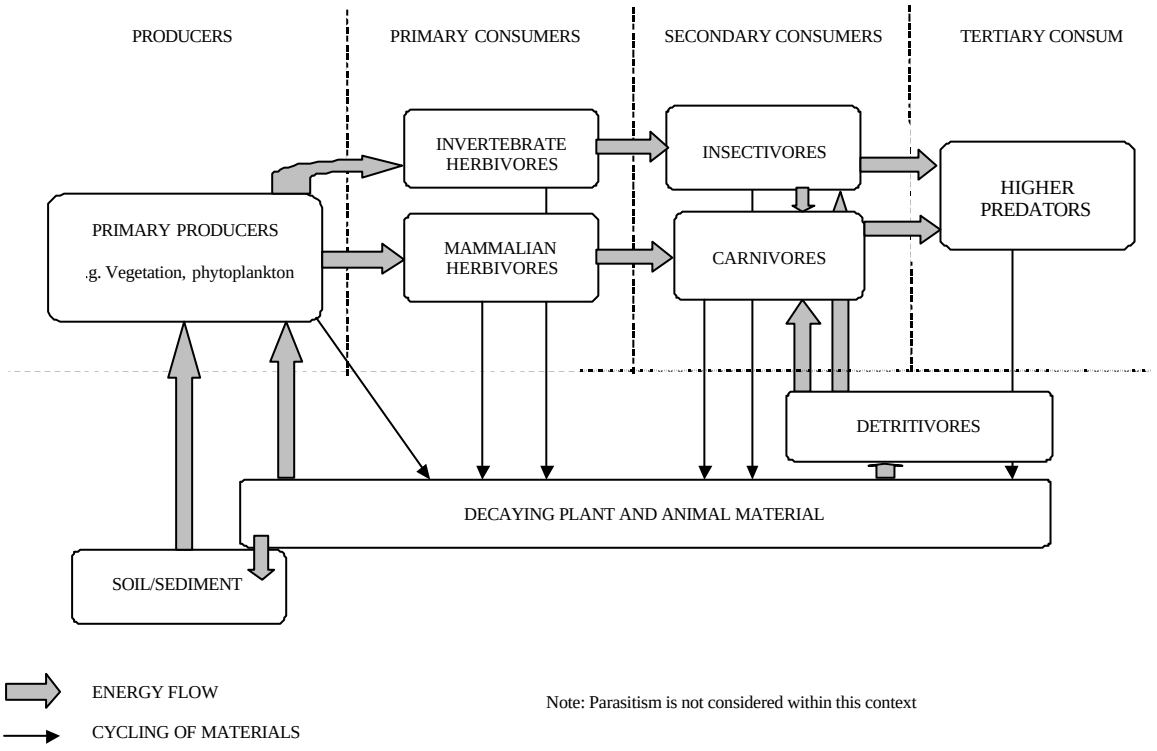

Figure 2.1 Simplified food chain diagram demonstrating the flow of energy and material recycling within an ecosystem

2.2 Ecosystem components

Radionuclides can enter the environment through atmospheric wet and dry deposition or/and discharges to the water. Figure 2.2 summarises the information presented in this Section.

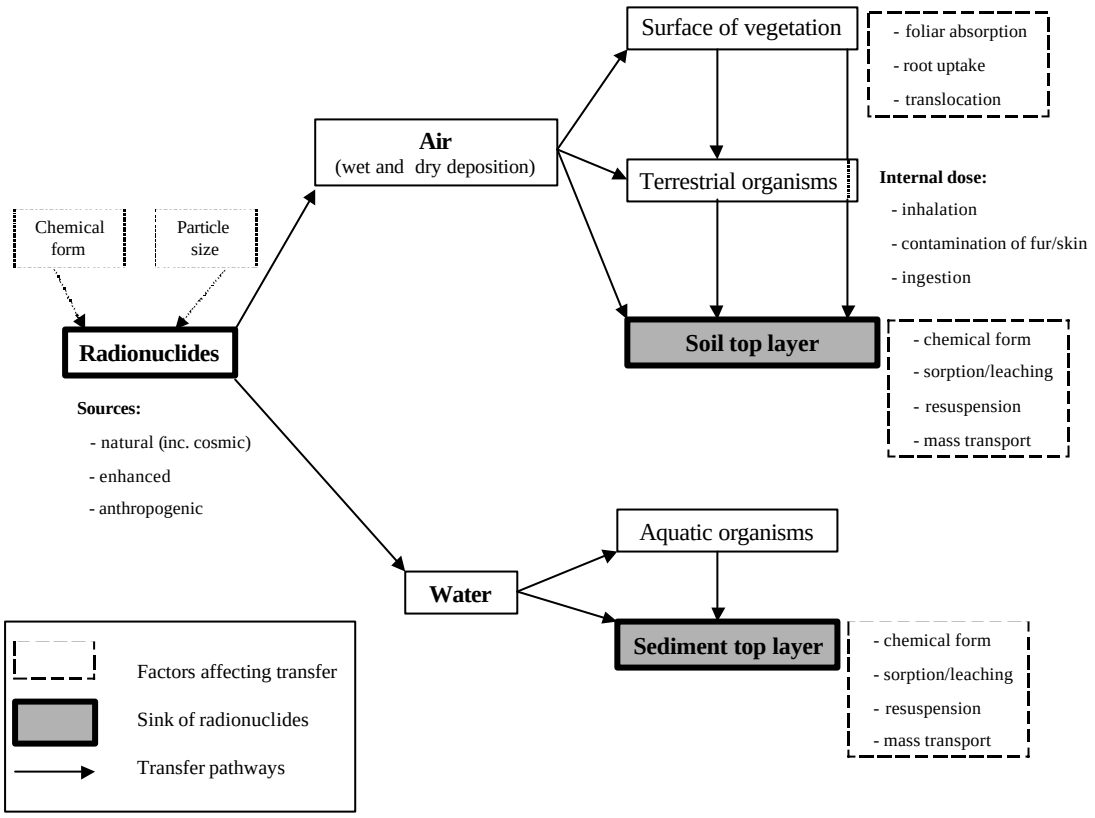

Figure 2.2 Summary of transfer pathways of radionuclides in ecosystem components

The majority of radionuclides are transported in the atmosphere as aerosols with various physico-chemical properties. This gives rise to different deposition mechanisms, which are generally grouped into:

- wet (or precipitation scavenging) deposition which involves the removal of particulate matter and gases from the air by different forms of precipitation, leading to the incorporation of radioactivity in rainwater; and
- dry deposition which occurs continuously, with particles and any associated radionuclides being deposited through diffusion, impaction, interception and sedimentation processes.

All deposition processes are affected by air turbulence, size and nature of airborne particulates, as well as the structure and nature of the ground and plant surfaces (Harrison *et al.*, 1993; Nicholson, 1988 a, b).

In terrestrial ecosystems, airborne radionuclides may be deposited onto the surface of vegetation or soil, whilst in aquatic ecosystems they may be deposited on the water or exposed sediments. Radionuclides may also enter the aquatic ecosystem via surface runoff, leaching through river catchment areas, or discharges to watercourses. In all cases, radionuclides will be further distributed into ecosystem components such as soil/sediment, plants and animals.

2.2.1 Soil/sediment

Most radionuclides released into the environment are ultimately transferred to soil or sediment. The behaviour of radionuclides therein, and their deposition rate to vegetation or soil and subsequent relocation to soil are dependent upon a number of factors, including: the particle size with which the radionuclides become associated, climatic conditions and retention of the deposited material on the surfaces of vegetation or entrapment of sediment. Radionuclides in the soil/sediment also contribute

significantly to the external irradiation of biota. As a result, soils and sediments form a suitable medium for studying and monitoring the spatial distribution of radionuclides in the environment and may be used as the basis for any impact assessment approach.

Major factors which influence the availability of radionuclides in soil or sediment are: their chemical form, sorption, resuspension, mass transport and leaching (Kirchmann *et al.*, 1993; Thiry, 1990; Thiry and Myttenaere 1993; Bruckmann and Wolters, 1994).

- **Chemical form**

Different sources of radioactivity can influence the chemical form of the radionuclide. For example, ^{90}Sr and ^{137}Cs deposited from weapons testing fallout consisted largely of water soluble and exchangeable forms, and so were potentially available for uptake into biota (Kirchmann *et al.*, 1993). In contrast, the same radionuclides released within the 30 km exclusion zone around the Chernobyl power plant were associated with fuel particles which are insoluble in water and therefore do not readily breakdown in soil and are biologically unavailable (Konoplev and Bobovnikova, 1990).

Many transuranic elements such as plutonium and americium exhibit a range of chemical forms depending upon their oxidation state. This is important when considering their mobility, e.g., plutonium is water soluble in higher oxidation states and thus, may be more biologically available. pH can also be a major influence of the oxidation status of transuranics (Berrow and Burridge, 1991; van Bergeijk *et al.*, 1992).

- **Sorption/leaching**

Deposited radionuclides may bind to ion exchange sites on particles or organic matter. They may also be present in the soil solution (Morgan, 1990). Leaching is determined by their sorption, the soil structure and rainfall rates. Leaching rates tend to be greater under high rainfall, or in soils containing a higher proportion of sand particles. For example, Schimmack *et al.* (1994) demonstrated that forest organic soils retained a higher proportion of ^{60}Co and ^{137}Cs under light rainfall compared with heavy rainfall.

Leaching is important as it determines the distribution of radionuclides in the soil profile, which influences the external exposure of biota. In most natural, or semi-natural, ecosystems the soil is undisturbed (e.g, not ploughed), and the bulk of the radioactivity is in the soil upper 10-15 cm (Copplestone *et al.*, 2000; Morgan, 1990). Furthermore, the top 10-15 cm is the rooting zone for many plant species, and where most organic matter is present. Both of these factors can influence plant uptake of radionuclides (van Bergeijk *et al.*, 1992; Burmann *et al.*, 1994).

Other metal ions in the soil solution will compete with radionuclides (Sposito, 1989) and affect the availability of radionuclides for plant uptake, a factor related to the soil cation exchange capacity (Shaw and Bell, 1991). Studies have shown that both caesium and plutonium can be absorbed by plants but plutonium being in an exchangeable form will not be readily available for uptake. One of the major causes for the continued plant uptake of Chernobyl derived ^{137}Cs in certain areas of the UK is due to exchange sites being more available in organic soils (Davydov *et al.*, 1990).

- **Resuspension**

Resuspension is defined as "*the entertainment into the atmosphere of surface contamination that was originally airborne but deposited to the ground surface*".

Many radionuclides remain bound in the upper soil layers, where resuspension can occur, due to their low mobility. Resuspension can be affected by wind speed, moisture, vegetation cover, season, mechanical disturbance and particle characteristics (Morgan, 1990). Resuspension can provide an important source of contamination for plants and animals and may be particularly significant if inhalation of fine resuspended particles occurs.

- **Mass transport**

Mass transport is the movement of radionuclides through soil or sediment by physical or biological processes. Physical movement involves the downward migration of soil particles through macropores. Biological transport is dominated by soil organisms.

The burrowing activity of larger animals facilitates migration and exposure of radionuclides through the soil profile. Bishop (1989) reported that this represents a significant pathway for the migration of radionuclides. Animals (e.g. ants, mice, rabbits and badgers) relocate material both vertically and horizontally during the construction of burrows, tunnels and chambers. Wildlife can be exposed during construction, and subsequent resuspension of soil particles. These tunnels can also facilitate the movement of gaseous radionuclides. Thus burrowing activities can result in both direct and indirect routes of exposure of organisms to radionuclides.

2.2.2 Vegetation

Much of the available data is focused on terrestrial crop plants because of the direct relevance and possible transfer to humans. Radionuclides enter aquatic and terrestrial plants through foliar absorption and/or root uptake. It has been demonstrated that radionuclide uptake is plant specific with recent work by Broadley *et al.*, (1999) demonstrating that this may be related to the genetic make up of plants.

- **Foliar absorption**

Data on foliar absorption of soluble radionuclides are scarce in the literature, relative to those for gaseous radionuclides. Shaw *et al.* (1992) discussed factors affecting the absorption and translocation of radionuclides applied in solution, and uptake of gaseous forms of radionuclides such as ^{35}S, ^{14}C and ^{3}H has been well studied (Collins and Gravett 1995). The stable isotopes of these radionuclides are used in processes such as: photosynthesis, respiration and transpiration. Models exist of their absorption, use and subsequent redistribution within the plant. Numerous studies have also investigated the deposition of radionuclides to terrestrial plants using, for example, wind tunnel experiments (Ould-Dada, 1996; Collins and Gravett, 1995).

Foliar absorption may occur via the stomata or across the cuticle. Stomatal absorption depends on a range of factors, including: surface tension of any solution, contact angle, morphology of pore and leaf, duration of contact, particle size of radionuclides (Kirchmann *et al.*, 1993). The chemical composition and thickness of the cuticle affect cuticlar uptake.

- **Root uptake**

Most radionuclides absorbed by root uptake become incorporated into plant tissues, where excretion may also take place. Both soil and plant factors determine root uptake. Soil properties determine the availability of radionuclides for root uptake (Section 2.2.1), which may occur via passive or active transport mechanisms. Different plant species can then exhibit varying degrees of fixation within the plant tissues. Many studies have investigated the uptake of radionuclides using different soils and plant species, and the level of uptake of a particular radionuclide into different plant species may be compared using soil to plant transfer factors (Bettencourt *et al.*, 1988).

Plant factors that affect root uptake include rooting depth, root morphology, and solute concentrations of a given radionuclide in both the soil solution and within the plant. Micro-organisms can also affect the plant uptake (Berthelsen *et al.*, 1995).

A number of models of radionuclide transfer have been produced (e.g. Thorne and Coughtrey, 1983; Crout *et al.*, 1990; Toal *et al.*, 2001). These have application in determining the internal concentrations of radionuclides within organisms for impact assessment purposes.

- **Translocation**

Following root or foliar absorption, radionuclides can be translocated within the plant. Many studies have focused on the above ground fraction of the plant, and their translocation to underground storage organs, because of the direct relevance to human exposure. Most studies have demonstrated that

radionuclides with natural analogues tend to be translocated to actively developing meristematic regions e.g. root tips, flowers, fruits and vegetative growing tips.

2.2.3 Transfer to terrestrial organisms

Many uptake, distribution and retention of radionuclides studies have focused on laboratory animal experiments and domestic stock to assess risks of radionuclide transfer to humans (Kirchmann *et al.*, 1993). Studies on the transfer and impact of radionuclides on wildlife are on the increase (e.g. Copplestone *et al.*, 1999; 2000; Rudge *et al.*, 1993a, b; Mascanzoni *et al.*, 1990; Matson *et al.*, 2000).

The important parameters involved in the transfer to, and metabolism in, animals and have been summarised by Kirchmann *et al.* (1993):

- the fraction of an orally ingested radionuclide absorbed by the gastro-intestinal tract, and transferred to the body's systemic circulation;
- the activity level and bioavailability of radionuclides in food items;
- the distribution of the ingested fraction into different organs and tissues;
- biological, ecological and physical half-lives of the radionuclide in the organism;
- the fraction of radionuclides excreted in urine, faeces, milk and sweat;
- the resuspension of radionuclides within the environment; and
- the fraction of radionuclides in the lungs.

The pathways of exposure can be summarised as:

- Inhalation;
- Contamination of fur and skin. (This may be important for actinide exposure because of their low gastro-intestinal transfer and may give rise to localised effects, and subsequent ingestion during grooming (Lang *et al.*, 1993);
- Ingestion. (This is the most significant pathway for the uptake of radionuclides, and is proportional to feeding rate and radionuclide concentration in food items.

The accidental ingestion of soil has been shown in a number of models to account for much of the internal dose to an organism (Toal *et al.*, 2001; Crout *et al.*, 1993; Beresford and Howard, 1990).

2.3 Transfer pathways in ecosystems

The semi natural ecosystems in the UK affected by authorised radioactive discharges, and which have been studied, include: grasslands, coniferous and deciduous woodlands, sand dunes, saltmarshes, and freshwater, estuarine and marine ecosystems. Following the Chernobyl accident, semi-natural ecosystems were contaminated in many countries. The resulting contamination led to exposure of both wildlife and humans who utilise semi natural ecosystems for food and other products (e.g, timber). The significance of this has been recognised (Desmet *et al.*, 1990).

Tables 2.1 to 2.3 provide a brief summary of the influential factors and components of ecosystems likely to accumulate radionuclides. The lists are not intended to be exhaustive, but serve to illustrate the components that play an important part in radionuclide transfer pathways.

The latest research on the behaviour and transfer of radionuclides in each ecosystem is reviewed in the following Sections. For each ecosystem, the literature on naturally occurring radionuclides is also summarised.

Table 2.1 Brief summary of key components: terrestrial ecosystems (based on Kennedy et al. (1990) and findings described in Section 2.3)

	Habitats, in decreasing order of radionuclide accumulation				
	Woodlands (most)		**Salt marshes**	**Coastal grasslands**	**Sand dunes (least)**
	Coniferous	**Deciduous**			
Main source of radionuclides	Atmosphere (aerosol and particulate)	Atmosphere (radionuclides in gaseous/aerosol form)	Sediments	Sand particles	Aerosol, wind driven
Main factors influencing radionuclides accumulation	Soil type (usually nutrient poor) Seasonal variation Availability of food items Needles characteristics affect aerosol deposition rate	Soil type (usually nutrient rich) Seasonality Prevailing weather conditions	Sediment type (e.g. actinides bound to fine grained particles, so biologically available) Sediment processes Particle deposition Heavy rain Seasonal factors Tidal movements	Soil type (higher organic content than in sand dunes) Organic loading of particles Solubility of radionuclides Soil properties*	Sand type Soil well drained so low bioavailability to vegetation and low accumulation of radionuclides Grain size (coarse sands have fewer available adsorption sites) Origin of the sand (i.e. physical characteristics)
Components most likely to accumulate radionuclides	Leaf litter on top soil Fruiting body of fungi (some species) Invertebrates involved with decomposition processes Small mammals (depending on food availability) Soil dwelling animals (for specific natural radionuclides e.g. radon) Increasing order of radio-resistance: coniferous tree (most sensitive)>deciduous tree>shrub>herbaceous plant>grassesand sedges>mosses, lichen and some algae (least sensitive)		Particle bound radionuclides accumulate in fine sediments Vegetation	Animals grazing on silt and vegetation Increased accumulation with increasing organic soil content Uptake to plants of transuranic elements and radiocaesium decreases with increasing clay content	

* Soil properties are depend on factors including: microbial activity, soil temperature, moisture, pH, cation exchange capacity, decomposition rate

Table 2.2 Brief summary of key components: freshwater ecosystems (based on Kennedy et al. (1990) and findings described in Section 2.3)

	Still waters	**Running water**
Main source of radionuclides	Atmospheric deposition (wet and dry) Erosion of catchment Discharge to water course	
Main factors influencing radionuclides accumulation	Dimensions of water body Erosion characteristics of catchment Chemical composition of water* Chemical and decay properties of radioisotope (e.g. Pu sorbed to clay, Sr soluble, Cs varies) Seasonality Rainfall intensity	Speed of flow Geological characteristics (slope, rock type, rainfall) Deposition rate of sediments Suspended sediment load Storm run-off
Components most likely to accumulate radionuclides	Bottom sediments: less reactive radionuclides concentrate in epilimnic layer *vs* particle reactive nuclides being evenly distributed. Higher silt/organic sediment content result in greater accumulation of natural radionuclides	Bottom sediments: increased radionuclide concentrations with increased silt/clay content and decreased flow
	Roots of plants Fish, depending on water chemistry (e.g. increased zooplankton consumption in eutrophic conditions) Bivalves for uptake of natural radionuclides	

* e.g. conductivity, dissolved oxygen, temperature, nutrient status, pH

Table 2.3 Brief summary of key components: estuarine/marine ecosystems (based on Kennedy et al. (1990) and findings described in Section 2.3)

	Estuarine/Marine Habitats
Main source of radionuclides	Tidal and wind current driven Radionuclides become bound to particles in water column
Main factors influencing radionuclides accumulation	Mixing/circulation processes Salinity Temperature Bioavailability of radionuclides (depending on isotope chemical form) Particle size and composition of bottom sediments Availability of food-stuffs Seasonal variation Reproductive cycle
Components most likely to accumulate radionuclides	Bottom sediments are the main sink of radionuclides Particles in water column may travel very long distances
	Lobster (marine), macroalgae, mussels Actinide enriched in sea-spray (marine) Conservative elements (e.g. Cs, Sr, Tc, tritium) usually in solution, and bioavailable to pelagic and benthic feeders and seaweed Non-conservative elements (e.g. Pu, Am) usually bind with particles, and are ingested by benthic detrital feeders

Plant and animal life is continually exposed to natural low-level radiation from cosmic rays and cosmogenic and primordial radioactivity. Indeed, the incidence of radiation on plant and animal cells is one of the causes of genetic mutation and hence may play an important role in the evolutionary process. Many naturally occurring radionuclides are important contributors to the dose received by both humans and wildlife. They are present throughout the environment, although there may be technologically enhanced levels in areas affected by human activities. Few studies have, however, considered the impact of naturally occurring radionuclides on wildlife. This report therefore attempts to evaluate and place into context the information available on naturally occurring radionuclides.

2.3.1 Terrestrial ecosystems

- **Woodlands**

Woodlands are very effective accumulators of atmospheric radioactivity (Sokolov *et al.*, 1993; Sombre *et al.*, 1990; Tikhomirov, 1990). Under-storey plants and animals can be exposed to, and accumulate, high levels of radionuclides after the deposited radioactivity is redistributed from the tree canopy. Redistribution pathways can involve radionuclides being:

- washed out by rain or deposited in litter in the form of fallen leaves, etc;
- absorbed and translocated within the tree, leading to contamination of the wood;
- resuspended by wind, fire or evapo-transpiration;
- absorbed onto decomposed litter;
- transported into deeper soil layers via leaching processes;
- reabsorbed by the tree, or uptake into plants in the field or shrub layers;
- transferred to wildlife or aquatic ecosystems (after Kliashtorin *et al.*, 1994; Kirchmann *et al.*, 1993 and Copplestone *et al.*, 1999, 2000).

Research into the behaviour of radionuclides in ecosystems has focused on woodlands since the Chernobyl accident. Forest products such as timber and game are used extensively by humans, so much work on the quantification of radionuclide levels has been carried out in order to assess the risk to humans (Berg *et al.*, 1990; Kammerer *et al.*, 1994; Johanson *et al.*, 1994; Kiefer *et al.*, 1996; Strebl *et al.*, 1996).

Such studies have been extended to include the examination of forest compartments, such as: soil, leaf litter, invertebrates, and vegetation including the trees themselves (Toal, 1999; Strandberg, 1994; Melin *et al.*, 1994; Sombre *et al.*, 1994).

Most studies on the behaviour of radionuclides in the UK have been carried out on deciduous and coniferous woodlands.

Deciduous forests often have distinctive vegetation layers: canopy, lower tree, shrub, and moss layers. Each of these layers supports a diverse number of species of micro-organisms, plants and animals. Many of the organisms are adapted to tolerate a shaded environment. Deciduous trees display seasonality, with a four to six month growing season and a period of dormancy over winter, after shedding their leaves in the autumn.

Coniferous woodlands tend to grow well on dry, acidic sandy soils which are often poor in nutrients. As with deciduous woodlands they have distinctive vegetation layers including tree, shrub, and moss, each with its own characteristic species. A large proportion of UK coniferous woodland results from crop planting for timber production. As a result, the woodland canopy is often so dense that the lower vegetation layers are non-existent.

– Deposition studies

Deposition velocity (Vg) is defined as the deposition flux (to a unit area of land) divided by the air concentration (Harrison *et al.*, 1993; Tveten, 1990), and is used extensively in the models predicting the consequences of airborne releases of pollutants, both radioactive and non-radioactive.

Models on both the deposition and subsequent translocation of radionuclides through woodlands have been developed (Belli, 2000). The models and field studies have demonstrated that coniferous woodlands are far more effective accumulators of airborne particulates compared with deciduous woodlands. This results partly from the greater surface area of coniferous needles, leading to greater interception of particulates, and partly from the deciduous canopy being only present for part of the year. Studies have also demonstrated that deposition to woodlands is greater than adjacent grasslands (Belot *et al.*, 1994; Kirchmann *et al.*, 1993; Shaw *et al.*, 1994; Toal, unpublished data).

Several studies on oak and pine trees confirm that coniferous woodlands tend to retain radionuclides in the tree canopy for longer than deciduous trees (Sombre *et al.*, 1990). This extra retention time may allow radionuclides to be incorporated into plant tissues through foliar absorption. Experiments have also demonstrated that some radionuclides, for example ^{137}Cs, are more readily translocated through deciduous trees. Others, such as strontium (biologically mobile), tend to accumulate in older tissues along with the calcium and not be recycled (Myttenaere *et al.*, 1993).

The behaviour of radionuclides in forest soils is determined by a number of physico-chemical and biological properties as reviewed in Section 2.2.1. As already indicated, the majority of the radionuclides are held in the upper regions of the soil. This is important when estimating doses to biota, as the dose received by burrowing animals could be significantly reduced when the organism is deep underground and thus shielded by soil from the soil layer containing the highest levels of radionuclides (Copplestone *et al.*, 2000).

A few studies examined bioavailability of radionuclides *in situ* (Andolina and Guillitte, 1990; Thiry and Myttenaere, 1990). These indicated the potential for misinterpretation of activity concentrations in soil if expressed by weight. They suggest that there is a need to determine the chemistry of soil solution to better understand bioavailability and mobility of radionuclides in forest soils as it is those radionuclides in the soil solution which are most available for uptake into plants and biota.

– Uptake studies

Uptake of radionuclides by fungi, in both deciduous and coniferous woodlands and mainly following the Chernobyl accident, have been studied because of the potential pathway to humans. These studies demonstrated that the accumulation of ^{137}Cs can vary considerably between fruiting bodies of different species (Kirchner and Daillant, 1998; Toal *et al.*, in press; Guillitte *et al.*, 1987; Randa *et al.*, 1990; Barnett *et al.*, 1996; 1999). For example, the review by Gillet and Crout (2000) showed that concentration factors for ^{137}Cs vary between <0.001 and > 10 m^2kg^{-1} across all fungi species studied, and over three orders of magnitude for individual species (e.g. *Boletus badius*). Laboratory experiments have also investigated the uptake of ^{85}Sr and ^{134}Cs via direct contamination of mycelium and fruitbody caps, as well as via soil contamination for the saprophytic species, *Pleurotus eryngii*. The time for uptake and concentration in the fruitbody reflected the mode of contamination (Baeza *et al.*, 2000). The uptake of radionuclides into fungal fruitbodies is being investigated as a potential remediation agent to reclaim radioactive contaminated sites (Entry *et al.*, 1999).

Uptake of radionuclides into understorey herbs and grasses is generally low but dominated by the root uptake pathway, although external contamination of the vegetation can be important (Toal, 1999). Transfer to soil invertebrates, mammals and other wildlife species tends also to be low. For example, Copplestone *et al.* (1999; 2000) and Toal (1999) show concentration factors to small mammals relative to soil concentrations of around 0.1 and 0.0003 for ^{137}Cs and $^{239+240}$Pu respectively. Seasonal variation in the uptake was recorded and related to availability of food items. For example, Toal (1999) demonstrated the significance of fungal mediated transfer to small mammals when mice exhibited similar radionuclide concentrations to the soil in the autumn (around 1,000 $Bqkg^{-1}$ in both components). The high level was attributed to increased consumption of fungal fruit bodies containing ^{137}Cs (in excess of 2,000 Bq kg^{-1}).

– Accumulation studies

Coniferous woodlands often have a thick mat of leaf litter undergoing decomposition, which may take 3-5 years (Schell *et al.*, 1996) compared with six months for deciduous woodlands. Consequently,

radionuclides present in coniferous woodlands reside for long periods in the acidic mat of organic matter.

The accumulation of radionuclides in invertebrates is generally low, except for those involved in decomposition processes. Wood lice in particular have been shown to accumulate actinides (typical concentration factors of 0.3) (Copplestone *et al.*, 1999). A similar mechanism for the accumulation of heavy metals in wood lice is thought to occur (Hopkin and Martin, 1984).

Few studies on the behaviour of naturally occurring radionuclides in woodlands were identified. Thomas (2000) investigated the uptake of uranium, radium, and polonium, and found the highest concentrations at the plant-soil interface. The litter and top-soil concentrations were also reported to be higher than that of trees and deeper soil layers. Studies on natural (^{210}Po and ^{40}K) and artificial (^{137}Cs) radionuclides have been conducted on components of the Western Ghat tropical forest ecosystem prior to the development of nuclear power plants. The study concluded that epiphytic plant species could be used to monitor radionuclide concentrations (Somashekarappa *et al*, 1996).

Radon gas studies have been carried out on terrestrial ecosystems but mainly from the human perspective (e.g. Lugg and Roberts, 1997; Woodward, 1991; Becker *et al.*, 1993). These studies identified that an activity concentration in excess of 200 Bq m^{-3} of air is a health hazard to humans. Such levels usually only build up inside inadequately ventilated buildings, or in mine workings. It is possible that soil dwelling organisms may also be exposed to relatively high levels of radon gas.

Exposure rates from naturally occurring radionuclides have been studied. Selvasekarapandian *et al.*, (2000) estimated doses from natural radionuclides in soils from Udagamandalam district, India to be 0.743 μGy h^{-1}.

- **Sand dunes**

Sand dunes are sub-maritime habitats in the UK. They are not inundated by the sea nor are they strongly saline but they do receive material which is derived from, or has been in, seawater. Exposure to onshore winds provides a regular supply of sand particles, which are trapped by the vegetation. The type of sand will determine the flora of the dune system. Dunes are generally well drained as they have large quantities of coarse sand particles and the cation exchange capacity tends to be low. These factors mean that radionuclides are readily leached through the sands, resulting in their low bioavailability to plants. It has been demonstrated that sea to land transfer of radionuclides associated with sea spray results in the external contamination of vegetation (Nellis, 1990). Sand dunes may also receive an input of radionuclides through wet and dry deposition from authorised releases to atmosphere.

In the process of sea to land transfer, radionuclides become airborne due to bubble bursting at the sea surface. As the bubbles rise through the water column, they can scavenge particulate material and transport it to the surface. Radionuclides such as plutonium and americium are often associated with the water borne particulates, and so actinide concentrations are enhanced in the material deposited on land. The process means that a higher proportion of the actinides is returned to the dunes, compared with the generally more mobile ^{137}Cs. Copplestone (1996) demonstrated that this can lead to an increase in the concentration of $^{239+240}$Pu and ^{241}Am in herbivorous snails feeding on grasses.

Most studies on sand dune ecosystems have demonstrated that they do not accumulate radionuclides and as a result exposure of wildlife to ionising radiation is generally low (Nellis, 1990, Copplestone, 1996; Copplestone *et al.*, 2001).

- **Coastal grasslands**

Coastal grasslands form behind many sand dune systems. They may accumulate higher concentrations of radionuclides than the sand dunes because of their higher organic soil content.

Aerial deposition of radionuclides to grasslands is low especially compared with woodlands (see Section 2.4.1). As with woodlands, soil type is a dominant factor in determining the availability and transfer of radionuclides for uptake in to plants and higher organisms.

The use of fertilisers or soil amendments to adjust soil properties can expose plants to naturally occurring radionuclides. The use of bauxite mining residues for agriculture in Western Australia has been proposed in coastal grasslands. The bauxite has 1,000 Bq kg^{-1} and 300 Bq kg^{-1} of thorium and ^{226}Ra respectively, and studies have demonstrated that plant uptake can occur (Cooper, 1995). However, the uptake and transport mechanisms for naturally occurring radionuclides in plants have not been established, with most studies simply investigating the comparative uptake of different natural radionuclides in a range of species (e.g. Mortvedt, 1994; Kocher, 2000).

- **Saltmarshes**

Most UK saltmarshes are found on river estuaries, with some on open coasts, as in Norfolk. Plant species characteristic of saltmarshes can tolerate varying concentrations of saline waters. As mudflats are colonised by vegetation, the marshes start developing. Saltmarshes accumulate sediments over long time periods, and can act as sinks for radionuclides and other contaminants.

The accumulation and distribution of radionuclides within saltmarshes is largely the result of sedimentation processes. Radioncuclides tend to be associated with fine-grained sediment, which accumulate in saltmarsh areas, e.g. . ^{60}Co, ^{95}Zr, ^{95}Nb, ^{103}Ru, ^{106}Ru, ^{141}Ce, ^{144}Ce, ^{134}Cs, ^{137}Cs, ^{241}Am, ^{238}Pu, $^{239+240}Pu$ and ^{241}Pu originating from the Sellafield nuclear fuel reprocessing plant have been found in the Ravenglass estuary (UK) (Howard and Livens, 1991). However, due to their strong affinity for fine-grained sediment, actinides present in aquatic systems are effectively biologically unavailable (Copplestone, 1996).

The lack of post depositional mobility of certain radionuclides permits the study of radiometric dating of saltmarsh cores, with an evaluation of non-radioactive pollutants, in order to determine chronological profiles of historic discharges to an estuary (e.g, Fox *et al.*, 1999; Jones *et al.*, 1995; Brown *et al.*, 1999). More labile radionuclides such as ^{237}Np, ^{137}Cs and ^{99}Tc have also provided evidence of significant post depositional remobilisation occurring in the Ravenglass estuary saltmarshes (Morris *et al.*, 2000). Further work is required to confirm the mechanisms involved and the possible significance of the remobilisation for wildlife. It has been recognised, however, that contaminated sediments in the Ravenglass estuary could act as a secondary source for the transfer of radioactive material if the saltmarsh sediments are re-mobilised.

Vegetation contamination is mainly due to externally bound sediment, but with evidence of small root uptake for ^{137}Cs (Jones *et al.*, 1994). Temporal variation in vegetation is consistent with contamination by deposition of suspended sediment during tidal inundation, with a subsequent reduction in contamination on the vegetation with a half-life of between 20 and 30 days (Jones *et al.*, 1994).

Copplestone (1996) reported that detritivorous invertebrates, such as the wood louse, exhibited the highest levels of ^{137}Cs, $^{239+240}Pu$ and ^{241}Am as reported for grasslands and woodlands. Detritivores inhabit the strand line material where washed up plant material consistently exhibited the highest levels of radioactivity.

Spiders inhabiting a saltmarsh exhibited the highest concentrations of ^{137}Cs and this was attributed to their feeding method, involving the release releasing of enzymes into prey items to pre-digest the soft tissues where ^{137}Cs is known to accumulate (Copplestone, 1996). Copplestone (1996) demonstrated that the actinides, $^{239+240}Pu$ and ^{241}Am are not transferred into spiders in this way. Studies on the uptake and transfer of other radionuclides are required.

Animals grazing on saltmarshes may ingest radionuclides attached to silt and vegetation, as quantifiable levels were found in their tissues (Howard *et al.*, 1996). They will also be also exposed to higher levels of external radiation (Sanchez *et al.*, 1998; Copplestone *et al.*, 2000). Copplestone (1996) found highest ^{137}Cs concentrations in the insectivorous shrew compared with other small mammals (mice and voles), also due to the consumption of prey from the ^{137}Cs contaminated strand line material.

In these saltmarsh environments, the radionuclides are strongly associated with sediments resulting in external exposure to gamma and beta emitting radionuclides. Any food items, or accidentally ingested sediment, will contain relatively high levels of radionuclides but for most radionuclides these will pass

through the gut as they do not disassociate from the sediment very readily (even under the acidic conditions found in the gut). More biologically mobile radionuclides or those, which have already become incorporated into biological tissues of food items, show any increased uptake into the higher trophic levels. The concentration factors derived in Section 6.4 have been determined empirically from studies of actual accumulation into biota. These concentration factors therefore reflect the range of values observed under different circumstances through different pathways and therefore provide a realistic estimate of the likely uptake of radionuclides. It should be remembered however that this provides a generic approach and site specific characteristics may need to be included in any impact assessment of ionising radiation on wildlife.

2.3.2 Freshwater ecosystems

Freshwater ecosystems are of two main types - standing and running water. Standing water occurs as lakes and ponds which may be naturally occurring or man-made for water storage. Specific characteristics of individual lakes can vary depending upon their nature (depth, size etc), erosion characteristics of the catchment area and chemical composition of the water.

Running waters range from fast flowing mountain streams to sluggish lowland rivers. The velocities of the stream and river currents are dependent upon local geological characteristics such as slope and underlying rock and local rainfall patterns. The deposition rate of bottom sediment and the suspended sediment load are determined by these parameters and consequently affect the mobility of radionuclides within these ecosystems.

Radionuclides enter freshwater ecosystems mainly from the atmosphere by wet and dry deposition both directly into the water body but also through passage through catchments as a result of erosion processes. Some nuclear facilities may also discharge directly into freshwater ecosystems e.g. in the UK, the nuclear installation at Trawsfynydd (which is now being decommissioned).

Radionuclides deposited in bottom sediments, and dispersed in water, become an integral part of the biogeochemical cycles within the hydrosphere. During these cycles, the radionuclides can be distributed within different compartments of the ecosystem and can migrate, accumulate and transform their physicochemical forms.

Accumulation of radionuclides by aquatic organisms is dependent upon:

- the concentration of the stable element-analogue,
- temperature,
- pH,
- mineralisation of water,
- the physico-chemical form of the radionuclide, and
- the ecological and physiological parameters of organisms.

This leads to great variability in radionuclide accumulation factors in different water bodies (Kryshev and Sazykina, 1994).

Freshwater ecosystems have not been as fully investigated in the UK, or in northern Europe, as elsewhere in the world, e.g. Canada. The review will concentrate mainly on data relevant to the temperate climatic conditions of the UK, but will also include data drawn from outside Europe.

- **Radionuclide behaviour in sediments**

Radioecological field studies show that most of ^{110m}Ag occurs in contaminated sediments in freshwater ecosystems, representing a potential source of radioactive pollution (Garnier Laplace *et al.*, 1992; Hammar *et al.*, 1991). The same is true for plutonium, where sediments and their overlying organic floc were found to be the major sinks of plutonium in pond ecosystems (Emery and Klopfer, 1975). Studies of ^{137}Cs and ^{210}Po have reached similar conclusions (e.g. for Cs: Hammar *et al.*, 1991; Broberg and Andersson, 1991; e.g. for Po: Hameed *et al.*, 1997a; Shaheed *et al.*, 1997). There is further

evidence for accumulation of radionuclides in sediment. Less particle reactive radionuclides (e.g. ^{75}Se, ^{85}Sr, ^{134}Cs) have been shown to occur in the epilimnetic sediments whilst the more particle reactive radionuclides (e.g. ^{59}Fe, ^{60}Co, ^{65}Zn, ^{203}Hg) are more uniformly distributed in the bottom sediments (Hesslein, 1987; Bird *et al.* 1998).

- **Radionuclide behaviour in the water column**

Studies of Chernobyl-derived ^{137}Cs in Sweden have underlined the recent decline in concentrations of the radionuclide in freshwater. Water samples from the summer of 1986 gave ^{137}Cs concentrations of 0.6 – 1.1 Bq.l^{-1} (after filtration). By 1990, concentrations in unfiltered water had declined to 0.1 Bq.l^{-1} (Hammar *et al.*, 1991). ^{137}Cs collected in fine sediment traps accounted for only 0.1-0.7% of that deposited in the catchment. Concentrations in sediments declined from 1,000,000 Bq kg^{-1} (dry weight) in 1986 to 125, 000 Bq kg^{-1} in 1988 (Hammar *et al.*, 1991).

Smith (2000) used modelling techniques to demonstrate that Chernobyl derived ^{137}Cs was associated with fine particulates in the water column, which settled, and thereby transported the ^{137}Cs to the sediment.

– Mobility studies

The mobility of ^{137}Cs, $^{239+240}$Pu and ^{210}Pb has been assessed in lake sediments by Crusius and Anderson (1995). ^{137}Cs was found to be present in sediments in two forms – 67-82% as an immobile form and 18-37% reversibly adsorbed onto the sediments. Mobility of ^{137}Cs can be enhanced by a low clay content and high porosity in sediments, in a similar manner to that observed in terrestrial ecosystems. $^{239+240}$Pu was found to be significantly less mobile in sediments than ^{137}Cs. No ^{210}Pb mobility was observed (Crusius and Anderson, 1995). Murdock *et al.* (1993) demonstrated in a freshwater stream that the percentage of clay and silt in the sediment was inversely proportional to the flow rate. ^{137}Cs concentration was proportional to the clay and silt contents.

In India, the distribution of natural ^{210}Po has been studied in water, sediments and biota (Shaheed *et al.*, 1997). Significant differences were found between ^{210}Po concentrations in running (0.77 mBq.l^{-1}) and impounded water (1.27 mBq.l^{-1}). Higher ^{210}Po concentrations in impounded water were due to additional aerial inputs, accumulation in rich silt and organic matter, and increased biological production (Shaheed *et al.*, 1997). The study of ^{210}Po concentrations in biota of impounded water also showed a higher level of ^{210}Po in soft tissues than in hard parts, such as the shell or bones (Shaheed *et al.*, 1997).

– Accumulation studies

Many radionuclides accumulate in the sediment around the root stock of aquatic plants (Hameed *et al.*, 1997). Shaheed *et al.* (1997) reported ^{210}Po concentrations of 2 – 10 Bq kg^{-1} in aquatic weeds and 19 – 28 Bq kg^{-1} in phytoplankton. Under eutrophic conditions, fish may consume more phytoplankton compared with their other dietary components and as a consequence, can lead to greater exposure of fish via intake of plankton (Bird *et al.*, 1998). In addition, Co, Cs, Hg, Se and Zn are known to become highly concentrated in algae (Reynolds and Hamilton Taylor, 1992, Hamilton-Taylor *et al.*, 1996). Most of the post-Chernobyl ^{137}Cs in fish taken from Lake Zurich was derived from consumption of algae (Santschi *et al.*, 1990).

Wide ranges of ^{210}Po concentrations have been measured in a few species of bivalve molluscs, e.g.. 57 – 106 Bq kg^{-1} (Shaheed *et al.*, 1997). It has therefore been suggested that they would be suitable biomonitors of ^{210}Po for freshwater (Shaheed *et al.*, 1997; Hameed *et al.*, 1997b). Gastropod mollusc ^{210}Po concentrations were 32 – 46 Bqkg^{-1}, and in prawns 12 – 19 Bqkg^{-1} (Shaheed *et al.*, 1997). Bioaccumulation of naturally occurring radium and thorium has also been reported in the bivalve mollusc, *Lamellidens marginalis*, by Hameed *et al.* (1996).

Radionuclide concentrations in fish reflect:

- biological parameters such as trophic level, feeding habits (including particulate ingestion with food), location, and fish physiology;

- physicochemical parameters such as pH, temperature and water chemistry e.g. the concentration of Ca is important in the uptake of ^{226}Ra, as the latter is an analogue of the former and can be incorporated into bones in the same way (Clulow *et al.*, 1998).

Concentrations of ^{210}Po in fish have been reported to be in the order of 2 – 4 Bq kg^{-1} (Shaheed *et al.*, 1997). Concentrations of ^{210}Po generally tend to be relatively higher in the digestive organs of fish than in the muscle tissue (Skwarzec and Falowski (1988). Gut and bone tissues were also found to be the highest accumulators of ^{226}Ra, ^{228}Th, U, and ^{210}Pb where the main uptake route is ingestion or direct uptake from the water, e.g. via the gills (Waite *et al.*, 1988; Clulow *et al.*, 1998).

The impact of Chernobyl-derived ^{137}Cs on lake ecosystems has been studied in northern Sweden. Uptake by Arctic char and brown trout was enhanced by the consumption of zooplankton, *Mysis relicta*, which accumulated ^{137}Cs. In addition, there was post-deposition mobilisation via benthic organisms to fish in successive years after the introduction (Hammar *et al.*, 1991).

The radioactive content of two turtle species (the pond slider, *Trachemys scripta* and the common snapping turtle, *Chelydra serpentina*) was investigated to determine their utility as possible bio-indicator species for radioactive contamination (and also in support of the use of biomarker techniques (Appendix 2)) as well as other non-radioactive contaminants (Meyersschone *et al.*, 1993).

2.3.3 Estuarine and marine ecosystems

Radionuclides discharged into estuarine and marine environments are dispersed by tidal and wind-driven currents, as well as by diffusion. They may interact with sediments, be transported on the suspended phase, then be deposited on the seabed or intertidal areas. Radionuclides can enter the aquatic foodchain by being dissolved in seawater or attached to sediments.

Beks (2000) estimated the inventory of radionuclides in North Sea sediments to be in the order of 2.8 Tbq ^{238}Pu; 75 TBq $^{239+240}$Pu; 730 TBq ^{137}Cs; and 40 TBq ^{241}Am. Using ^{238}Pu/$^{239+240}$Pu ratios in sediments it has been concluded that nuclear fuel reprocessing at Sellafield and Cap de la Hague have been the main contributors of plutonium to the North and Irish Sea. Beks (2000) estimated that approximately 7% of all Sellafield discharged plutonium is stored in North Sea sediments. Of all the ^{137}Cs transported through the North Sea, about 2% is stored in the sediment. ^{241}Am is scavenged faster than plutonium, and is probably derived from ^{241}Pu transported to the North Sea (Beks, 2000).

Kershaw and Baxter (1995) showed that soluble plutonium from Sellafield can travel to Arctic waters. ^{99}Tc may take 2.5 years to reach the North Sea and Norwegian waters from discharges into the Irish Sea (Brown *et al.*, 1999; Kershaw *et al.*, 1999), compared with just over 4 years for Cs to reach the southern Norwegian Sea (Wedekind *et al.*, 1997), 3 years to the North Sea and 4 years to Norwegian waters (Dahlgaard, 1995).

Naturally occurring radionuclides tend to be associated with sediments, and Strezov *et al* (1998) showed that the accumulation of artificial and natural radionuclides was dependent on the nature of the sediment, particularly on the silt content.

Feng *et al.* (1999) demonstrated a relationship between thorium isotopes, ^{7}Be, and sediment in the Hudson river estuary; ^{7}Be entered the water directly from the atmosphere before becoming associated with sediment, whilst thorium isotopes were produced from dissolved uranium parents present in the water column where the concentration varied with salinity.

Cochran *et al.* (2000) found ^{137}Cs, plutonium isotopes, ^{237}Np and ^{129}I in water and sediment samples collected from the Ob River system in western Siberia. The sources were identified as tropospheric fallout from the former Soviet Union test site at Semipalatinsk, and reprocessing of spent fuel at Tomsk-7. The radionuclides were associated with suspended sediments.

Raisbeck and Yiou (1999) estimated that the ocean content of ^{129}I to be 100 kg before the nuclear age. ^{129}I levels in the ocean have increased by more than one order of magnitude due to anthropogenic activities. For example, discharges of ^{129}I from reprocessing plants at Sellafield (UK) and La Hague (France) have released 720 kg and 1640 kg respectively. Little information is available on the uptake

and behaviour of ^{129}I in the environment, even though iodine is an essential element required by organisms and is very biologically mobile.

– Accumulation studies

Investigations of radionuclide levels in aquatic biota have centred on the marine environment. Radionuclide concentrations in marine molluscs, crustaceans, fish and macroalgae have been measured in most of northern Europe. The majority of studies also concentrate on the uptake of radionuclides (mainly ^{99}Tc, ^{137}Cs, ^{239,240}Pu and ^{241}Am) by fish (e.g. Atlantic cod, *Gadhus morhua* and plaice, *Pleuronectes platessa*); European lobster (*Homarus gammarus*); winkles (*Littorina littorea*); mussels (*Mytilus edulis*); and brown seaweed (*Fucus vesiculosus*)

Rheinfelder and Fisher (1991) found a low assimilation of ^{241}Am by ingestion in zooplankton, due to short gut residence times and preference for the absorption of soluble material (Rheinfelder and Fisher, 1991).

The metabolism of macroalgae *Fucus vesiculosus* preferentially accumulates, and loses, ^{99}Tc from different parts of the plant. Over time, the radionuclide integrates into older parts of the plant (Masson *et al.* 1995). ^{99}Tc in *Fucus* can quickly reach equilibrium with water – recorded times are in the order of a few hours (Busby, 1998). As the dissolved Cs is also easily accumulated in *Fucus*, the macroalgae is considered an excellent bio-indicator for Cs radionuclides (Masson *et al.*, 1989; Carlson and Erlandsson, 1991).

Cockles feed on phytoplankton, zooplankton and organic detritus when submerged. Accumulation of radionuclides is therefore by ingestion, and dependent upon the food-stuffs. Cockles are poor accumulators of ^{99}Tc and Cs, and are not particularly efficient at retaining ^{239,240}Pu and ^{241}Am. Soft tissue uptake of ^{241}Am is more significant than that of other radionuclides.

Mussel tissues exhibiting the highest potential for bio-accumulation are: the viscera, gill, periostracum and byssal threads (McDonald *et al.*, 1993). Winkles do not display a consistent pattern of bioaccumulation. Gamma emitting radionuclides (^{40}K, ^{106}Ru, ^{137}Cs, ^{95}Nb, ^{210}Po) were detectable throughout mussels and winkles as a whole, but ^{95}Zr, ^{103}Ru, ^{144}Ce and ^{241}Am were found only in the viscera (McDonald *et al.* 1993). Investigations of ^{210}Po in mussels showed that the concentration was extremely variable (Ryan *et al.*, 1999). This variability was not linked to seasonal parameters or to the activity concentration in the water column or suspended sediment (Ryan *et al.*, 1999). ^{137}Cs in mussels have been shown to exhibit seasonal variation linked to the mussel reproductive cycle (Charmasson *et al.*, 1999).

Lobster accumulate ^{99}Tc via ingestion of seawater or food, and the hepato-pancreas appears to be a primary sink for the radionuclide (Busby, 1998). Hepato-pancreas is also known to accumulate other elements, such as metals. The biological half-life of ^{99}Tc in the adult lobster has been reported at around 51 days (Smith *et al.*, 1998; Knowles *et al.* 1998). In contrast, ^{137}Cs is present throughout the lobster body, with highest concentrations in the soft tissue. Caesium appears to behave as potassium in physiological processes, and displays a similar distribution at the sub-cellular level (Durand *et al.* 1994). Long-term monitoring has shown a significant decline in the concentration of ^{137}Cs in lobsters (wet weight) in the Irish Sea, from 81 Bqkg^{-1} in 1985 to 3 Bqkg^{-1} in 1998 (BNFL, 1999).

The uptake of ^{99}Tc by marine fish is suggested to be generally low from both laboratory studies (e.g. Pentreath, 1981a,b) and environmental monitoring (e.g. MAFF, 1996). Concentration factors of 10 have been derived for marine fish (Smith *et al.*, 1997; Brown *et al.*, 1998). Food is the main uptake route for ^{137}Cs in fish (Kasamatsu and Ishikawa, 1997). ^{239,240}Pu and ^{241}Am appear to be less important contributors to the radioactivity body burden of fish, as compared with crustaceans and molluscs (Vives I Battle, 1993). Although radiocaesium levels appear to be lower in shellfish than in fish, the opposite is true for transuranics.

There have been a few measurements of ^{137}Cs and Pu levels in large marine mammals. It has been demonstrated by comparing mammal flesh and fish concentrations of radionuclides that the levels in the mammals reflected the radionuclide concentrations in their fish diet. The results are, however, based on a small sample of seals and porpoises, and may not be representative of the populations as a whole (Watson *et al.*, 1999).

Consumption of fish leads to collective dose rates to the global human population of 160 manSv for ^{137}Cs and 30,000 manSv for ^{210}Po (Aarkrog *et al.* (1997). Thomas (2000) also highlighted that the dose from ^{210}Po as high in terrestrial ecosystems affected by discharges from a uranium mill, compared with other naturally occurring and artificial radionuclides.

2.3.4 Vulnerable ecosystems

It is well known that certain components of environmental pathways accumulate large amounts of specific radionuclides. Hence the critical group (i.e. most at risk) approach has been adopted for human radiological protection. In a similar way, recent studies have identified ecosystems that may be considered fragile or at risk from the presence of radioactive or non-radioactive pollutants (e.g. Barrie *et al.*, 1992; Howard, 2000). Examples of such vulnerable ecosystems include both terrestrial and aquatic environments in the Arctic and Antarctica.

Radioecological sensitivity analysis attempts to firstly integrate current knowledge on pathways and the spatial variation in radionuclide deposition, and secondly determine the transfer and hence radiation exposure in different areas. This will then identify areas that are at risk. The technique can be applied to both humans and non-human biota. The approach takes into account data and modelling uncertainties, and produces probability distributions for use in the models rather than single datum input values (Smith *et al.*, 1998).

The Arctic has been identified as a vulnerable area. A recent review by the Arctic Monitoring and Assessment Programme (AMAP) concludes that "parts of the arctic [human] population could be several hundred times more exposed than the average population of temperate areas" (Strand *et al.*, 1997). This could be due to high concentrations of radionuclides in food items from terrestrial and freshwater ecosystems. AMAP also identified that food products derived from semi-natural pathways were particularly important pathways for radionuclide uptake (Howard, 2000) because the species affected demonstrate high rates of accumulation. Under similar radionuclide concentrations, it can therefore be extrapolated that wildlife are likely to be more impacted in the Arctic compared with temperate regions, particularly due to the accumulation of radionuclides through food.

2.4 Summary

Radionuclides can enter ecosystems by many routes and become widely dispersed within their component parts. The behaviour of radionuclides in soil and sediment determines the impact of ionising radiation on biota in both terrestrial and aquatic ecosystems. Table 2.1 illustrates the components of an ecosystem most at risk of exposure which is usually related to where the radionuclides accumulate within an ecosystem (also given).

Some aspects of the behaviour of radionuclides in soils are still poorly understood, particularly with respect to chemical form and bioavailability for uptake (Desmet *et al.*, 1990). Furthermore, the role of micro-organisms in modifying bioavailability has been little studied. It is known that for example, several species of fungi can accumulate large concentrations of radionuclides (particularly ^{137}Cs) in their fruit bodies compared to the substrate, many of these fruit bodies are important food resources for higher organisms which can then take in higher concentrations of radionuclides than otherwise would be predicted. Toal (1999) demonstrated for example the effect of fungal mediated transfer of ^{137}Cs to mice, other micro-organisms may influence plant uptake by modifying the chemical form of the radionuclides etc. In terms of the impact assessment using current knowledge the fact that the concentration factors presented in Section 6.4 are derived from empirical measurements lends confidence that the influence of micro-organisms has been included in the assessment approach. However, site-specific issues may arise depending upon the ecosystems under assessment and this should be considered further by the assessor if required (refer to Section 6.5).

Most of the studies demonstrate that the transfer of radionuclides through successive trophic levels is limited, with ^{137}Cs and ^{90}Sr being the most biologically mobile. However, only a relatively small number of radionuclides have been studied in terms of their environmental behaviour. This is mainly because releases of particular some radionuclides are low, and/or because analytical techniques are difficult and costly (e.g. ^{129}I). This lack of information on specific radionuclides is a limitation in our

ability to understand and account for the risks associated with exposure to ionising radiation from particular sources.

When undertaking an impact assessment of exposure to ionising radiation, it is necessary to consider the importance of seasonal and spatial variation in radionuclide concentrations.

Data are sparse on the behaviour and pathways of naturally occurring radionuclides to wildlife, particularly for the terrestrial ecosystem. Most studies have investigated the impact of uranium mine discharges to aquatic ecosystems. Most information is available for ^{40}K, ^{210}Po, ^{226}Ra, ^{238}U and ^{232}Th and assesses the geochemistry rather than biological uptake. The uptake of ^{222}Rn has also been assessed but mainly from the human perspective.

Uptake of naturally occurring radionuclides can give rise to high concentrations within biota compared with anthropogenic ones. This is an area, which requires further research to establish the consequences and impact of natural exposure.

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3. Effects of Ionising Radiation on Biota

This Chapter summarises the reported effects of ionising radiation on both terrestrial and aquatic biota. The Chapter has been sub-divided to consider terrestrial and aquatic biota separately. This allows account to taken of the different physiology of aquatic and terrestrial biota, and the behaviour of radionuclides in these separate ecosystems.

The Chapter is based on the review conducted by UNSCEAR (1996), with the inclusion of data obtained from laboratory and field experiments and studies conducted following accidental releases such as Chernobyl (Appendix 1) published since then. The Chapter is not a complete review of the literature, but is representative of the data available to enable broad conclusions to be reached.

A number of endpoints have been considered including mortality, fertility, fecundity and genetic mutation as outlined by the FASSET working group and UNSCEAR (1996). New molecular and genetic techniques for measuring the effects of ionising radiation are under development and these are discussed in Appendix 2.

The text provides an overview of the effects observed. Summary Tables of reported effects of chronic and acute exposure to ionising radiation are provided at the end of the Chapter and should be referred to during the impact assessment process described in Chapter 6.

3.1 The interaction of radiation with biological material

Radiobiological research has demonstrated that there is a wide range of biological consequences of exposure to ionising radiation. Radiation can interact either directly or indirectly with biological structures, and the damage can be propagated to various levels of biological organisation i.e. from the molecule to cell, tissue, organ, individual, population, community, ecosystem etc. Initial damage results from the mode of action of ionising radiation at the molecular level, i.e. absorption of energy from the radiation (*via* ionisation[3]) may lead to dissociation of DNA molecules, with the effect dependent on the amount and type of radiation and the biological tissues exposed (Martin and Harbison, 1996). The dissociation of DNA molecules can lead to gene mutation in either somatic or germ cells. However, organisms contain mechanisms and processes with which to repair such damage. Consequently, there are a number of possible outcomes from damage:

- the damage may be repaired and the cell will survive and function normally;
- the damage may be mis-repaired, giving latent damage that may be expressed in the cell or its progeny;
- the damage may kill the cell or cause it to die (apoptosis).

More generally, these effects can be described as:

a) Stochastic effects

Stochastic effects are those in which the probability but not the severity of the effect increases as the radiation dose increases. An example of this is cancer induction in the exposed individual. If the cell affected is involved in reproduction, the damage may be transmitted to offspring leading to hereditary effects.

There is a consensus of opinion that stochastic effects, other than heritable genetic damage, are likely to be of little relevance to non-human biota (IAEA, 2000). Although there is some evidence of tumour formation in some wild animals it is generally reported that these are the result of exposure to other anthropogenic carcinogens and not radiation (IAEA, 2000). Furthermore, there is a general consensus (Chapter 6) that environmental protection criteria should be based on the population rather than the individual for non-human species. This makes non-stochastic effects likely to be more significant in these studies. Little, if any, research has been carried out on the significance of stochastic effects in populations of long-lived species.

[3] The process by which a neutral *atom* or *molecule* acquires or loses an electric charge.

Long-lived marine species in particular may be exposed to a wide range of environmental pollutants during their life and are the most at risk of exhibiting stochastic effects. Further work is required to assess this.

b) Non-stochastic or deterministic effects

Deterministic effects are those in which an effective threshold dose exists below which no observable effects arise. At doses above the threshold the severity of the effect is directly related to the radiation dose. An example of a deterministic effect is cell death within an organ of the body. Below the threshold dose the proportion of cells damaged will be insufficient to affect organ function and so no observable effect on the organ, or organism, as a whole will arise. However, above the threshold dose the number of cells dying will be sufficiently large that an effect on the organ and possibly organism will be observed. Above the threshold, cell death, and thus the severity of the effect, will increase in proportion to the dose received.

Deterministic effects include changes in morphology, physiology, biochemistry, fecundity (through life shortening, reduced fertility or reproductive ability), population (size, composition and succession), primary production and immune competence (IAEA, 2000). The main problem for impact assessment is that by the time most deterministic effects are observed, the population may have already received an unacceptable level of damage from the exposure to individuals, and be significantly harmed.

Much data exist on the impact of ionising radiation on biota in terms of deterministic effects, notably reproduction. It is possible that a series of mechanistic endpoints, relating to either stochastic or deterministic outcomes, may be of use in demonstrating radiation induced damage and thus providing an early warning system. Such an approach may look for chromosomal aberrations, mutations in specific gene markers, or biochemical changes within the cell. This is discussed further in Appendix 2.

3.2 General considerations

Most of the research into the effects of ionising radiation on wildlife has focussed on the impact to individuals, rather than populations, and many of these studies have looked at acute rather than chronic exposure. Radioactive discharges to the environment generally result in low-level chronic exposure of individuals, thus chronic irradiation studies are considered to be the most useful in investigating impacts on biota. Woodhead (1993) suggested that the total accumulated dose necessary to cause death might be 2-10 times greater than the acute lethal dose, when considering chronic exposures over the whole life of the organisms. Acute radiation studies, and the calculation of LD_{50} doses, may be of use to crudely rank organisms in terms of radiosensitivity (Rose 1992): the lower LD_{50} value, the more radiosensitive the organism (Figure 3.1).

Radiation exposures are reported as either:

- dose rate (ì Gy h^{-1}): generally used in studies exposing biota to chronic radiation or
- total accumulated dose (Gy): used for reporting acute exposure.

For example, exposure of young adult crickets to dose rates of 0.3 Gy h^{-1} and 2 Gy h^{-1}, leading to a total accumulated dose of 50 Gy induced 10 and 50% mortality respectively (UNSCEAR, 1996).

Laboratory research has frequently concentrated on external ã radiation rather than internal exposure to radionuclide mixtures. Radioactive effluent discharges often contain a mixture of radionuclides with differing half-lives, thus the dose received by individuals and/or populations at different times is complex and must be considered when investigating the potential impacts of ionising radiation.

Field observations from areas with significant anthropogenic contamination, usually following accidental releases such as at Chernobyl, have contributed significantly to our understanding of the impact of ionising radiation on biota. This information is reviewed in Appendix 1.

The effect of ionising radiation is most easily measured at the level of the individual. The impacts on individuals likely to be significant at a population level include:

- individual mortality (affecting death rate and population density);
- fertility (affecting birth rate thus population density);
- fecundity (production of viable offspring); and
- mutations.

There are limitations to this approach and care must be taken when extrapolating information, e.g. an observed effect that may be deleterious to an individual may have little impact on the population. However, if the survival of an individual can directly influence the success of the population, as is the case of threatened or endangered species, it is necessary to consider protection of individuals regardless (Suter *et al.*, 1995). Additionally, changes in non-lethal responses, such as growth rate can affect competitive ability, hence community structure. Therefore, although protection criteria may be set at population level, the measurements and/or assessments will have to be made at the level of the individual (Woodhead, 2000a).

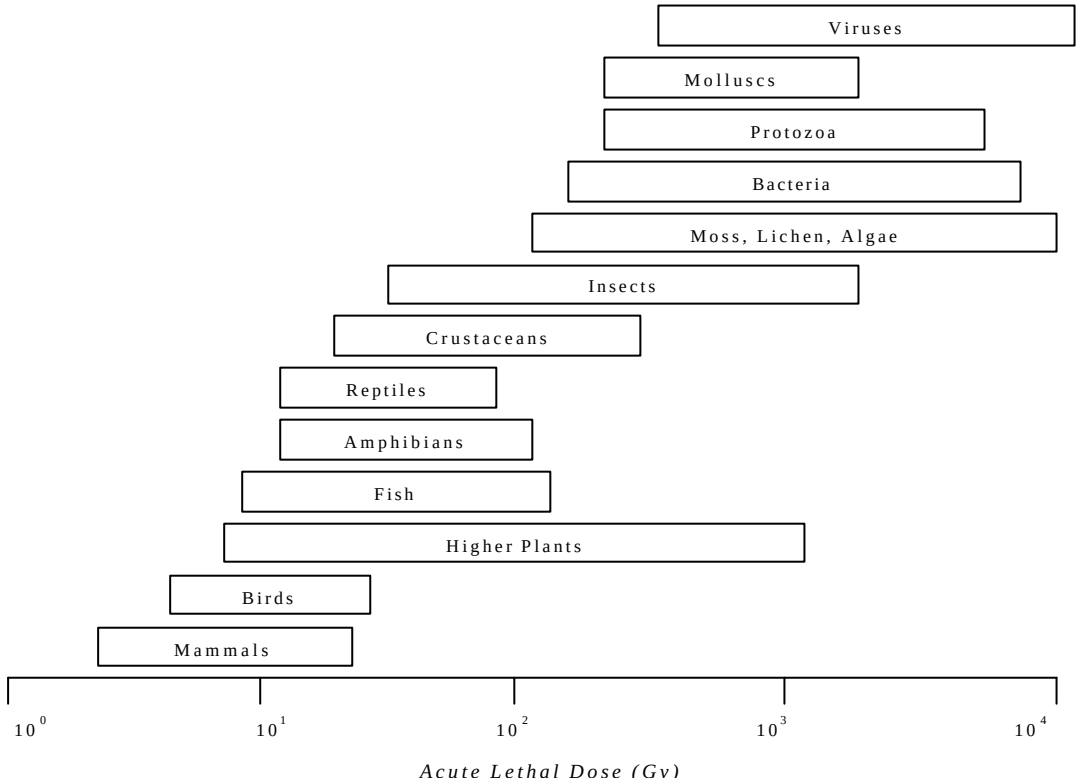

Figure 3.1 The range of acute lethal doses of ionising radiation for different organisms demonstrating their radiosensitivity (Blaylock et al., 1996)

3.3 Effects of radiation on biota

Over the last decade, there have been a number of extensive reviews of the impact of ionising radiation on wildlife. These include the International Atomic Energy Agency (IAEA, 1992) and United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR, 1996). Whicker (1997) and Woodhead (1998) have also reviewed the impact of radiation on plants and animals. Tables 3.6 to 3.19 provide an overview of the research conducted to date, and are located at the end of this Chapter. The text and summary Tables are based on the above named reviews, with additional data published since UNSCEAR (1996). They are constructed from both laboratory and field experiments, and are presented by taxa. In the laboratory experiments, exposure was usually to a ^{60}Co or ^{137}Cs source for the γ radiation, ^{3}H as β and ^{210}Po as α. These Tables can be referred to during the impact assessment process described in Chapter 6.

Rose (1992) and Blaylock et al. (1996) (Figure 3.1) have reported the acute lethal doses of ionising radiation on individuals of different taxa. More recently, Polikarpov (1998) proposed a conceptual model of chronic effects of ionising radiation on a community (Figure 3.2). In the model, Polikarpov reported that doses within an 'ecological masking zone' and 'damage zone to ecosystems' are significant for populations. The conceptual model does not however consider the difference in radiosensitivity between taxa (Figure 3.1) and so the doses presented in Figure 3.2 are meant to be indicative only.

Guideline dose limits for biota have been recommended by international organisations such as the IAEA (1992) (Table 3.1), below which significant effects are unlikely. A number of countries such as Canada and the USA have also suggested dose limits for biota (Table 3.1). The dose limits for biota recommended by the IAEA have generally been well received. The Environment Agency uses the IAEA guidelines when following its current assessment approach to determine the likely impact of exposure to ionising radiation from authorised discharges.

The IAEA guidelines in Table 3.1 are recommended by the authors for use in impact assessments, subject to periodic updates as some genetic and reproductive effects at dose rates below the guideline limits have been observed on mice, fish and aquatic invertebrates (Section 3.4.2 and Tables 3.11, 3.15, 3.17, and 3.19) although the relevance of these observations to population levels are uncertain. Consequently these guidelines may change in the future. The impact assessment approach described in this report develops the existing EA approach to provide a generic impact assessment. It is therefore important to recognise that the assessor must consider site specific features such as the presence of rare species when using generic guideline values given in Table 3.1 to evaluate the impact of ionising radiation on wildlife. In such instances generic guidelines should be used with caution and possible re-evaluation of the dose limits recommended within this report may be required. It must be recognised that the setting of standards to protect biota from ionising radiation must be decided by politicians/international organisations based on available scientific evidence.

Chapter 4 discusses the existing regulatory framework for protection of the environment from ionising radiation in a number of countries including Canada and the USA and the implications of the dose limits are considered further.

Table 3.1 Guideline and recommended dose limits (μGyh^{-1}) to biota

	NCRP, 1991	IAEA, 1988a and 1992	Thompson, 1999[1,2]	USA, Department of Energy[3]
Terrestrial				
Plants		*400*		*400*
Animals		*40*		*40*
Mammal			*10*	
Birds			*50*	
Amphibians/Reptiles			*10*	
Aquatic				
Freshwater organisms	*400*	*400*		*400*
Benthic invertebrates			*100*	
Fish			*50*	
Deep ocean organisms		*1000*		

1-calculated from annual 'critical' dose limits, which correspond to the lowest doses at which effects are observed. To incorporate a safety factor a 'no effects dose' has also been devised set at 1/10th of the corresponding critical dose. 2-Currently under public consultation in Canada. 3-Stephen Domotor pers. comm. IAEA Specialists Meeting on Protection of the Environment from the Effects of Ionising Radiation, International Perspectives, August 29-September 1, 2000.

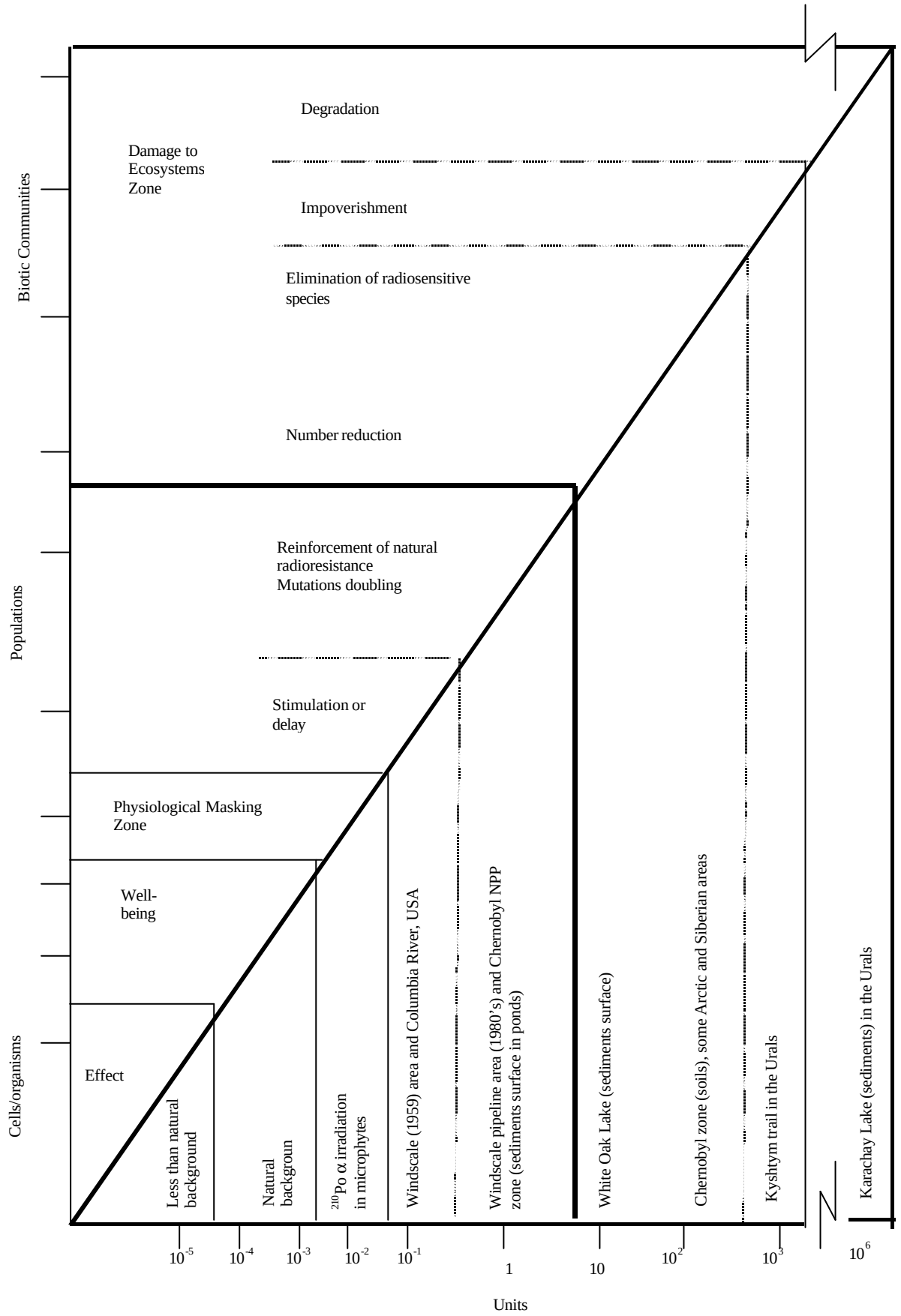

Figure 3.2 Dose and effect in the biosphere (from Jackson *et al*. (2000) based on Polikarpov (1998))

3.3.1 Terrestrial plants

The background dose rate to wild terrestrial plants is estimated to be between 0.1 and 0.7 μGy h^{-1} and 0.02 and 0.6 μGy h^{-1} from low and high linear energy transfer (LET) type radiations respectively (Table 3.2). The estimated dose rate to plants from contamination resulting in a dose of 1 mSv y^{-1} to a human residing in the same environment is 5.4 μGy h^{-1} from ^{137}Cs (Table 3.3).

Table 3.2 Dose rates (mGyh^{-1}) to wild terrestrial and freshwater organisms from the natural background, from Woodhead (1998), based on the reviews of IAEA (1976) and UNSCEAR (1996)

Organism	*Radiation LET type*	*Cosmic radiation*	*External radionuclides*	*Internal radionuclides*	*Total*
			Terrestrial		
Plants	Low	0.032	0.008 to 0.34	0.050 to 0.24	0.09 to 0.71
	High	<0.001	NA	0.020 to 0.56	0.02 to 0.56
Mammals	Low	0.032	0.008 to 0.089	0.02	0.06 to 0.14
	High	<0.001	NA	0.010 to 0.44	0.01 to 0.44
			Freshwater		
Phytoplankton	Low	0.027	< 0.001 to 0.009	ND	0.032 to 0.041
	High	<0.001	<0.001 to 0.053	ND	<0.001 to 0.053
Zooplankton	Low	0.027	< 0.001 to 0.009	ND	0.032 to 0.041
	High	<0.001	NA	ND	<0.001
Benthic organisms	Low	0.022	0.015 to 0.16	ND	0.047 to 0.18
	High	<0.001	NA	ND	<0.001
Fish	Low	0.022	< 0.001 to 0.007	0.04	0.022 to 0.065
	High	<0.001	NA	<0.001 to 0.01	<0.001 to 0.013

NA-Not applicable ND-No data.

A number of reviews (e.g. UNSCEAR, 1996) have concluded that sensitivity to acute and chronic radiation differs between species, with plant radiosensitivity decreasing in the order:

coniferous trees > deciduous trees > shrubs > herbaceous plants > lichen, bryophytes and fungi

In general the 'above ground' part of plants receive the greatest doses of radiation from atmospheric deposition. This is significant, as 70-80% of the total radioactive material released from Chernobyl was deposited on forest stands.

Radiation induced injury is expressed in plants as: abnormal shape or appearance (morphology changes), reduced growth, vigour and yield, loss of reproductive capacity, and death at high exposures (UNSCEAR, 1996). Tables 3.6 and 3.7 summarise the reported effects of chronic and acute irradiation on plants.

Table 3.3 Estimated dose rates to organisms from controlled discharges of radionuclides that would result in a dose of 1mSv y^{-1} to a man residing in the same environment (UNSCEAR (1996), based on IAEA (1992) and NCRP (1991))

Radionuclide	*Dose rate (mGy h^{-1})*		
	Plants[a]	Animals[a,b]	Fish[c]
^{3}H	5.8	5.8	0.59
^{14}C	18	11	
^{32}P	32	28	4.8
^{60}Co			0.53
^{90}Sr	2.0	0.042	67
^{95}Zr	38	2.0	
^{99}Tc			3.8
^{131}I	1.2	0.058	
^{137}Cs	5.4	3.1	0.72
^{226}Ra			3.6
^{235}U			2.6
^{238}U			4.7
^{239}Pu	0.023	0.00055	0.49
^{241}Am			0.71

a Discharges to atmosphere
b Domestic sheep
c Discharges to water (lakes)

- **Mortality**

– Chronic

Chronic doses of radiation in excess of 40,000 μGy h^{-1} over many months are required to induce non-stochastic effects in higher plants (Kennedy *et al.*, 1990). The cause of death of chronically irradiated pines is considered to result from damage to needles (Woodhead, 1998). Chronic dose rates less than 400 μGy h^{-1} should have only slight effects on sensitive plants, such as pine, and are unlikely to significantly affect mortality (UNSCEAR, 1996). The dose rate threshold for effects on lichen community composition is 125,000 μGy h^{-1}, although species densities are modified at dose rates below this threshold (Woodwell and Gannutz (1967) cited in UNSCEAR 1996; Brodo (1964) cited in UNSCEAR, 1996).

– Acute

Acute lethal doses to plants range from 10-1,000 Gy and for some lower plants, such as mosses and lichens, the upper limit may be 10 times greater (Woodwell and Whittaker (1968) cited in UNSCEAR, 1996). Reports on the impact on lower plants are sparse. Developmental stages of the plant have different radiosensitivities, with seeds being less radiosensitive than, for example, reproductive cells. LD_{50} ranges of 5-63 Gy have been reported for dormant pine seeds compared with 4.6-16 Gy for the vegetative phase of pines (Sarapultsev and Geraskoin (1993) cited in UNSCEAR, 1996). Table 3.4 highlights the greater radiosensitivity of higher plant communities.

Table 3.4 Dose levels from short-term irradiation (30 days) producing damage to plant communities (from UNSCEAR (1996), based on Whicker and Schultz (1982))

Plant community	*Dose range to produce effects (Gy)*		
	Minor effects[a]	Intermediate effects[b]	Severe effects[c]
Coniferous forest	1-10	10-20	>20
Deciduous forest	10-100	50-350	>100
Shrubland	10-50	50-200	>200
Tropical rain forest	40-100	100-400	>400
Rock outcrop	80-100	100-400	>400
Old field	30-100	100-1,000	>1,000
Herbaceous forest understorey	200-400	400-600	>600
Grassland	80-100	100-1,000	>1,000
Herbaceous invaders	400-600	600-1,000	>1,600
Moss lichen	100-1,000	500-5,000	>2,000

a. Minor effects including changes in productivity, reproduction and phenology. Recovery occurs rapidly after irradiation. b. Changes in species composition and diversity through selective mortality. Recovery may require from one to several generations. c. Drastic changes in species composition and mortality of most higher plant species. Recovery may be slow (years to decades or more).

– Recent studies

Chronic irradiation of a Boreal forest over 14 years induced tree death and modified the forest structure, with dose rates of 25,000 μGy h^{-1} (Amiro and Sheppard, 1994). The most sensitive species was the black spruce, followed by birch. Dose rates of 500-2,000 μGy h^{-1} induced death of some conifers but little change to the population (Amiro and Sheppard, 1994). Herbaceous plants were less radiosensitive thriving at doses up to 65,000 μGy h^{-1} (Amiro and Sheppard, 1994).

Whicker (1997) reviewed literature on the impact of ionising radiation arising from the Kyshtym and Chernobyl accidents. Chronic irradiation at dose rates of 2,000 μGy h^{-1} induced mortality of pines, whilst dose rates greater than 125,000 μGy h^{-1} were required to induce complete mortality of higher plants. Acute radiation at 20-100 Gy induced severe mortality of pines, while doses greater than 200 Gy induced mortality of deciduous trees and 700 Gy induced damage to the herbaceous community.

These studies demonstrate the need for dose rates in excess of the IAEA recommendation, and thus a dose limit of 400 μGy h^{-1} will ensure protection of terrestrial plant populations.

- **Growth and morphology**

– Chronic

Chronic doses greater than 1,000 μGy h^{-1} reduced the photosynthetic capacity of pines resulting in modified leaf morphology, which lead to reduced growth and delayed maturation (Bostrack and Sparrow 1970 cited in Woodhead, 1998).

Herbaceous species are less radiosensitive than pine species, with dose rates of 20,000 to 125,000 μGy h^{-1} required to inhibit growth and induce anomalies of herbaceous species (Woodwell and Oosting (1965) cited in UNSCEAR, 1996).

– Acute

Studies with herbaceous species indicate 40-50% and 25-35% of the lethal dose to inhibit growth and seed setting respectively. Normal appearance is maintained at less than 10% of the lethal dose (Sparrow and Woodwell (1963) cited in UNSCEAR, 1996).

- **Genetics**

– Chronic

Exposure of *Crepis tectorium* (Hawkshead) to β radiation at dose rates of 150 μGy h^{-1} increased the occurrence of chromosome aberrations (Abramov and Shevchenko 1987), but no effect on the population was reported.

– Acute

The reproductive organs of female (seed) and male (pollen) plant species are the most radiosensitive. Pollen exposed to ã radiation at 1 Gy led to three times as many chromosome aberrations as in the normal situation (Taskayev *et al.* (1992) cited in UNSCEAR, 1996).

– Recent studies

Studies conducted 8 years after the Chernobyl accident reported that the seeds of birches and pines in contaminated regions to have higher than normal rates of chromosome aberrations (Cherezhanova, 1998). The mutation rates in herbaceous seeds collected from polluted areas were also elevated compared with control plots as reflected by a reduced germinating capacity (Shevchenko, 1998).

- **Radioadaptation**

Exposure of birch buds and herbaceous seeds collected from contaminated sites (3700 kBq/m) within the Southern Urals, with acute doses of 100 and 150 Gy, suggest that adaptation to ionising radiation may arise. Acute irradiation induced lower abnormality rates in buds collected from contaminated sites than those from control sites (Cherezhanova, 1998). This higher resistance of plants growing in polluted areas may be attributed to the selection of less radiosensitive buds (Cherezhanova, 1998), possibly reducing genetic diversity resulting in an adverse effect, but further investigations are required.

- **Protection of communities**

Ecological dose limits (EDL) have been derived in an attempt to provide protection of plant communities (Table 3.5). They are calculated using the radiosensitivity of the dominant species as the best indicator of the overall sensitivity of a plant community. Net primary productivity of the ecosystem, not mortality, is used as the criterion of significant radiation effects. 1,100 ì Gyh^{-1} is proposed as the ecological dose rate limit that would guarantee environmental protection (Romanov and Spirin 1991, cited in Woodhead, 1998).

Table 3.5 Ecological Dose Limits (EDL) for typical ecosystems of the Northern Hemisphere (Romanov and Spirin, 1991)

Ecosystem	*Dominant*	*EDL (Gy)*
Coniferous forest	Standing timber	20-40
Deciduous forest	Standing timber	300-400
Herbaceous (meadow, steppe, Waste land, fallow land).	Mixed grass	400-500
Agricultural Crops	Monoculture of agricultural crops	50-60
Cultured pasture	Sown perennial herbs	80-100
Freshwater	Phytoplankton	300-500

3.3.2 Soil fauna and invertebrates

There are lack of published data concerning the impact of ionising radiation on soil fauna. Some data on the effect of β irradiation on soil invertebrates have been reported following the Kyshtym accident, and it is generally concluded that adult insects are relatively resistant to the effects of radiation because little cell division and differentiation occurs.

Tables 3.8-3.9 summarise the reported effects of chronic and acute irradiation on invertebrates.

- **Chronic exposure**

– Population structure

Generally soil invertebrate numbers are only reduced at high dose rates, 20,000-70,000 μGy h^{-1}, but dose rates of 1,000 μGy h^{-1} have been reported to reduce earthworm numbers (Krivolutsky, 1987).

β radiation arising from contamination levels (165-340 mBqm^{-2} from ^{90}Sr, dose rate not given) in soil reduced the mesofuna population density by more than 50%. Saprophages (earthworms and millipedes) were the most severely affected group, possibly as a result of their relatively sedentary lifestyles rather than greater intrinsic radiosensitivity. Thirty years after the accident, the earthworm population had recovered but the number of juveniles was under-represented in the irradiated population (Krivolutsky *et al.*, 1993 cited in UNSCEAR, 1996). It was concluded that invertebrate species most likely to be affected were those whose early stages are spent in the leaf litter and surface soil.

– Indirect effects

Indirect effects of chronic ionising radiation can arise as a result of tree canopy modification and consequent reduction in available leaf litter (Poinsot-Balaguer *et al.*, 1991). Chronic irradiation of a mixed forest at dose rates up to 10,000 μGyh^{-1} reduced species diversity of arthropods and microbial biomass (Poinsot-Balaguer *at al.*, 1991). The disappearance of trees from the ecosystem as a result of irradiation was considered the primary cause of disturbance in the soil invertebrates.

- **Acute exposure**

- Mortality

LD_{50} values for adult insects range from 20-3,000 Gy, much higher than in mammals, birds, reptiles or amphibians (Woodhead, 1998; Figure 3.1), indicating lower radiosensitivity of invertebrates compared with more complex organisms. LD_{50}s for the developing stages of insects indicate that they are more radiosensitive than adults, with effects observed in weevil, wasp and fruit fly embryos at 1-2 Gy (Woodhead, 1998).

- Fecundity and Reproduction

A dose of 20 Gy delivered to earthworms during early embryogenesis reduced hatching success of embryos, whilst 20 Gy to mature adults affects hatchability of eggs laid post irradiation (Suzuki and Egami 1983 cited in UNSCEAR, 1996).

- **Recent studies**

No additional data were found.

3.3.3 Mammals

A large number of laboratory studies have been conducted investigating the impact of radiation on small mammals, and many have specifically investigated genetic damage. It is, however, difficult to extrapolate these data to assess the effects on mammals in the natural ecosystem because of the potential presence of other stressors. Much work in the field has been conducted since Chernobyl, investigating the impact of the fallout on mammals living within the 30 km exclusion zone, but the lack of dosimetry data makes it difficult to compare field investigations with laboratory results.

The background dose rates to wild terrestrial animals are estimated to be in the ranges 0.06-0.14 μGy h^{-1} and 0.01-0.44 μGy h^{-1} from low and high LET type radiation, respectively (Table 3.2). The estimated dose rate to animals that would result in a dose of 1 $mSvy^{-1}$ to a human residing in the same environment is 3.1 μGy h^{-1} from ^{137}Cs (Table 3.3).

Tables 3.10-3.11 summarise the reported effects of chronic and acute irradiation on mammals.

- **Mortality**

– Chronic

Chronic lifetime exposure of mice to low LET radiation at dose rates that do not induce bone marrow failure (at around 3,800 μGy h^{-1}) can reduce life expectancy in a linear manner for total accumulated doses of 0 to 45 Gy (UNSCEAR, 1996). UNSCEAR (1996) concluded that there is little evidence of any change in mortality rates at dose rates less than 400 μGy h^{-1}.

– Acute

LD_{50} for mammals range between 2 and 15 Gy (Woodhead, 1998), lower than that of all other phyla (Figure 3.1). The developmental stages of other phyla may however be more radiosensitive.

Rice and Baptiste (1974) have reported LD_{50}s for:

- Monkey 6 Gy
- Dog 2.5 Gy
- Pig 2.5 Gy
- Hamster 6 Gy
- Mouse 6.4 Gy
- Rabbit 7.5 Gy

The LD_{50} of a mouse embryo is lower at 1 Gy, demonstrating the greater radiosensitivity of developmental stages (Woodhead, 1998). No apparent effect at the organism level or short-term lethality (within 30 days) are observed below a threshold of about 1 Gy for acute exposure (UNSCEAR, 1996).

- **Fertility/Reproduction**

– Chronic

Effects on the reproductive system can arise at doses less than 10% of that which can induce direct mortality, with effects on reproduction evident when no other observable responses are apparent (UNSCEAR, 1996). This may be a consequence of the rapid cell division and differentiation, taking place during spermatogenesis or oogenesis.

A chronic dose rate of 180 μGy h^{-1} can induce sterility in beagle dogs within a few months. A chronic dose rate of 36 μGy h^{-1} over the whole life did not induce a response **(**Committee on Biological Effects of Ionising Radiation (1980) cited in UNSCEAR, 1996).

The IAEA (1992) concluded that dose rates less than 40 μGy h^{-1} are unlikely to exert any effect on reproductive capacity on mammals. Below 40 μGy h^{-1} modification of fertility, fecundity or the survival of offspring is unlikely (UNSCEAR, 1996). As a general rule, chronic effects on an individual begin at accumulated doses of 10% of the LD_{50} value, whilst effects on reproductive cells can occur at accumulated doses of 1% of the LD_{50} (Environment Canada, 2000).

– Acute

Males can recover from acute exposure as gametogenic stem cells are produced during the whole reproductive life, thus a proportion of stem cells are always in the less radiosensitive resting phase (as opposed to those undergoing division and differentiation into spermatocytes).

The effect of ionising radiation on the embryo is dependent upon the dose received and stage of development, with radiosensitivity decreasing with time after conception (Figure 3.3). Acute radiation during the pre-implantation stage of development (blastogenesis) will induce death of the embryo (reducing fecundity or viability of the offspring). Irradiation during organ formation (organogenesis) will induce malformations and exposure during the foetal period will induce teratogenetic [4] effects.

Exposure to 0.05 Gy during blastogenesis has been shown to induce the death of rat embryos whilst exposure to the same dose during the later developmental stage of organogensis may result in malformation of the offspring but not actual mortality (Woodhead, 1998). Dose rates less than 100 μGy h^{-1} are unlikely to exert any impact on the fecundity of populations as a whole (UNSCEAR, 1996). Mice are amongst the most sensitive species to the reproductive effects of radiation, with reproduction impaired by acute doses of 0.2 Gy for females and 3.2 Gy for males (IAEA, 1992).

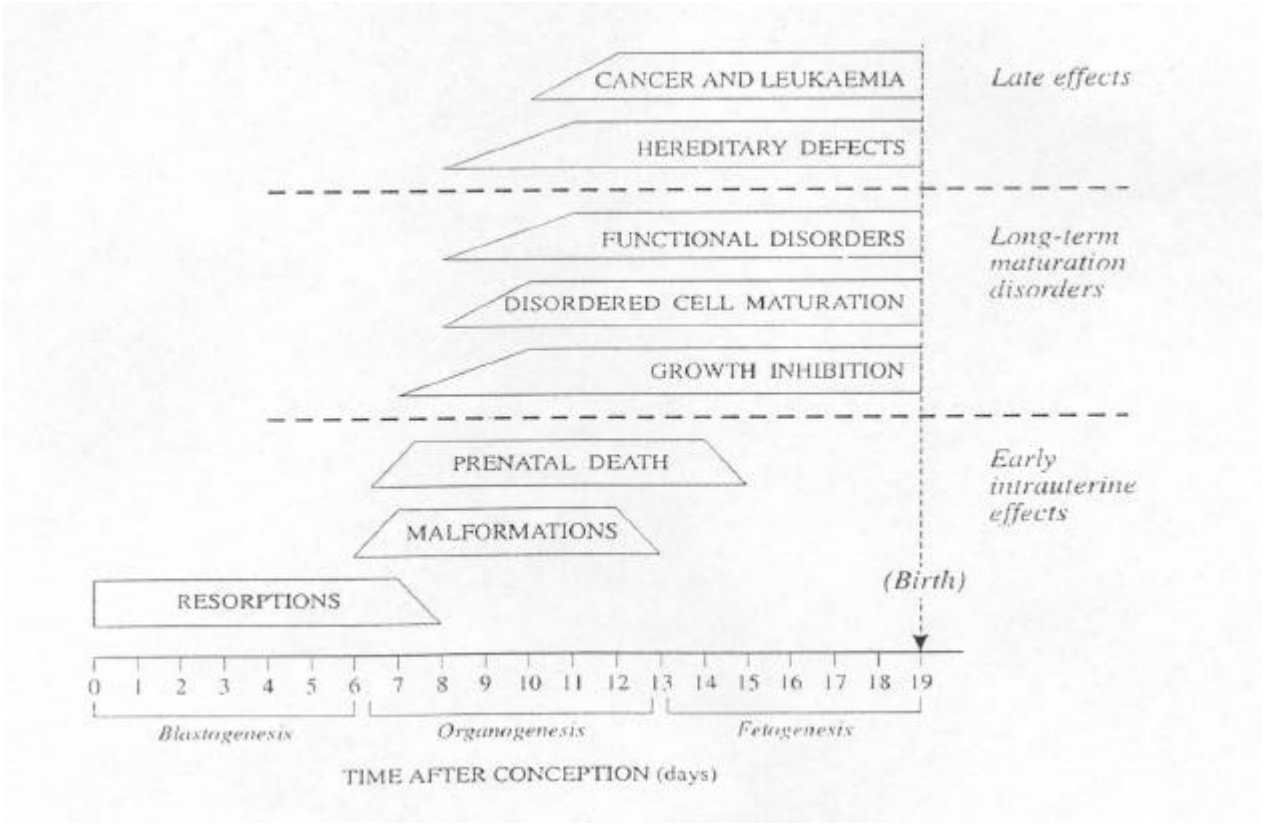

Figure 3.3 Prenatal phases for induction of radiation effects in the mouse (from Commission on Radiological Protection (1989) in UNSCEAR (1996))

– Recent studies

Late foetal irradiation of mice with ã radiation failed to induce foetal mortality or growth retardation of young at exposures below 0.25 Gy (Devi *et al.*, 1994). Exposure of paternal mice to a total accumulated ã dose of 2.8 Gy during spermatogenesis (at dose rate of 0.28 Gy $ħ^{1}$) did however increase mortality in embryos (Muller *et al.*, 1999).

[4] Embryo malformation

Dose rates required to induce significant changes in mammal fertility range from 23-30 μGy h^{-1} (Harrison and Anderson 1996), but the development stage of the gonads at the time of exposure influenced the response.

Acute irradiation of mice at a dose of 0.05 Gy was reported to induce perturbations in spermatogenesis of mice, although normality was restored 70 days after exposure (Jagetia *et al.*, 1995). Acute exposure (10-25 Gy) was sufficient to reduce mouse populations following the Chernobyl accident (Table A1.4, Appendix 1).

- **Genetic damage**

– Chronic

Lifetime chronic exposure (>400 μGy h^{-1}) of mammals is reported to reduce their lifespan mainly as a result of induction of malignant tumours, but this effect occurs in laboratory studies where no other stresses are present (e.g. predation) (UNSCEAR, 1996).

– Acute

Malformation of the central nervous system has been studied, with little evidence for any pathological response at doses of 0.1 Gy or less in mice, rats and primates (UNSCEAR, 1996).

– Recent studies

Radiation induced genetic damage of bank voles collected from around the Chernobyl NPP following the accident have been reported. Bank voles are generally considered to be one of the least radiosensitive rodent species. However, increases in chromosome aberrations (3-5 times greater than normal) were observed at sites with high levels of ^{137}Cs deposition (90 and 1,525 $kBqm^{-2}$) (dose rates not given) (Goncharova and Ryabokon, 1995).

- **Behaviour**

– Acute

Exposure of offspring *in-utero* can result in modifications in their development and behaviour. â radiation exposure (0.1 Gy) to mouse embryos during early pregnancy significantly altered behaviour, which may reduce viability in the natural environment (Wang *et al.*, 1996 cited in UNSCEAR, 1996).

– Recent studies

The lowest acute dose reported to induce sub-lethal effects, i.e. impaired offspring reflexes, on pregnant rats was 0.01 Gy (Rose 1992). *In-utero* exposure of mice to 0.3-1.5 Gy modified postnatal behaviour (reducing locomotor and exploratory activities), whilst emotional activities (such as rearing and grooming, did not change. A significant impairment of learning and memory functions was also induced at these doses (Devi *et al.*, 1999).

- **Exposure to mixed radionuclides**

Many experiments have been conducted to investigate the impact of ionising radiation from one radionuclide. A study was conducted over six years on the biological effects of a single injection of mixed radionuclides (^{239}Pu + ^{90}Sr), and of separate injections of the same radionuclides on sheep, with ^{239}Pu administered at 7,400 $Bqkg^{-1}$ and ^{90}Sr at 148,000 $Bqkg^{-1}$. Co-administration of the radionuclides resulted in greater radiation-induced effects (e.g. suppression of blood cell production), reduction in immunity, and shorter life span than that of separate injections. During the last six months of the experiment the sheep receiving ^{90}Sr alone were found to have normal leukocyte composition in the blood, whilst restoration of blood leukocyte composition was slower in sheep receiving co-administration of ^{239}Pu + ^{90}Sr (Martyushov, 1998). This study highlights the importance of considering multiple radionuclide releases.

3.3.4 Birds

There have been fewer irradiation studies on birds than mammals. This is partly due to the greater mobility of birds, making controlled field experiments more difficult. Differences in species biology, such as feeding habits and nesting behaviour, may determine species vulnerability to radiation. Kennedy *et al.*, (1990) reported that published data concerning controlled radiation exposure experiments were limited to acute high-level exposure. Few chronic irradiation studies are available (UNSCEAR, 1996). No studies have reported the minimum dose required to induce an effect (UNSCEAR, 1996).

Tables 3.12 and 3.13 summarise the reported effects of chronic and acute irradiation on birds.

- **Chronic exposure**

  Radiation chronic effects in birds are similar to those in small mammals (UNSCEAR ,1996).

- **Acute exposure**

  Birds are more radiosensitive to acute radiation than reptiles, amphibians and fish, but have similar sensitivity to mammals (with LD_{50} of from 5-20 Gy) (Woodhead, 1998; Figure 3.1). The development stages of birds are more radiosensitive than adults, with chick embryo having an LD_{50} of 7 Gy (Woodhead, 1998).

- **Recent studies**

  Field irradiation studies at ^{90}Sr contaminated (up to 5.55 x 10^7 $kBqm^{-2}$) sites of the Southern Urals have reported that modification of nesting success is species dependent. The nesting success of the Great Tit did not differ between the contaminated and control sites, whilst that of the Pied Flycatcher was significantly higher in the uncontaminated areas. This difference in nesting success was attributed to the nest composition. Pied Flycatcher nests predominantly consist of dried leaves with a higher level of radioactive contamination than moss, which makes up Great Tits nests. This highlights the importance of nesting habits in influencing external exposure of different bird species, and their subsequent impact on nesting success (Lebedeva, 1998).

3.3.5 Reptiles and amphibians

There have been few studies on the impact of ionising radiation on reptiles and amphibians. Kennedy *et al.* (1990) reported investigations on radiation doses necessary to induce damage. Furthermore, no studies were conducted on amphibians and reptiles in the Urals following the Kryshtym accident (Sokolov and Krivolutsky, 1998), and none are reported for the UK.

Tables 3.14 and 3.15 summarise the reported effects of chronic and acute irradiation on reptiles and amphibians.

- **Chronic exposure**

– Reproduction

  No experimental data exist concerning chronic exposure of amphibians to ionising radiation (UNSCEAR, 1996). However, studies post Chernobyl have reported reduced fertility of brown frogs collected close to the nuclear power plant (dose estimates not provided) (Cherdentsev *et al.* (1993) cited in UNSCEAR, 1996).

  Chronic exposure of a lizard species at 630 μGyh^{-1} induced sterility after 3.5 years, whilst exposure to 210 $\mu Gy\ h^{-1}$ induced sterility after 5.5 years (Turner *et al.*, 1971 cited UNSCEAR, 1996). The IAEA concluded that chronic doses of 400 $\mu Gy\ h^{-1}$ might impact on reproduction of some reptiles (IAEA, 1992).

– Genetic Damage

  In Belarus, β dose rates of 7 $\mu Gy\ h^{-1}$ have been reported to increase frog chromosome aberrations by between 2 and 10 times compared with pre-Chernobyl levels (Eliseyev *et al.*, 1990 cited in UNSCEAR, 1996).

- **Acute exposure**

  LD_{50} ranges of 10-40 Gy for reptiles and 7-50 Gy for amphibians have been reported (Woodhead 1998). The fertilised egg is the most radiosensitive stage for acute exposure in the frog, with an LD_{50} of 0.6 Gy (Panter, 1986 cited in Environment Canada 2000) compared with 25 Gy for the adult.

- **Recent studies**

  No additional data were found.

3.4 Effects of radiation on aquatic ecosystems

The most recent reviews into the effect of ionising radiation on aquatic organisms are; NCRP (1991), IAEA (1992), UNSCEAR (1996) and Woodhead (1998). The NCRP review (1991) is solely concerned with aquatic biota and reviewed the effects of acute and chronic radiation on mortality, morbidity, fertility, fecundity and hereditary mutations.

3.4.1 Marine mammals

No experimental data are available describing the impact of ionising radiation on aquatic mammals such as cetaceans and seals (Woodhead, 1998).

3.4.2 Fish

The natural background dose rates to fish are estimated at 0.022 - 0.065 $\mu Gy\ h^{-1}$ and <0.001 - 0.013 $\mu Gy\ h^{-1}$ from low and high LET type radiation respectively (Table 3.2). The estimated dose rate to fish that would result in a dose of 1 $mSvy^{-1}$ to a human residing in the same environment is 0.72 $\mu Gy\ h^{-1}$ from ^{137}Cs (Table 3.3).

Tables 3.16 and 3.17 summarise the reported effects of chronic and acute irradiation on fish.

- **Fertility/Reproduction**

– Chronic

  Endpoints associated with reproduction and the developing embryo generally show greater radiosensitivity than mortality, as reported for mammals (UNSCEAR, 1996). NCRP (1991) reported the following effects on fish reproduction, under laboratory conditions:

  - 1,600 $\mu Gy\ h^{-1}$ – modification of reproductive behaviour
  - 2,700 $\mu Gy\ h^{-1}$ – reduced fertility
  - 4,000 $\mu Gy\ h^{-1}$ – retarded gonadal development
  - 7,700 $\mu Gy\ h^{-1}$ – sterility of offspring .

  They concluded that significant effects in fish gonads due to chronic irradiation are unlikely to be observed at dose rates less than 1,000 $\mu Gy\ h^{-1}$.

– Recent studies

  Dose rates less than 100 $\mu Gy\ h^{-1}$ were reported to increase anomalies in fish reproductive systems (Kryshev and Sazykina 1998, see Table A1.8, Appendix 1). This effect is observed at a dose below the NCRP (1991) and IAEA (1992) guideline to protect freshwater organisms and may result from the interaction of radionuclides with other pollutants present in that water body. However, laboratory experiments exposing plaice to chronic gamma radiation at 240 $\mu Gy\ h^{-1}$ for 197 days (also below the NCRP and IAEA guideline) report significant reductions in plaice testis weight, being consequent with a reduction in the amount of sperm (Knowles, 1999). It was concluded that plaice testis are more radiosensitive than the more investigated tropical fish and of similar radiosensitivity to mammalian testis.

- **Mortality**

– Acute

It has been concluded that fish are the most radiosensitive of aquatic organisms to the lethal effects of acute radiation exposure (Woodhead 1998, UNSCEAR 1996), with LD_{50} of adults ranging from 7-60 Gy (Woodhead, 1998). Fish also require a longer period of exposure for acute mortality to be expressed (60-90 days) in contrast to mammals (30 days), thus fish can appear less radiosensitive than terrestrial mammals. As reported for other taxa radiosensitivity is dependent upon developmental stage, with the LD_{50} of fish embryos ranging from 0.16-25 Gy (Environment Canada, 2000).

– Recent Studies

Chronic dose rates around 1,000 μGy h^{-1} can induce mass death of fish, whilst a dose rate greater than 12,000,000 μGy h^{-1} was reported to induce total death of a lake ecosystem (Kryshev and Sazykina 1998, see Table A1.8 in Appendix 1). The 1,000 μGy h^{-1} is 2.5 times higher than the dose limit recommended for freshwater organisms by Woodhead (1998) (Table 3.1).

- **Genetic damage**

– Acute

Only limited investigations into the mutagenic effect of ionising radiation on aquatic organisms have been conducted (IAEA, 1992).

– Recent Studies

Exposure to gamma radiation at doses as low as 240 μGy h^{-1} for 197 days failed to induce genotoxic damage in plaice (Knowles, 1999).

3.4.3 Aquatic Invertebrates

Tables 3.18 and 3.19 summarise the reported effects of chronic and acute irradiation on invertebrates.

- **Mortality**

– Chronic

Dose rates less than 10,000 μGy h^{-1} are unlikely to influence the mortality of aquatic invertebrates (Woodhead, 1998). Chronic exposure to dose rates of between 10,000 and 30,000 μGy h^{-1} have little effect on the mortality of aquatic molluscs and crustaceans (e.g. snails, scallops, clams and blue crabs) (UNSCEAR, 1996).

– Acute

LD_{50} for acute radiation have been reported for the following species:

- Adult annelid – 100-500 Gy (Harrison and Anderson 1994a)
- Mollusca, early life stage – 11 Gy (Blaylock and Trabalka 1978)
- Mollusca adult – 50-5,000 Gy (Templeton *et al.*, 1971)
- Crustaceans, adult – 2-1,000 Gy (Chipman 1972, Engel *et al.*, 1974)

- **Fertility and fecundity**

– Chronic

Laboratory populations of the Polychaete worm, *Neanthes arenaceodentata* exposed to 17,000 μGy h^{-1} during their lifetime produced reduced numbers of embryos. A dose rate of 3,200 μGy h^{-1} reduced the percentage of live embryos and increased the numbers of abnormal embryos in the broods (Harrison and Anderson, 1994). Dose rates greater than 3,200 μGy h^{-1} affected reproductive performance of the polychaete *Ophryotrocha diadema*, observed as a decrease in the

numbers of egg sacs, eggs and larvae produced (Knowles and Greenwood, 1994). Chronic dose rates greater than 3,200 μGy h^{-1} also reduced reproductive capacity of the freshwater snail, *Physa heterostropha* (UNSCEAR, 1996).

– Recent studies

Exposure of *Ophryotrocha diadema* to β radiation at 7,300 μGyh^{-1} reduced the number of eggs surviving to the larval stage, but did not affect egg production. This is in contrast to previously reported effects of γ irradiation, where egg production is reduced but not the number becoming larvae (Knowles and Greenwood, 1997).

3.4.4 Aquatic plants

Few investigations have been conducted on the impact of ionising radiation on aquatic macrophytes (Woodhead, 1998).

Early work showed phytoplankton to be less radiosensitive to radiation exposure than higher trophic levels. The LD_{50} of blue green algae ranges from 400-12,000 Gy, and for other aquatic algae from 3-120 Gy (Woodhead, 1998). The LD_{50} range for blue green algae is similar to that reported for mosses, lichens, algae and bacteria.

The lowest dose reported to induce sub-lethal effects on aquatic plants ranges between 2,000 and 5,000 μGyh^{-1} (Chandorkar and Szachrajuk, 1978 cited in Environment Canada, 2000).

3.5 Summary

A large variation in radiosensitivity between taxa exists. Radiosensitivity tends to increase with increasing biological complexity of the organism, as indicated by lethal doses (Figure 3.1), with birds, mammals and few trees species considered the most radiosensitive. There is a positive correlation between radiosensitivity and increased DNA content of cells. As demonstrated in the impact assessment (Chapter 6), those organisms likely to experience the highest dose rates within an ecosystem tend to be less complex (e.g. bacteria) and as shown in Figure 3.1 are representative of the least sensitive groups.

Acute doses of 4,000 μGy h^{-1} induce persistent, measurable detrimental changes in populations and communities of terrestrial plants and animals (Barnthouse, 1995). Chronic irradiation (40 μGy h^{-1}) did not induce detrimental changes in any terrestrial populations (Barnthouse, 1995). Birds, mammals, reptiles and a few tree species are considered the most radiosensitive to chronic doses.

Developmental stages are generally considered more radiosensitive than adults. IAEA (1992) also found that lower doses (either acute or chronic) were required to induce reproduction effects, compared with the other assessed endpoints.

NCRP (1991) and IAEA (1992) recommended dose limits to ensure protection of aquatic and terrestrial populations (Table 3.1). The more recent studies provide further evidence that these values remain appropriate. However, the review does provide examples of laboratory studies where effects that may lead to population consequences are reported at chronic dose rates lower than the limits in Table 3.1. However, recent work in the exclusion zone around Chernobyl (Appendix 1) has demonstrated that wildlife exposed to high chronic doses are thriving under field conditions. It may be that in particular scenarios the intention is to protect all components of the ecosystem, such as nature conservation situations, thus the significance of the impact from ionising radiation must be considered further along with the need for additional studies.

Table 3.6 Reported impacts of acute ionising radiation on plants

Species	*Dose (Gy)*	*Radiation*	*Description*	*End point*	**Reference**
	<0.1		No visible damage	Morp	*Whicker 1997*
Pine	0.1-1	γ	Chromosome aberrations in reproductive organs (pollen) exposed during early stage of development.	Gen	*Tskayev et al., 1992 in UNSCEAR 1996*
"		γ	Chromosome damage in pines	Gen	*Kozubov et al.,1990 in UNSCEAR 1996*
"		γ	Damage to pine reproductive organs- pine cone size and pollen production rate	Rep	*Tikhomirov et al., 1978 in UNSCEAR 1996*
"		γ	Minor reductions in pine reproduction	Rep	*Whicker[a] 1997*
"		γ	Minor reduction in growth of pines	Gro	*Whicker[a] 1997*
Pine	1-5	γ	Temporary reduction in fertility and viability of pine pollen	Rep	*Tikhomirov et al., 1978 in UNSCEAR 1996*
"		γ	Growth inhibition of pines	Gro	*Whicker[a] 1997*
	5-10	γ	Disruption at Ecosystem level		*Alexakhin et al., 1993*
Pine		γ	Sterility of pines	Rep	*Whicker[a] 1997*
"		γ	Severe growth reduction of pines	Gro	*Whicker[a] 1997*
Pine	10-25	γ	Growth cessation of pines and severe crown damage	Gro	*Whicker[a] 1997*
	25-100	γ	Significant ecosystem level disruption		*Whicker[a] 1997*
Herbaceous		β	Delay in germination of herbaceous species and reduced germination rate	Rep	*Murphy et al., 1971 in UNSCEAR 1996*
Deciduous		γ	Delayed sprouting and early leaf fall of deciduous trees	Morp	*Whicker[a] 1997*
Herbaceous		γ	Morphological variation in herbaceous understorey species	Morp	*Smirnov and Melankholin 1979 in UNSCEAR 1996*
Shrub		β	Increase in aberrant flowers (extra petals, incomplete flowers) of shrubs	Morp	*Murphy et al., 1971 in UNSCEAR 1996*
Pine		γ	Severe mortality of pines	Mort	*Whicker[a] 1997*
Ash	>100	γ	Reduced survival of germinated ash seeds	Rep	*Heaslip 1971 in UNSCEAR 1996*
Deciduous		γ	Serve crown damage of deciduous trees	Morp	*Whicker[a] 1997*
Pine		γ	Complete mortality of pines	Mort	*Whicker[a] 1997*
Deciduous	> 200		Mortality of deciduous trees	Mort	*Whicker[a] 1997*
Herbaceous	>700		Damage to herbaceous communities.	Mort	*Whicker[a] 1997*

a Based on Tikhomirov and Shcheglov 1994, Alexakhin 1993, Arkhipov et al*., 1994, Kryshev 1992, Skuterund* et al*., 1994 and Smirnov 1993). b Based on J.R.Trabalka in Barnthouse 1995*.

Table 3.7 Reported impacts of chronic ionising radiation on plants

Species	*Dose Rate (mGy h^{-1})*	*Radiation*	*Description*	*End point*	*Reference*
	<100	γ	No likely effect at species or canopy level		*Amiro 1994*
	100-1000	γ	Canopy cover remains constant	Morp	*Amiro 1994*
Pine		γ	Reduced trunk growth of mature pine trees	Gro	*Woodwell and Miller 1963 in UNSCEAR 1996*
Pine		γ	Death of some conifers, population changes little.	Mort	*Amiro 1994*
Pine	(1-5) x10^3	γ	Reduced canopy cover of individual conifers, whole canopy remains constant.	Morp	*Amiro 1994*
"		γ	Decreased stem growth of saplings	Gro	*Amiro 1986 in UNSCEAR 1996*
"		γ	Reduced photosynthetic capacity of pines thus growth	Gro	*Bostrack and Sparrow 1970 in Woodhead 1998*
Pine	(5-10) x10^3	γ	Death of all conifers within 2-3 years	Mort	*Amiro 1994*
	(10-20) x10^3	γ	Reduced seed production and germination	Rep	*Whicker[b] 1997*
		γ	Morphological changes in leaves of some plants	Morp	*Whicker[b] 1997*
		γ	Withered crowns	Morp	*Whicker[b] 1997*
Birch		γ	Under developed leaves in birch trees	Gro	*Whicker[b] 1997*
Herba-ceous	>20 x10^3	γ	Reduced reproductive potential of herbaceous species	Rep	*Woodwell 1967 in UNSCEAR 1996*
Birch		γ	Death of birch trees	Mort	*Amiro and Sheppard 1994 and Whicker[b] 1997*
Grasses		γ	Death of grasses and forbs	Mort	*Whicker[b] 1997*
	>100 x10^3	γ	Death of all higher plants	Mort	*Amiro and Sheppard 1994 and Whicker 1997*
Lichen	>1000 x10^3	γ	Reduced diversity of lichen communities following exposure for 1 year	Mort	*Woodwell 1967 and Brodo 1964 both in UNSCEAR 1996*

Table 3.8 Reported impacts of acute ionising radiation on terrestrial invertebrates

Species	Dose (Gy)	Radiation	Description	End Point	Reference
	<0.1		No data available		
	0.1-1		No data available		
	1-5		No data available		
	5-10		No data available		
Earthworm	10-50	γ	Reduced hatching success of embryos irradiated in early embryogenesis.	Rep	*Suzuki and Egami 1983 in UNSCEAR 1996*
Earthworm		γ	Reduced hatching success of eggs laid by adults following irradiation	Rep	*Suzuki and Egami 1983 in UNSCEAR. 1996*
Soil and litter fauna		γ	No impact on adult soil or litter fauna but impacts on developing stages and juveniles observed		*Krivolutzkii and Pokarzhevskii 1992 in UNSCEAR. 1996*
Insect		γ	Reduced life expectancy of insects	Mort	*Meninick 1969 in UNSCEAR. 1996*
Earthworm	>100	γ	Inhibition of juvenile earthworm growth		*Suzuki and Egami 1983 in UNSCEAR. 1996*

Table 3.9 Reported impacts of chronic ionising radiation on terrestrial invertebrates

Species	Dose Rate (mGy h^{-1})	Radiation	Description	End point	Reference
	<100		No data available		
	100-1000		No data available		
Earthworm	(1-5) x10^3		Reduced population size	Mort	*Krivolutsky 1987*
	(5-10) x10^3		No data available		
	(10-20) x10^3		No data available		
	>20 x10^3		Reduced population size of soil invertebrates	Mort/ Rep	*Krivolutsky 1987*

Table 3.10 Reported impacts of acute ionising radiation on mammals

Species	*Dose (Gy)*	*Radiation*	*Description*	*End point*	*Reference*
Mouse	<0.1	γ	Increase in skeletal malformations following irradiation of developing embryo	Gen	*UNSCEAR 1986 in UNSCEAR. 1996*
		γ	No chromosome damage	Gen	*Max 1977 in UNSCEAR 1996*
		γ	Reduction in oocyte numbers of new-borns following irradiation of developing embryo	Rep	*Oakberg 1962 in UNSCEAR 1996*
		γ	Reduced sperm production	Rep	*Jagetia et al., 1995*
	0.1-1	γ	Malformation of trunk following irradiation during organogenesis	Gen	*Commission on Radiological Protection 1989 in UNSCEAR 1996*
		γ	Chromosome damage	Gen	*Whicker[a] 1997*
		β	Reduce oocyte numbers by 80%	Rep	*Dong et al., 1985 in UNSCEAR 1996*
		γ	No impact on sterility	Rep	*Shevchenko et al.,1992 in UNSCEAR 1996*
		γ	Impairment of female reproduction	Rep	*Gowen and Stadler 1964 in IAEA 1992*
		γ	No effect on mortality following late foetal exposure (during organogenesis)	Fec	*Devi et al., 1994*
		γ	Threshold for physiological and behavioural effects following exposure on day 13-18 of gestation	Beh	*Wang et al., 1993*
		γ	No effect on post natal growth following late foetal exposure (during organogenesis)	Dev	*Devi et al., 1994*
	1-5	γ	Increase in foetus malformation following parental exposure	Gen	*Muller et al., 1999*
		γ	Malformation of central nervous system and skull following irradiation during organogenesis	Gen	*Commission on Radiological Protection 1989 in UNSCEAR 1996*
		γ	Modified chromosome number and structure in oocytes of exposed adults	Gen	*Griffin and Tease 1988 in UNSCEAR 1996*
		γ	Impairment of male reproduction	Rep	*Gowen and Stadler 1964 in IAEA 1992*
		γ	Temporary sterility	Rep	*Shevchenko et al., 1992 in UNSCEAR 1996*
		γ	Increased mortality of embryos following parental exposure	Fec	*Muller et al., 1999*
		γ	Death of embryo exposed during early developmental stage	Fec	*Muller 1994 in UNSCEAR 1996*
		γ	Growth retardation following *in-utero* irradiation	Dev	*Hossain et al., 1999*

Table 3.10 cont.

	1-5 cont	γ	Impairment of adult brain function following exposure in late foetal development	Dev	*Devi et al 1999*
		γ	Increased postnatal mortality following exposure in gestation	Mort	*Hossain et al., 1999*
	5-10	γ	Temporary sterility of irradiated adults	Rep	*UNSCEAR 1977 in UNSCEAR 1996*
		γ	Permanent sterility of exposed juveniles	Rep	*UNSCEAR 1977 in UNSCEAR 1996*
		γ	Physical changes	Dev	*Whicker[a] 1997*
	>10	γ	Permanent sterility	Rep	*Shevchenko et al., 1992 in UNSCEAR 1996*
		γ	Reduction in populations	Mort/ Rep	*Whicker[a] 1997*
Rat	<0.1	γ	Embryonic and foetal mortality	Fec	*UNSCEAR 1986 in UNSCEAR 1996*
	0.1-1	γ	Sterility following exposure during early embryonic development	Rep	*UNSCEAR 1986 in UNSCEAR 1996*
	1-5	γ	Malformation of central nervous system and skull following irradiation during organogenesis	Gen	*Commission on Radiological Protection 1989 in UNSCEAR 1996*
		γ	Reduced testis weight and germ cell production of adults exposed as embryos	Rep	*Commission on Radiological Protection 1989 in UNSCEAR 1996*
		γ	Behavioural alterations becoming more marked with maturation following irradiation on day 15 of gestation	Beh	*Norton et al., 1991*
	5-10		No data found		
	>10	γ	reduced body weights following prenatal exposure on day 20 of gestation	Dev	*Zaman et al., 1997*

Table 3.11 Reported impacts of chronic ionising radiation on mammals

Species	*Dose Rate (mGy h^{-1})*	*Radiation*	*Description*	*End point*	*Reference*
Mouse	<100	α	Chromosome damage	Gen	*Searle et al., 1976 in UNSCEAR 1996*
		β	Reduction in oocyte numbers following in-utero irradiation	Rep	*Dobson and Cooper,1974 in UNSCEAR 1996*
		γ	Reduction in numbers of offspring	Rep	*Leonard et al., 1983 in IAEA 1992*
		α	Reduction in sperm output by 10% (5-8 months)	Rep	*Searle et al., 1976 in UNSCEAR 1996*
	100-1000	γ	Decreased germ cell production	Rep	*UNSCEAR 1986 in UNSCEAR 1996*
		α	Reduction in oocyte numbers by 20%	Rep	*Samuels 1966 in UNSCEAR 1996*
		γ	No effect on fertility following exposure over 10 generations	Rep	*Stadler and Gowen 1964 in UNSCEAR 1996*
		γ	Reduced survival	Mort	*French et al., 1969 in UNSCEAR 1996*
	(1-5)	γ	Increased genetic defects of sperm	Gen	*Pomerantseva et al., 1997*
	$x10^3$	γ	Sterility following irradiation during early embryonic development	Rep	*Brown et al., 1964 and Ronnback 1965 in UNSCEAR 1996*
		γ	Reduced lifespan following lifetime exposure	Mort	*UNSCEAR 1982 in UNSCEAR 1996*
	(5-10) $x10^3$		No data available		
	>10 $x10^3$	β	Embryo mortality	Fec	*Yamada 1982 in UNSCEAR 1996*
Rat	<100		No data available		
	100-1000	γ	Reduction in germ cell production following irradiation of embryo	Rep	*UNSCEAR 1986 in UNSCEAR 1996*
		β	Reduction in offspring oocyte numbers following parental irradiation of parent	Rep	*Pietrazak-Flis and Wasilewska 1984 in UNSCEAR 1996*
		β	Reduction in brain size of offspring following maternal irradiation during early pregnancy.	Dev	*Cahill et al., 1976 in UNSCEAR 1996*
	(1-5) $x10^3$	β	Reduction in ovary size following irradiation of embryo	Rep	*Cahill and Yuile 1970 in UNSCEAR 1996*
		β	Reduction in offspring weight following irradiation during gestation	Dev	*Cahill and Yuile 1970 in UNSCEAR 1996*
	(5-10) $x10^3$		No data available		
	> 10 $x10^3$		No data available		

Table 3.11 cont.

Species	*Dose Rate (mGy h^{-1})*	*Radiation*	*Description*	*End point*	*Reference*
Monkey	<100		No data available		
Pig			No data available		
Dog		β	No increase in incidence of cancer of the bones	Gen	*NCRP 1991b in UNSCEAR 1996*
“		γ	No effect on reproduction following lifetime exposure	Rep	Commission on the Biological Effects of Radiation 1980 in *UNSCEAR 1996*
“		β	No decrease in lifespan	Mort	*NCRP 1991b in UNSCEAR 1996*
Monkey	100-1000	β	Sterility following neonate exposure	Rep	*Dobson 1982 in UNSCEAR 1996*
Pig		γ	Reduction in gonad weight of offspring	Rep	*Erickson and Martin 1976 in UNSCEAR 1996*
“		γ	Reduction in number of germ cells following in-utero exposure	Rep	*UNSCEAR 1986 in UNSCEAR 1996*
Dog		γ	Sterility	Rep	*Commission on the Biological Effects of Radiation 1980 in UNSCEAR 1996*
Monkey	(1-5) x10^3		No data available		
Pig		γ	Sterile offspring following parental exposure	Rep	*Erickson and Martin 1976 in UNSCEAR 1996*
“		γ	Reduction of post natal brain weight	Dev	*UNSCEAR 1977 in UNSCEAR 1996*
Dog			No data available		
Monkey	(5-10) x10^3		No data available		
Pig			No data available		
Dog			No data available		
Monkey	> 10 x10^3		No data available		
Pig			No data available		
Dog			No data available		

Table 3.12 Reported impacts of acute ionising radiation on birds

Species	Dose (Gy)	Radiation	Description	End point	Reference
	<0.1		No data available		
	0.1-1	γ	No testicular damage	Rep	*Lofts and Rofblat 1962 in IAEA 1992*
		γ	No effect on growth and development of irradiated hatchlings	Dev	*Zach and Mayoh 1986a in UNSCEAR 1996*
	1-5	γ	Increased time to hatching (increase incubation time)	Rep	*Zach and Mayoh 1986b in UNSCEAR 1996*
		γ	Depressed growth	Dev	*Zach and Mayoh 1986b in UNSCEAR 1996*
		γ	No increase in mortality of newly hatched birds	Mort	*Zach and Mayoh 1986a in UNSCEAR 1996*
	5-10	γ	Reduction of nestling growth	Dev	*Zach and Mayoh 1986a in UNSCEAR 1996*
	>10		No data available		

Table 3.13 Reported impacts of chronic ionising radiation on birds

Species	Dose Rate (mGy h^{-1})	Radiation	Description	End point	Reference
	<100	γ	No impact on breeding performance over a season	Rep	*Zach et al., 1993 in UNSCEAR 1996*
		γ	No impact on production of fully fledged young	Rep	*Zach et al., 1993 in UNSCEAR 1996*
		γ	No impact on growth	Rep	*Zach et al., 1993 in UNSCEAR 1996*
		γ	No effect on embryonic mortality	Mort	*Zach and Mayoh 1982 in UNSCEAR 1996*
	100-1000	γ	No data available		
	(1-5) $x10^3$		No data available		
	(5-10) $x10^3$	γ	Reduced nesting of birds	Rep	*Whicker[b] 1997*
	>10 $x10^3$	γ	Embryonic mortality	Fec	*Zach and Mayoh 1986 in UNSCEAR 1996*

Table 3.14 Reported impacts of acute ionising radiation on reptiles and amphibians

Species	Dose (Gy)	Radiation	Description	End Point	Reference
	<0.1		No data available		
	0.1-1		No data available		
	1-5		No data available		
Reptile	5-10	γ	Reduced production of offspring following irradiation of adults	Rep	*Tinkle 1965 in UNSCEAR 1996*
		γ	Reduced survival of offspring and increase abnormalities.	Mor	*Blair 1960 in UNSCEAR 1996*
Toad	>10	γ	No impact on breeding activity of toads immediately after irradiation (pre hibernation) but increased mortality of toads post hibernation (1 year after irradiation).	Rep	*Tester et al., 1970 in UNSCEAR 1996*
Lizard		γ	Temporary sterility of lizard	Rep	*Dana et al., 1965 in UNSCEAR 1996*

Table 3.15 Reported impacts of chronic ionising radiation on reptiles and amphibians

Species	Dose Rate ($mGy\ h^{-1}$)	Radiation	Description	End point	Reference
Frog	<100	β	Increased chromosome aberration rate 2-10 fold	Gen	*Eliseyev et al., 1990 in UNSCEAR 1996*
Reptile	100-1000	γ	Regression of ovaries	Rep	*Turner et al., 1971 in UNSCEAR 1996*
"		γ	Induction of sterility in males	Rep	*Turner et al., 1971 in UNSCEAR 1996*
"		γ	Impact on maximal lifespan of some reptile species	Mor	*Turner et al., 1969 in UNSCEAR 1996*
	(1-5) $x10^3$		No data available		
	(5-10) $x10^3$		No data available		
	>10 $x10^3$		No data available		

Table 3.16 Reported impacts of acute ionising radiation on fish

Species	*Dose (Gy)*	*Radiation*	*Description*	*End point*	*Reference*
	<0.1		No data available		
Teleostei	0.1-1	γ	Reduction in sperm production	Rep	*Rackham and Woodhead 1984 in Greenwood and Knowles 1996*
Salmon		γ	Disruption of developing embryos	Fec	*Donaldson and Foster 1957 in NCRP 1991*
Plaice		γ	Mortality of larvae	Mort	*Ward 1971 in UNSCEAR 1996*
	1-5	γ	No visible changes in fish populations		*Whicker[a] 1997*
Trout		γ	Increased body deformities following irradiation of sperm	Gen	*McGregor and Newcombe 1972 in NCRP 1991*
Medaka		γ	Reduction in sperm proliferation and testis weight	Rep	*Konno and Egami 1966 in UNSCEAR 1996*
Salmon		γ	Reduction in female fertility	Rep	*Welander et al., 1948 in NCRP 1991*
Medaka	5-10	γ	Reduced mating success (medaka)	Rep	*Hyodo-Taguchi 1980 in NCRP 1991*
"		γ	Sterility	Rep	*Hyodo-Taguchi 1980 in NCRP 1991*
Trout		γ	Reduced fertility	Rep	*Konno 1980 in Harrison and Anderson 1996*
	>10	γ	Permanent sterility	Rep	*NCRP 1991*
Plaice		γ	Mortality of adult	Mort	*Templeton 1966 in Greenwood and Knowles 1996*

Table 3.17 Reported impacts of chronic ionising radiation on fish

Species	*Dose Rate (mGy h^{-1})*	*Radiation*	*Description*	*End point*	*Reference*
Plaice	<100		Anomalies of reproductive system		*Kryshev and Sazykina 1998*
Plaice	100-1000	γ	Decrease sperm production	Rep	*Knowles 1999*
Medaka		γ	Reduction in testis mass	Rep	*Hyodo-Tagachi 1980 in Greenwood and Knowles 1996*
Roach		γ	Lower fecundity, delayed spawning	Fec, Rep	*Peshkov et al 1978 in NCRP 1991*
	(1-5) $x10^3$	γ	Minor effects on fish gonads	Rep	*Woodhead 1984*
Guppy		γ	Infertility induced	Rep	*Purdom and Woodhead 1973 in NCRP 1991*
Plaice		γ	Reduced testis weight and sperm content (168 days)	Rep	*Greenwood and Knowles 1996*
Eelpout		γ	Decrease testis weight/sperm content	Rep	*Greenwood and Knowles 1995*
Medaka		γ	Reduced fertility	Rep	*Hyodo-Taguchi 1980 in NCRP 1991*
Medaka	(1-5) $x10^3$ cont	β	Severe depletion of spermatogonia (30d)	Rep	*Hyodo-Tagachi and Egami 1977 in NCRP 1991*
Guppy		γ	Fecundity reduced (988d)	Fec	*Woodhead 1977 in NCRP 1991*
Guppy		β	Reduced male courtship activity (17d)	Beh	*Erickson 1973 in NCRP 1991*
		γ	Death of fish	Mort	*Kryshev and Sazykina 1998*
Medaka	(5-10) $x10^3$	γ	Depletion of spermatogonia (120d)	Rep	*Hyodo-Tagachi 1980 in NCRP 1991*
"			No effect on mortality	Mort	*Hyodo-Tagachi 1980 in Greenwood and Knowles 1996*
Medaka	(10-50) $x10^3$	γ	Increase in vertebral anomalies	Gen	*Hyodo-Taguchi and Etoh 1993 in UNSCEAR 1996*
"		β	Increase in vertebral anomalies	Gen	*Hyodo-Taguchi and Etoh 1993 in UNSCEAR 1996*
"		β	Reduction in larval survival	Rep	*Hyodo-Taguchi and Etoh 1993 in UNSCEAR 1996*
"		β	No effect on hatching rate	Rep	*Hyodo-Taguchi and Etoh 1993 in UNSCEAR 1996*
"		γ	No effect on hatching rate	Rep	*Hyodo-Taguchi and Etoh 1993 in UNSCEAR 1996*
Guppy		γ	Sterility (288d)	Rep	*Woodhead 1977 in Harrison and Anderson 1996*
Guppy	>50 $x10^3$	γ	No impact on offspring survival following parental irradiation	Mort	*Woodhead 1977 in NCRP 1991*

Table 3.18 Reported impacts of acute ionising radiation on aquatic invertebrates

Species	*Dose (Gy)*	*Radiation*	*Description*	*End point*	*Reference*
	<0.1		No data available		
Polychaete	0.1-1		DNA strand breakage	Gen	*Harrison and Anderson 1994a*
"			Reduced fertility	Rep	*Harrison and Anderson 1994a*
Polychaete	1-5	γ	Increase in chromosome aberrations	Gen	*Anderson et al., 1990 in Harrison and Anderson 1996*
"		γ	Reduced brood size of adults	Rep	*Anderson and Harrison 1986 in NCRP 1991*
Polychaete	5-10	γ	Altered juveniles		*Anderson and Harrison 1986 in NCRP 1991*
Freshwater snail	10-50	γ	Reduced fertility	Rep	*Templeton et al., 1971 in Harrison and Anderson 1996*
Polychaete		γ	Sterility	Rep	*Harrison and Anderson 1994a*
Polychaete	>50	γ	Sterility	Rep	*Harrison and Anderson 1994a*
Freshwater snail		γ	Reduced fertility	Rep	*Templeton et al., 1971 in Harrison and Anderson 1996*
Polychaete		γ	Decrease in lifespan	Mort	*Anderson et al., 1990 in Harrison and Anderson 1996*

Table 3.19 Reported impacts of chronic ionising radiation aquatic invertebrates

Species	*Dose Rate (mGy h^{-1})*	*Radiation*	*Description*	*End point*	Reference
Midge	<100		Increase in chromosome aberrations	Gen	*Blaylock 1966 and 1973 in NCRP 1991*
Marine polychaete	100-1000	γ	Reduce number of hatching larvae, reduced fertility	Fec	*Harrison and Anderson 1994*
Marine polychaete	(1-5) x10^3	γ	Reduced breeding performance	Rep	*Knowles and Greenwood 1994 in UNSCEAR 1996*
"		γ	No effect on growth rate	Gro	*Knowles and Greenwood 1994 in UNSCEAR 1996*
Marine polychaete	(5-10) x10^3	β	No effect on egg production, decreased survival of eggs to larvae	Fec	*Knowles and Greenwood 1997*
"		γ	Decreased egg production, no effect on larvae survival	Fec	*Knowles and Greenwood 1997*
Blue crab		γ	No effect on mortality	Mort	*Engel 1967 in UNSCEAR 1996*
Marine scallop		γ	No effect on juvenile mortality	Mort	*Baptist et al, 1976 in UNSCEAR 1996*
Marine polychaeta	(10-50) x10^3	γ	Gamete killing, reduce fertilisation success	Rep	*Harrison and Anderson 1994b*
"		γ	Sterility	Rep	*Harrison and Anderson 1994b*
Freshwater snail		γ	reduced egg production (98 d)	Rep	*Cooley 1973 in NCRP 1991*
Daphnia	>50 x10^3	γ	reduced fecundity	Fec	*Marshall 1962 in NCRP 1991*
Blue crabs		γ	Reduction of growth	Gro	*Engel 1967 in UNSCEAR 1996*
Daphnia		γ	Increased mortality with the additional stress of food limitation.	Mort	*Marshall 1962 in NCRP 1991*
Freshwater snail		γ	Reduced survival	Mort	*Cooley and Miller 1971 in UNSCEAR 1996*
Blue crabs		γ	Increased mortality	Mort	*Engel 1967 in UNSCEAR 1996*

4. Existing Regulatory Frameworks for Environmental Protection

This Chapter reviews the existing regulatory frameworks for environmental protection from radiation adopted internationally. It reviews in more detail the legislation which affects the UK and places the recommended impact assessment (as discussed in Chapter 4) in context.

4.1 Introduction

No internationally agreed set of criteria exists at present for the protection of the environment from the effects of ionising radiation. This is due in part to the belief that such a set of criteria has not been warranted in the past. In fact, most of the existing framework and legislation on the protection of the environment from ionising radiation is based upon the ICRP statement:

> *''The Commission believes that the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species. At the present time, the Commission concerns itself with mankind's environment only with regard to the transfer of radionuclides through the environment since this directly affects the radiological protection of man''* ICRP (1991).

However, criticisms of the ICRP approach include:

- there appears to be no explicit account taken of environments where humans (and the pathways leading to exposure of humans) are absent either now or in the future (Pentreath, 1999);
- detrimental impacts may not necessarily be avoided in environments such as the Arctic and Antarctic regions; i.e. the concept of fragile ecosystems.
- no direct evidence has been cited to confirm the statement and, the information, upon which the statement was originally based, has been questioned (Thompson, 1988);
- the premise may be challenged in situations where humans have been deliberately excluded from an area, perhaps as the result of accidental releases of radionuclides.

There is an inconsistency between the protection of the environment and protect of humans. However, there is now increasing pressure to demonstrate that the environment is protected in its own right. This requires a reversal in thinking in radiological protection so that "to protect humans is to protect the environment" becomes "to protect the environment is to protect humans", i.e. the adoption of the "precautionary principle" (Santillo *et al.*, 1998; Hey, 1993). Protection of the environment from non-radioactive contaminants has already adopted this precautionary approach.

Recent Developments

The aim of the OSPAR Convention is to prevent pollution of the marine environment (NE Atlantic) from land-based sources. The Sintra Statement summarises the 1998 strategy agreed by the OSPAR Commission:

> *"To undertake the development of environmental quality criteria for the protection of the marine environment from adverse effects of radioactive substances and report on progress by year 2003."*

The objectives of the strategy are:

- to prevent pollution of the maritime area from ionising radiation, specifically the North East Atlantic, through progressive and substantial reductions of discharges, emissions and losses of radioactive substances; and

- to achieve concentrations of radioactive substances to "near background" for naturally occurring radioactive substances and close to zero for artificial radioactive substances (the exact meanings and approach is still being debated).

In response to this agreement, the UK Government published a consultation document "UK strategy for Radioactive Discharges 2001-2020" in June 2000 (DETR 2000). The strategy is set in the wider context of environmental protection in the UK. Its aims are:

- progressive and substantial reductions in radioactive discharges from UK as a whole and from each of the main sectors responsible for such discharges;
- progressive reduction of human exposure to ionising radiation resulting from radioactive discharges; and
- progressive reductions of radionuclide concentrations from radioactive discharges into in the marine environment to add close to zero above historic levels by 2020.

The consultation strategy provides a policy base for the regulatory review of discharge authorisations in the future, and for planning by nuclear operators.

4.2 UK approach to the protection of the environment from ionising radiation

The nuclear industry in the UK has always been subject to stringent national legislation (available on line at http://www.hmso.gov.uk/acts.htm) with respect to the use of nuclear materials, containment of radiation sources and radioactive waste discharges and disposals:

Nuclear Installations Act, 1965 (NIA 65)

The NIA 65 requires nuclear sites to be licensed by the Health and Safety Executive (HSE). A nuclear site licence is required to install or operate a nuclear installation and also to store, process and dispose of nuclear fuel and other radioactive matter resulting from the production and use of nuclear fuel. The Nuclear Installations Inspectorate of the HSE regulates the storage of radioactive waste at licensed sites.

The Basic Safety Standards Regulations, 2000

The regulations came into UK law in May 2000. The Regulations implemented the European Basic Safety Standards (BSS) Directive 96/29. It sets out requirements to protect workers and the general public against the dangers of ionising radiation. The principles and standards of radiological protection contained in the BSS Directive are based on the recommendations of the International Commission on Radiological Protection (ICRP). These seek to provide an appropriate standard of protection to humans without unduly limiting the beneficial uses of the practices giving rise to radiation exposure. Government has directed the Environment Agency to ensure that relevant dose constraints for humans are observed.

Environment Act, 1995

The Act established the Environment Agency by merging the regulatory and administrative powers of the National Rivers Authority, Her Majesty's Inspectorate of Pollution, and the waste regulation duties of Local Authorities. The Environment Act 1995 created new duties for the Environment Agency such as conducting research, providing information on the environment, and the duty of care. The emphasis of the Environment Act is to contribute to sustainable development through pollution prevention, with the object of the Environment Agency to protect and enhance the environment taken as a whole.

Radioactive Substances Act, 1993 (RSA 93)

The RSA 93, as amended by the Environment Act 1995, is concerned with controlling radioactive materials and waste. All forms of radioactive waste, including discharges of liquid and gaseous effluents, are regulated under this Act. The Act requires registration prior to use of radioactive materials and authorisation for disposal and accumulation of radioactive waste.

Where radioactive waste is stored on sites licensed under the Nuclear Installations Act 1965, the HSE has statutory powers to regulate that storage. However, the Environment Agency is responsible for regulating disposals of all forms of radioactive wastes on or from those HSE licensed sites.

Disposal of radioactive waste by a means other than that set out in the authorisation granted by the regulatory body is prohibited. The Environment Agency is named as the responsible body for authorising radioactive discharges under the Act in England and Wales. Sites covered by RSA 93 regulations include nuclear sites, which hold an operating licence under the Nuclear Installations Act 1965, and other sites where the handling or use of radioactive substances is not the main activity but minor levels of radioactivity are discharged into the environment e.g. hospitals.

Protection of the environment from ionising radiation in the UK is achieved primarily through legislation concerned with pollution control, such as the Radioactive Substances Act 1993. Other legislation concerned with nature conservation also contributes to the protection of the environment from ionising radiation, by including consideration of the potential environmental impacts from any permission, such as discharge consents. The most relevant Regulations derived from EU Directives are:

Countryside and Rights of Way Act 2000

The Countryside Rights of Way Act 2000 amends the Wildlife and Countryside Act 1981. It introduces a statutory basis for biodiversity conservation placing a duty on government departments to consider biological diversity when undertaking their duties.

Some of the provisions of the Act came into force in January 2001, more in April 2001, with the remaining parts coming into effect over the next few years. In terms of nature conservation, the Act strengthens legislation and facilitates better management of Sites of Special Scientific Interest (SSSI). It also places a general duty on any government department to have regard to conservation of biodiversity and to further and enhance the conservation of SSSIs.

Conservation (Natural Habitats) Regulations 1994

The UK Habitats Regulations 1994 implement the Habitats Directive (Council Directive 92/43/EEC on the conservation of natural habitats and of wild flora and fauna), and provides mechanisms to protect sites designated under the Birds Directive. The regulations require measures to be taken to maintain or restore to favourable conservation status in their natural range, habitats and species of wild flora and fauna of Community interest and listed in Annexes to the Directive.

The Habitats Directive provides for a European ecological network of **Special Areas of Conservation (SACs)**, which together with **SPAs** are known as 'Natura 2000' sites.

The Habitats Regulations 1994, Wildlife and Countryside Act 1981, and the Countryside and Rights of Way Act 2000 are the major pieces national legislation protecting many UK species of mammals, birds, invertebrates, plants and their habitats from harm. These regulations collectively offer the principle means where by protection of designated nature conservation sites from potential damaging effects from operations such as radioactive discharges is achieved.

Town and Country Planning (Assessment of Environmental Effects) Regulations 1988

This provides a mechanism for controlling pollution at source: 'prevention is better than cure'. All planning applications for schedule 1 listed installations (including nuclear sites) must undergo an environmental impact assessment as an integral part of the planning permission process. This ensures that the impact of development on the environment is fully considered by local Authorities, in consultation with the Environment Agency, prior to construction. This Directive also requires the assessment of potential effects on the environment before nuclear reactors can be decommissioned. Site operators require consent from the Health and Safety Executive before dismantling or decommissioning work can start.

Along with the legislation already implemented within the UK, new Directives proposed by the EC will also impact on protection of the environment from ionising radiation.

Water Framework Directive (2000/60/EC)

The ultimate aim of this Directive is to:

"achieve the elimination of priority hazardous substances and contribute to achieving concentrations in the marine environment near background levels for naturally occurring substances."

The Directive does not differentiate between radioactive substances and other contaminants. The aims are to protect all waters, i.e. groundwaters and surface waters, freshwaters and coastal waters, and to achieve 'good status' for all waters. Law, regulations and administrative provisions necessary to comply with the Water Framework Directive should be introduced by December 2003.

The above regulations and pending Directive underpin the need to develop a framework to assess the risk to wildlife from ionising radiation as a result of authorised discharges that impact on designated sites.

4.3 Role of regulatory bodies

The Environment Agency and English Nature, together with the Countryside Council for Wales (CCW), are working in partnership to develop joint guidance on assessing and reviewing Agency permissions and activities under the Conservation (Natural Habitats) Regulations 1994. This report will be used to inform this process and will specifically address the assessment of radiological impact to wildlife.

4.3.1 Environment Agency

The Environment Agency is a non-departmental public body. Its principal aim is to protect or enhance the environment, taken as a whole, and to contribute to sustainable development through pollution prevention. Other pollution control duties are to prevent, minimise, remedy or mitigate the effects of pollution to the environment.

Under RSA93, radioactive discharges from nuclear installations must be authorised prior to disposal. Authorisations place limits and conditions on operators to ensure that the radiation doses to humans resulting from radionuclide discharges remain within internationally agreed limits.

Authorisations usually:

- include limits on the amounts of α and β radioactivity that can be discharged in a given time period and will control discharges of certain named isotopes.
- stipulate the means of disposal and the radioactive content of the waste. 'Disposal' includes the removal, deposit, destruction or discharge into water, air, sewers, drains and burial, and encompasses all types of solid, liquid and gaseous radioactive discharges.
- stipulate the discharge location, manner by which the discharge can occur and the monitoring programme that the site operators must undertake. Discharge limits must not be exceeded.
- are re-evaluated regularly, with a full review undertaken usually every four years.

The Environment Agency is responsible for issuing new authorisations, varying and reviewing existing authorisations. The Environment Agency also reports on the state of the environment based on their own independent monitoring programmes (EA 2001a).

Under the Habitats Regulations, the Environment Agency is required to review consents and authorisations for discharges affecting Natura 2000 sites and to assess the possible impacts of new authorisations and consents. These discharges, whether directly released into the designated sites or having a potential impact on them, must exert no adverse effect on the integrity of the site.

Non-nuclear sites [i.e. Sites not licensed under NIA 65]

Discharges of small amounts of radioactive liquid wastes to sewer systems from non-nuclear organisations are permitted in the UK. Certificates of Authorisation to dispose of radioactive, liquid waste are issued by regulatory agencies (Environment Agency in England and Wales) under RSA 93. The radiological risks of such discharges must be small and the disposal route selected considered the best option, as for the nuclear sites.

4.3.2 English Nature and the Countryside Council for Wales (CCW)

English Nature and Countryside Council for Wales are statutory advisers to the government on nature conservation. English Nature currently has a lead agency role on behalf of CCW on toxic substances, including matters to do with radioactive substances. English Nature, along with other conservation agencies, have greater power under the Countryside and Rights of Way Act 2000 to refuse consents for activities considered to be damaging.

It is a duty of English Nature to inform the Environment Agency of any land of special conservation interest. Once notified the Environment Agency must consult English Nature over any matters concerning discharges to that area. This provides a mechanism to protect the environment from effluent discharges and prevent pollution and deterioration of designated sites.

This report provides the Environment Agency and English Nature with an assessment tool with which to review consents and authorisations by determining the impact on wildlife within England and Wales. This report also provides information on the likely effects of exposure to ionising radiation with which to compare assessment results, along with guidance as to how these two parts should be interpreted.

4.4 International practice on the protection of the environment from ionising radiation

International thinking, practice and consensus have been slow to develop on protecting the environment from ionising radiation. However in recent years the topic has gained emphasis as a consequence of public pressure to ensure standards for the protection of the environment are similar to those that protect workers in the nuclear industry. For example:

- Canada, Sweden and Australia sponsored two international conferences in Stockholm (1996) and Ottawa (1999). These events highlighted the fact that their regulatory systems could not ensure environmental protection from ionising radiation, and that there was no international consensus or guidance on the approach to be taken.
- The International Commission of Radiological Protection (ICRP) and the International Union of radioecologists (IUR), have both instigated Task Groups to consider aspects of the protection of the environment from the effects of ionising radiation (Dublin, 2000).
- European funding for projects such as FASSET (Section 4.6) has demonstrated that there is a perceived need to develop a suitable (and agreed) framework for the protection of the environment.

Guideline dose limits for biota have been recommended by international organisations such as the IAEA (1992) (Table 4.1), below which significant effects are unlikely. A number of countries such as Canada and the USA have also suggested dose limits for biota (Table 4.1). The dose limits for biota recommended by the IAEA have generally been well received. The Environment Agency uses the IAEA guidelines when following its current assessment approach to determine the likely impact of exposure to ionising radiation from authorised discharges.

The IAEA guidelines in Table 4.1 are recommended by the authors for use in impact assessments, subject to periodic updates as some genetic and reproductive effects at dose rates below the guideline limits have been observed on mice, fish and aquatic invertebrates (Section 3.4.2 and Tables 3.11, 3.15, 3.17, and 3.19). Consequently these guidelines may change in the future. The impact assessment approach described in this report develops the existing EA approach to provide a generic impact

assessment. It is therefore important to recognise that the assessor must consider site specific features such as the presence of rare species when using generic guideline values given in Table 4.1 to evaluate the impact of ionising radiation on wildlife. In such instances generic guidelines should be used with caution and possible re-evaluation of the dose limits recommended within this report may be required. It must be recognised that the setting of standards to protect biota from ionising radiation must be decided by politicians/international organisations based on available scientific evidence.

Table 4.1 Guideline and recommended dose limits (mGyh^{-1}) to biota (replicate of Table 3.1)

	NCRP, 1991	IAEA, 1988 and 1992	Thompson, 1999[1,2]	USA, Department of Energy[3]
Terrestrial				
Plants		*400*		*400*
Animals		*40*		*40*
Mammal			*10*	
Birds			*50*	
Amphibians/Reptiles			*10*	
Aquatic				
Freshwater organisms	*400*	*400*		*400*
Benthic invertebrates			*100*	
Fish			*50*	
Deep ocean organisms		*1000*		

1-calculated from annual 'critical' dose limits which correspond to the lowest doses at which effects are observed. To incorporate a safety factor a 'no effects dose' has also been devised set at 1/10th of the corresponding critical dose. 2-Currently under public consultation in Canada. 3-Stephen Domotor pers. comm. IAEA Specialists Meeting on Protection of the Environment from the Effects of Ionising Radiation, International Perspectives, August 29-September 1, 2000.

Finland

Two pieces of legislation provide the legal basis in Finland for protection of the environment:

- The Nuclear Energy Act (1987) requires that the use of nuclear energy must be safe, shall not cause injury to people, or damage the environment. An operation licence is granted once the protection of workers, population and the environment have been adequately considered.
- Nuclear Energy Decree (1988, amended in 1994 to add the need for environmental impact assessment) requests that applications outline the effects of the nuclear facility on the environment, and describes the design criteria to avoid damage and restrict burden on the environment.

The term 'environmental protection' is interpreted in different ways depending on the purpose of the regulation:

- it means limitation of discharges and monitoring of the environment to ensure public protection when considering discharge regulations;
- it includes the need to protect the non human environment in the field of waste disposal.

The Government Decision on Safe Disposal of Spent Nuclear Fuel (1999) requires environmental impact assessments on species of flora and fauna from spent fuel disposal facilities. A Guide on long term safety of spent nuclear fuel disposal is due out in 2001. This guide will set out the requirements to assess typical radiation exposures to terrestrial and aquatic biota within the vicinity of the disposal site to ensure detrimental impacts on flora and fauna are prevented.

Norway

The Norwegian Radiation Protection Authority (NRPA) recognises that the protection of the environment in addition to humans is required within a regulatory framework. The Act on Radiation and Use of Radiation was introduced in 2000 to ensure that the harmful effects of radiation on the health of the population are prevented, and that the protection of the environment is considered. This law applies to all aspects of the nuclear industry such as: production, import, export, transport, transfer, possession, installation, use and disposal of radioactive sources, and processes that enhance natural levels of ionising radiation. NRPA does not routinely apply dose assessments to non-human biota, but *ad hoc* assessments are to form the basis for developing such a methodology.

Sweden

The revision of the Radiation Protection Act in 1988 widened legislation to include the protection of the environment from the harmful effects of radiation. The Swedish Radiation Protection Institute sets the goals for environmental protection in the recently revised regulations on the management of spent nuclear fuel and nuclear waste, and on discharges from nuclear installations. The Government has introduced 15 environmental quality goals with the aim of controlling pollutants, contaminants and related activities. A safe radiation environment is listed as one of such goals. The methodology is under development to assess environmental impact and compliance with these goals, building on experience from the EC-funded FASSET project (see Section 4.6).

USA

The US Nuclear Regulatory Authority is responsible for regulation of radiological contaminants from the entire fuel cycle (uranium mills through to waste disposal). Their goals include the protection of the environment and an increase in public confidence. The United States Department of Energy (USDOE) is proactive in developing frameworks, methods and guidance to demonstrate the protection of the environment from the impacts of ionising radiation (Table 4.1). The Biota Dose Assessment Committee (BDAC) has devised an approach for evaluating doses to biota, and radiation dose standards are in place to protect aquatic organisms, http://homer.ornl.gov/oepa/public/bdac/.

Canada

The Canadian Nuclear Safety Commission (CNSC) was established in June 2000 (formerly the Atomic Energy Control Board (AECB) of Canada) when the Nuclear Safety and Control Act (NSCA), and pursuant regulations, came into force. The NSCA and pursuant regulations have a focus on the principle of pollution prevention and the ecosystem approach.

The CNSC is responsible for all aspects of nuclear installations and uranium mining activities, and has the obligation under the NSCA to prevent unreasonable risk to the environment. Holders of nuclear facility licenses (e.g. nuclear power plants, research and production facilities, uranium mines, mills and refineries, fuel fabrication facilities and waste management facilities) have an obligation to take all reasonable precautions to protect the environment and to prevent the release of nuclear and hazardous substances to both the on-site and off-site environment.

A regulatory policy document "Protection of the Environment P-233" (CNSC 20001) has been published and several regulatory guides are in preparation. Guidelines are being developed for the protection of different taxonomic groups of organisms from exposure to radiation (Table 4.1). These standards contain dose levels (for a no-effect level) for different taxonomic groups and will be used to identify situations where no damage may be expected. These 'no effect' dose levels are derived from conservative 'critical levels'.

Australia

Australia has a pragmatic approach to managing environmental impacts of radioactivity. It is centred on the management of uranium mining operations that are close to or in National Parks. Sites require a full assessment of the possible chemical, radiological and physical impacts on the environment arising from the operations of the mines and their closure (Needham, 1996).

Water quality standards have been set for both radioactive and non-radioactive pollutants based on ecotoxicity tests on biota found in the ecosystems impacted. As a result safety standards that have been set are concentration-based and site-specific, but the issue of setting dose limits for those biota impacted by ionising radiation has not arisen, perhaps because the chemical toxicity of the effluents is more significant than the radiological impact (Needham, 1996; Needham, 1999; Jackson, 1999).

4.5 Major national and international organisations

There are a number of major national and international organisations that play a role in environmental radiation protection.

International Atomic Energy Agency (IAEA)

http://www.iaea.org/worldatom/

The International Atomic Energy Agency (IAEA) is an intergovernmental, science and technology-based organisation, which serves as the world's central inter-governmental forum for scientific and technical co-operation in the nuclear field.

The IAEA was established in 1957. As of December 1999, 130 States were members of the IAEA. The IAEA is required to establish standards for the protection of human health, which often uses UNSCEAR estimates of exposure. Principle 2 of the IAEA Safety Fundamentals for the Management of Radioactive Waste (1995) states:

> *"Radioactive waste shall be managed in such a way as to provide an acceptable level of protection for the environment".*

The IAEA publishes safety guides on issues such as; regulatory control of radioactive discharges to the environment; strategies for development of monitoring programmes for radionuclides in the environment; and generic models suitable for assessing the impact of radioactive substance discharges to the environment. IAEA guidance has implications for radioactive waste disposal, discharge control, environmental assessment and monitoring and environmental restoration.

Present guidance is concerned with the protection of critical human groups. In the future the guidance is likely to incorporate the protection of flora and fauna through methodologies for assessing doses and for monitoring compliance with new protection criteria. The IAEA plans to develop a Safety Guide on 'Principles for the protection of flora and fauna against the effects of ionising radiation.

International Commission on Radiological Protection (ICRP)

http://www.icrp.org/

The International Commission on Radiological Protection (ICRP) is an independent charity based in Stockholm, Sweden. Founded in 1928 by the International Society of Radiology, the ICRP was established to advance for the public benefit the science of radiological protection, in particular by providing recommendations and guidance on all aspects of protection against ionising radiation. The Commission concerns itself primarily with the protection of humans, with the environment only considered as a pathway for the transfer of radionuclides to man (ICRP, publication 60, 1990).

- The ICRP is composed of a main Commission and four standing committees, including one on radiation effects and one on doses from radiation exposure.

The ICRP uses Task Groups (performing defined tasks) and Working Parties (developing ideas) to prepare its reports from the fields of medical radiology, radiation protection, health physics and radiation biology.

International Union of Radioecologists (IUR)

http://www.iur-uir.com/

The IUR is a non-political and non-profit scientific organisation for professional radioecologists. With a membership of over 600 from more than 40 countries, the IUR represents an authoritative source of information on all aspects of radioecology.

It aims first at being a forum of information exchange between the radioecologists. IUR activities are organised through a number of task groups, workshops, conferences and training courses. A major programme of research and discussion under way currently is concerned with the exposure to ionising radiation and effects on biota.

Nuclear Energy Agency (NEA)

http://www.nea.fr/

The Nuclear Energy Agency (NEA) is a semi-autonomous body within the Organisation for Economic Co-operation and Development (OECD), located in Paris. The objective of the Agency is to contribute to the development of nuclear energy as a safe, environmentally acceptable and economical energy source through co-operation among its participating countries. The Agency's mission is:

> *"To assist its Member countries in maintaining and further developing, through international co-operation, the scientific, technological and legal bases required for a safe, environmentally friendly and economical use of nuclear energy for peaceful purposes, as well as to provide authoritative assessments and to forge common understandings on key issues, as input to government decisions on nuclear energy policy and to broader OECD policy analyses in areas such as energy and sustainable development."*

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR)

UNSCEAR is the only body of the UN with a mandate to assess and report levels and health effects of exposure to ionising radiation. The Committee takes an independent and neutral position when reviewing published data. Conclusions of UNSCEAR are used by the UN General Assembly to form recommendations concerning radiation. Governments and organisations throughout the world also rely on UNSCEAR reports to assess the risk from ionising radiation and to devise radiation protection and safety standards.

National Radiological Protection Board (NRPB), UK

http://www.nrpb.org.uk/

The National Radiological Protection Board (NRPB) was created by the Radiological Protection Act (1970) and has the following statutory functions:

- by means of research and otherwise, to advance the acquisition of knowledge about the protection of mankind from radiation hazards;
- to provide information and advice to persons (including Government Departments) with responsibilities in the United Kingdom in relation to the protection from radiation hazards either of the community as a whole or of particular sections of the community.

4.6 Future Developments

It is the view of the European Commission that the understanding of radiation impacts on the environment is insufficient to permit the introduction of new measures at Community level. The two main research projects under the EC 5th Framework Programme are aiming at developing by 2004 a

framework for protection of the environment from ionising radiation in Europe and in the Arctic: FASSET (Framework for ASSessment of Environmental impacT); and EPIC (Environmental Protection from Ionising Contaminants in the Arctic).

The Environment Agency is participating in FASSET, which aims to:

- identify target organisms in terrestrial and aquatic ecosystems of Europe and to develop corresponding dosimetric models;
- use ecosystem-based approaches to identify critical internal and external exposure situations and corresponding models;
- identify critical effects and biological organisation levels of concern; and
- link these components into a framework, taking account of, *inter alia*, existing frameworks for managing risks from other pollutants.

The framework will be a useful tool for assessing environmental impact, judging compliance against environmental quality criteria and standards, and communicating to different stake-holders the likely environmental consequences of projects in a planning stage. Further information on FASSET is available online at http://www.fasset.org/.

Future research is also needed to develop an integrated approach to assess the total environmental impact of a site discharging both radioactive and non-radioactive toxic pollutants. In this way environmental protection will evolve to ensure both wildlife and humans are sufficiently protected from harm.

5. Dosimetric Methods

5.1 Introduction

The principal quantity, which needs to be evaluated in assessing radiation doses to organisms, is the *absorbed dose* – that is, the amount of energy absorbed by the organism from ionising radiation, per unit mass of the organism.

Radiation dose may arise either from radionuclides present in the soil, sediment, water or air surrounding the organism *(external dose)* or from radionuclides taken up internally by the organism *(internal dose*). Slightly different approaches are necessary in assessing these two sources of dose, as described below.

The International Commission on Radiological Protection has, over many years, developed a comprehensive system of dosimetry for humans. The system for assessing internal dose, in particular, is quite complex. It provides for the distribution of radionuclides between the different body organs and for modelling their subsequent clearance over a period of time following their initial incorporation. However, apart from the broad concepts involved, these systems are of little help in developing dosimetric methods for wildlife since both the dimensions and the underlying metabolism of the organisms to be considered differ greatly from those of humans. Moreover, a wide variety of different organisms need to be considered.

The current 'state of the art' in wildlife dosimetry therefore involves a high degree of simplification. Various approaches have been advanced.

Amiro (1997) has set out an extensive Table of conversion factors for external exposure together with factors for internal exposure, related in both cases to the concentration of radionuclides in the medium (soil, water, air or vegetation) surrounding the organism or the concentration of radionuclides in the organism itself. However, no provision is made for applying weighting factors to the á or low energy â components of the radiation emitted. In calculating internal doses, organisms are assumed to be infinitely large; that is, no allowance is made for the proportion of radiation energy, which escapes from the organism without being absorbed. This will lead to a very significant over-estimate of the internal dose due to ã ray emissions, and also of the internal dose due to â particle emissions in small organisms. External doses from â emitters are calculated as 'surface doses' at a depth of 70 μm in the exposed organism, which in most circumstances will significantly over-estimate the doses to sensitive internal organs.

The USDoE (2000) has established an assessment protocol which uses a 'graded approach', starting with very simple and conservative assumptions and progressing to a more realistic assessment if the doses to organisms appear to be significant. However, even the most realistic assessment approach assumes organisms are simultaneously infinitely large (when calculating internal doses) and infinitely small (when calculating external doses). The approach is therefore likely to be very conservative.

The most realistic approach to wildlife dosimetry so far developed is that advanced by Woodhead and others (e.g. Woodhead, 1979; NCRP, 1991; Woodhead, 2000b). In this approach organisms are represented by ellipsoids of appropriate dimensions, and the proportion of radiation absorbed within the volume of the organism is estimated using formulae which describe the distribution of radiation doses around point sources within the organism. It is necessary to integrate the resulting radiation doses over all hypothetical 'point sources' and 'point receptors' within the organism. For simple cases, this can be done 'analytically' by use of calculus. For more complex cases, or for rapid calculations to cover a range of different radiation energies and organism dimensions, it is more convenient to use numerical methods in a suitable computer programme.

The main obstacle to use of the above methods has been that results have only been published for a relatively small number of different ellipsoid dimensions. Moreover, the results are presented simply in the form of graphs from which accurate data for use in calculations cannot very easily be obtained.

For this report, a calculation approach has been developed which makes the above scheme of dosimetry more readily accessible through a series of spreadsheet-based computer programmes.

In this approach, organisms can be characterised for the purpose of dose calculations by:

- **Dimensions.** Organisms are represented as ellipsoids of unit density, with defined major and minor axes (the minor axes need not be equal in length, so that both the longitudinal and transverse cross sections can be elliptical).
- **Concentration of radionuclides within the organism.** A concentration factor[5] is specified for each organism and each nuclide. The concentration factors for terrestrial organisms are specified relative to soil, and relative to water for aquatic organisms.
- **Distribution of internal contamination and dose.** Internal contamination is assumed to be uniformly distributed within the organism, and the resulting absorbed dose is calculated as an average throughout the volume of the organism.
- **Location of the organism relative to soil water or sediment.** The fractional occupancies underground, on the soil/sediment surface, and fully immersed in air or water are specified for each organism when calculating external doses.

For the assessment of internal dose it is necessary to first estimate the fraction of energy which is absorbed within the organism. The internal dose rates can then be calculated. For the assessment of external dose, simple formulae are used for dose in an infinite or semi infinite absorbing medium. The methods used for these calculations are described in the Sections below.

The calculation scheme developed in this report, and implemented in the accompanying spreadsheets, differs from that set out by Woodhead (2000b) only in the way in which â dose is averaged within the volume of the organism. Woodhead's method is based on the â dose calculated at the centre of the organism; the method described here evaluates the â dose averaged throughout the volume of the organism. Woodhead's method will tend to produce higher doses from internally incorporated â emitters, but lower external doses from â emitters within the surrounding environmental media, than does the method described here.

Either method is probably an equally satisfactory approximation to the â dose to the 'critical' or most sensitive organ within the exposed organism, unless and until such critical organs are identified and all the data necessary for a more explicit dose calculation become available. The calculation scheme described here has been compared with Woodhead's scheme based on values for the dose per unit concentration in water derived from the latter (Woodhead, 2000b). The results of the two schemes compared were very comparable, differences as large as a factor of two only arising when doses from â emitters are dominant and being readily explained by the different basis for the assessment of â doses from internal sources between the two schemes.

5.2 Calculation of absorbed fractions

Absorbed fractions from γ, β and α radiation are treated separately due both to physical differences in their distributions of dose around point sources, and to differences in their effects on biota (refer to Chapter 3). The formulations of the equations are explained in this Section, and the resulting absorbed fractions are used for calculating internal doses. Absorbed fractions have been calculated for all the organism dimensions considered in this report, and incorporated into the accompanying dose assessment spreadsheets in the form of dose per unit concentration (DPUC) factors (see below). Thus, it is not necessary for the user to calculate absorbed fractions in order to make an assessment of doses to organisms considered in this report.

[5] A concentration factor is defined as the ratio of element or nuclide in the consumer (or specific tissue, organ etc.) to that in what is consumed, or to that in the environmental medium (Warner and Harrison, 1993). A CF takes into account all physiological and physico-chemical properties, which may influence the uptake and accumulation of radionuclides into biota.

5.2.1 Gamma ray photons

Monte Carlo[6] based codes, which replicate the interactions between ã ray photons and tissue, have been used in dosimetry models to calculate absorbed fractions for humans, (e.g. Cristy and Eckerman, 1987). These codes are complex and time consuming to run, but can produce accurate results for very detailed representations of the geometry of organs or organisms. As described in the above introduction, a simpler approach has developed for this report, based on existing dosimetric systems for biota (e.g. IAEA, 1979; NCRP, 1991). This approach utilises simple semi-empirical formulae, which represent the distribution of dose in space around a point radiation source located in an infinite isotropic absorbing medium.

Berger (1968) provided dose distribution data for photons in terms of the *point isotropic specific absorbed fraction* $\Phi_E(r)$, which is the fraction of energy absorbed per gram of absorbing medium at a distance r cm from a point source of ã ray photons of energy E MeV:

$$\Phi_E(r) = \left[\frac{\mu_{abs} e^{-\mu_{att} r}}{4\pi r^2 \rho}\right] B_E(\mu_{att} r) \qquad (1)$$

where:

μ_{abs} is the linear photon energy absorption coefficient (cm^{-1}) at energy E (MeV);

μ_{att} is the linear photon attenuation coefficient (cm^{-1}) at energy E (MeV);

ρ is the density of the medium (g cm^{-3}); and

r is the distance from the source (cm).

$B_E(\mu_{att} r)$ is the *energy-absorption build-up factor*, which takes into account the contribution to absorbed dose of scattered photons. Berger tabulated values of this build-up factor for a range of photon energies between 0.015 and 3 MeV, and for values of $(\mu_{att} r)$ up to 20.

$(\mu_{att} r)$ is the distance from the source expressed as the number of mean free paths of photons in the absorbing medium.

Polynomial functions can be derived from Berger's tabulated data to provide a continuous interpolation of $\Phi_E(r)$ for each of the discrete photon energies provided.

Consider an arbitrary volume of absorbing medium, of total mass M grams, which is subdivided into a large number N of volume elements which all have equal mass. If the medium is uniformly contaminated, each volume element can be considered to be either a source of radiation emission or a receptor for the absorption of emitted energy but not both. There are $N(N-1)$ such pairs in the volume; if F_E is the fraction of energy emitted within the volume that is absorbed within it, it is simple to show that:

$$F_E = \frac{M}{N^2} \sum_{i=1}^{i=N(N-1)} \Phi_E(r_i) \qquad (2)$$

where:

r_i is the separation (cm) between the i^{th} source-receptor pair.

For sufficiently large N, $N(N-1) \approx N^2$ and thus, for large N,

---

[6] 'Monte Carlo' codes are complex computer programmes which simulate the interaction of radiation with tissue in a statistical manner. In essence, the programme selects a point at random from within the source volume. From that point, a ã ray photon (or â particle, etc) is emitted in a randomly chosen direction. Subsequent interactions of the emitted photon with the matter though which it is passing are selected according to equations which describe their probability, and the resulting energy deposition is calculated. This action is repeated hundreds of thousands of times and ultimately a picture is built up of the distribution of dose.

$$F_E = M\overline{\Phi_E(r_i)} \quad (3)$$

where:

$\overline{\Ö_E(r_i)}$ represents the arithmetic average over all source-receptor pairs in the volume.

The simple relationship between Berger's point-isotropic specific absorbed fractions and the absorbed fraction for a uniformly contaminated absorbing volume allows the estimation of absorbed fractions by simple numerical methods.

Organisms are represented as ellipsoids of varying dimensions for the dosimetric calculations in this report. A simple code has been developed, using Microsoft Visual Basic for Applications (VBA) run within Excel 97, which implements the following method:

- Pairs of co-ordinates (x_1, y_1, z_1) and (x_2, y_2, z_2), both of which lie within the specified ellipsoid, are selected using a random number generator;
- The distance between the points, and hence the value of $\Ö_E(r_i)$, is calculated;
- Iteration of the above two steps, averaging of the values of $\Ö_E(r_i)$ so generated, and multiplication by the mass of the ellipsoid yields an estimate of the value of F_E.
- A few thousand iterations provide estimates of F_E with very satisfactory accuracy.

For any given ellipsoid, values of the absorbed fraction F_E can readily be obtained by this method for those ã ray photon energies which have been tabulated by Berger. To calculate the absorbed fractions for ã ray photons of any energy, these point estimates are converted into a continuous function by fitting the results to an equation:

$$F_g(E) = e^{-\left(\frac{E}{2s}\right)^n} + a.e^{-lE^m} \quad (4)$$

where:

E is the photon energy (MeV) and a, $ó$, n, $ë$ and m are fitting constants.

The form of the function was selected to provide a reliable interpolation between calculated values, and avoid the unstable behaviour that can occur when fitting data points to polynomials.

Examples of the photon absorbed fraction calculated points and fitted curve and the derived fitting constants for organism geometries used in the assessment of the aquatic ecosystem in this report, are given in Figure 5.1 and Table 5.1 respectively.

5.2.2 Beta particles

Berger provided a tabulation of the dose distribution around point sources of â particles (Berger, 1971). The process is complicated because â particles from individual radionuclides are emitted over a range of energies, with the maximum â particle energy for a given transition being two to three times higher than the average â decay energy for that transition. <A>, a given radionuclide may have several possible transitions, of differing b-energy, in the decay scheme. Even so, it is still possible to derive functions to describe the point isotropic specific absorbed fraction as a function of distance from point source, $\Ö_{â}(r_i)$. The same calculation method can then be used to estimate the absorbed fraction $F_{â}$ for a uniformly contaminated ellipsoidal volume as that for ã ray photons.

This method is computationally inefficient because the short range of â particles in tissue dictates that source-receptor pairs separated by very short distances make the major contributions to $F_{â}$. Accordingly, a large number of iterations is necessary ensure that a representative 'sample' of such short source-receptor distances is obtained. We have therefore adopted a slightly more complex, but computationally more efficient, calculation method for â absorbed fractions.

Berger (1971) tabulated values of r_p, the radius r of a sphere within which p% of the energy is absorbed from a point â source at the centre of the sphere. These values are readily transformed to values of $f_{â}(ò)$, the fractional absorption from a point â-emitter within a sphere of radius ò, equal to

r/r_{90}, around the point source. The transformation applied to the radius makes $f_{\hat{a}}(\grave{o})$ relatively independent of energy.

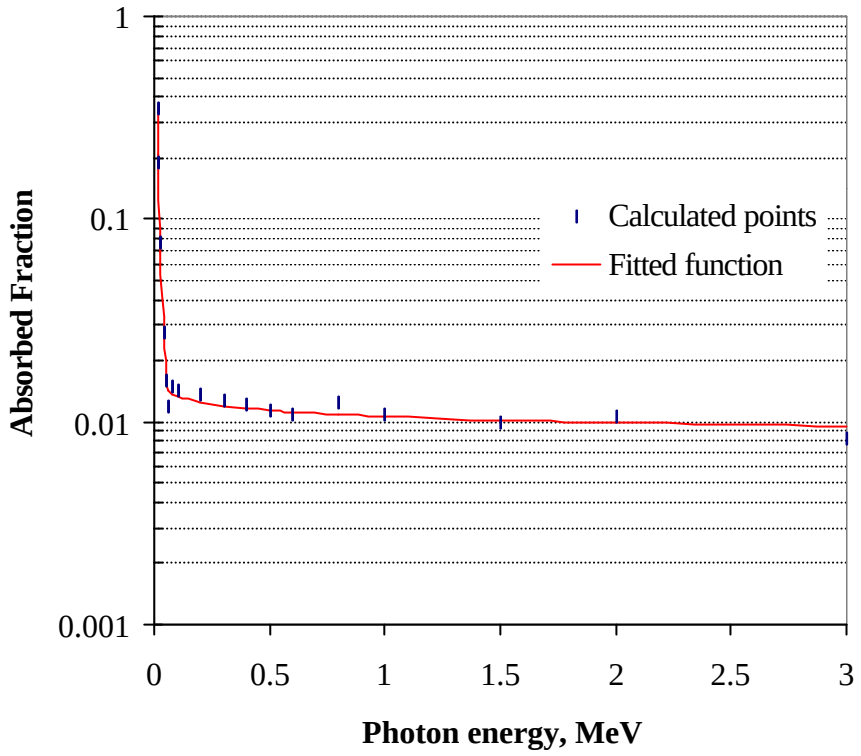

Figure 5.1 ***Calculated absorbed fractions for internally incorporated photon emitters of different energies in a benthic mollusc (dimensions 2.5cm x 1.2cm x 0.62cm)***

Table 5.1 ***Curve fitting constants for photon absorbed fractions, derived for examples of organisms in the freshwater aquatic ecosystem***

	Zooplankton	Macrophyte roots	Benthic mollusc	Fish	Waterbird
axis 1 (cm)	0.005	10	2.5	45	15
axis 2 (cm)	0.005	0.2	1.2	8.7	11
axis 3 (cm)	0.005	0.2	0.62	4.9	7.6
σ	6.96E-03	9.64E-03	1.40E-02	1.71E-02	2.87E-02
n	1.01E+00	1.24E+00	1.41E+00	8.77E-01	1.44E+00
a	7.80E-01	3.54E-03	1.08E+00	3.12E+00	1.27E-01
λ	5.64E+00	1.09E-01	4.62E+00	3.46E+00	1.30E-01
m	1.24E-02	1.23E+00	2.14E-02	1.51E-02	1.31E+00

Tabulated data from Berger have been selected for radionuclides, which undergo a single â decay with close to 100% decay probability. These are taken to be representative of any â particle emission with the same average decay energy <E>. For each value of <E>, and for the point values of ò tabulated, continuous functions are generated by curve fitting to the equation:

$$f_{<E>}(V) = 1 - e^{-lV^n} \tag{5}$$

An arbitrary volume of absorber, uniformly contaminated with radionuclides can be subdivided into a large number *N* of equal volume elements. Consider a single one of these elements, ş, acting as a point source of â emissions. The volume surrounding ş can be divided into a large number *L* of sectors, each subtending the same solid angle at ş. The fraction of energy emitted by ş, which is absorbed within the volume, is simply:

$$F_i = \left(\frac{1}{L} \sum_{k=1}^{L} f_{<E>}(V_k) \right)_i = \left(\overline{f_{<E>}(V_k)} \right)_i \tag{6}$$

where

ò$_k$ is equal to r_k/r_{90},

r_k being the distance along sector *k* to the edge of the absorbing volume.

The fraction of the total energy emitted by all the volume elements, which is absorbed within the absorbing volume itself, is then simply:

$$F = \frac{1}{N} \sum_{i=1}^{N} \left(\overline{f_{<E>}(V_k)} \right)_i = \overline{\left(\overline{f_{<E>}(V_k)} \right)_i} \tag{7}$$

β-particle absorbed fractions were calculated using VBA run in Microsoft Excel 97 (as for ã ray absorbed fractions) which:

1. Selects co-ordinates (*x, y, z*) at random within the defined ellipsoid.
2. Generates a vector through (*x, y, z*) defined by randomly selected angles *è$_1$* (rotation from *x* axis) and *è$_2$* (elevation above the *xz* plane)
3. Calculates *r*, the distance along the vector to the surface of the ellipsoid, and hence æ
4. Evaluates $f_{<E>}$(æ) using the fitted function of equation 5.
5. By repetition of steps 1 to 4, averaging the results to estimate $F_{<E>}$ as in equation 7.

This method of calculation results in rapid convergence to an accurate value for $F_{<E>}$ within a few thousand iterations.

Point estimates of $F_{<E>}$ for the specific nuclides tabulated by Berger (1971) are thus produced. These are used to generate a continuous function $F_â(<E>)$ by fitting the results to an equation of the form:

$$F_b(<E>) = \frac{1}{1 + a<E>^n} \tag{8}$$

where *a* and *n* are the fitting constants, and <*E*> is the average energy (MeV) of the â particles emitted.

Examples of the calculated β-particle absorbed fraction points and fitted curve and the derived fitting constants for organism geometries used in the assessment of the aquatic ecosystem in this report, are given in Figure 5.2 and Table 5.2 respectively.

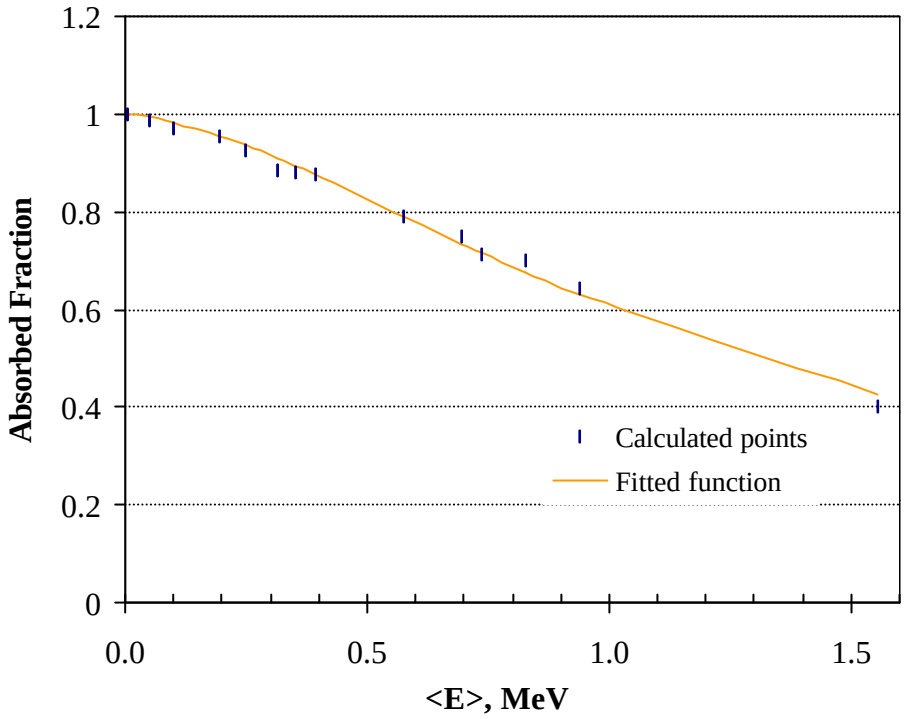

Figure 5.2 ***Calculation of absorbed fractions for internally incorporated â emitters in a benthic mollusc, for a range of average â particle energies***

Table 5.2 ***Curve fitting constants for â particle absorbed fractions, derived for organisms in the aquatic ecosystem***

	Zooplankton	Macrophyte roots	Benthic mollusc	Fish	Waterbird
axis 1 (cm)	0.005	10	2.5	45	15
axis 2 (cm)	0.005	0.2	1.2	8.7	11
axis 3 (cm)	0.005	0.2	0.62	4.9	7.6
A	1.68E+04	4.04E+00	6.52E-01	6.20E-02	5.10E-02
N	1.58E+00	1.56E+00	1.63E+00	1.36E+00	1.51E+00

5.2.3 Alpha particles

The range of á particles in living tissue is very small - typically, about 50 microns. Relatively simple equations for the energy loss per unit path length can be used to estimate dose distributions around a point source of á particles (e.g. Harley and Pasternack, 1972). Knowledge of the spatial distribution of á emitting radionuclides within the tissue of interest is required to make use of such calculations in any dose assessment. The only real application of these techniques has been in the dosimetric model for the respiratory tract in human dosimetry (ICRP, 1994).

The assumption has been made that internally incorporated radionuclides are distributed uniformly within the organisms of interest for dose calculations in this report. Therefore, the absorbed fraction for á particles is unity for all organisms except bacteria. The absorbed fraction for internally incorporated á, â and ã emitters is zero for bacteria, as their dimensions are around a few microns. Doses to micro-organisms are assumed to be equal to the absorbed dose (including the absorbed dose from á emissions) in the medium within which they are incorporated, according to the protocol described below.

5.3 Dose rate per unit concentration values

The approaches to wildlife dosimetry summarised in the introduction to this Chapter all focus on estimating the radiation dose rate to exposed organisms. There is a direct correlation between the concentration of a particular radionuclide in the tissues of an organism, or in the medium surrounding it, and the radiation dose rate to the organism. Environmental transfer models for radionuclides are conventionally aimed at estimating radionuclide concentrations in organisms and the surrounding environmental media, so it is a natural extension to incorporate wildlife dosimetry into these models by introducing factors which relate radionuclide concentrations to the resulting radiation dose rate.

5.3.1 Internally incorporated radionuclides

In this report ecosystems are represented by simple equilibrium models for the concentration of radionuclides in soil, sediment, water and biota. 'Dynamic' models, which represent the rates of transfer of radionuclides between different ecosystem components and the consequent time variation of radionuclide concentrations, would present a more comprehensive description of the ecosystems. However, this type of model is simply not presently available for natural ecosystems, with the exception of a few models developed for ^{137}Cs transfer following the Chernobyl accident.

The dose rates delivered to organisms are evaluated from the calculated radionuclide concentrations. This is derived by calculating the dose to each organism per unit concentration of each internally incorporated radionuclide.

For each radionuclide, data for the energy and yield of â particle, electron, photon and á particle emissions have been extracted from the literature (ICRP, 1983).

Each organism is characterised by particular dimensions and by corresponding values of the fitting constants in equations (4) and (8) above, for $F\tilde{a}(E)$ and $F_{\hat{a}}(<E>)$ respectively.

For each radionuclide and each set of organism dimensions overall absorbed fractions for photon and â particle emissions are calculated as:

$$\overline{F_g} = \frac{\left(\sum_i p_i F_g(E_i)\right)_g}{\left(\sum_i p_i E_i\right)_g}$$
$$\overline{F_b} = \frac{\left(\sum_i p_i F_b(<E_i>)\right)_b}{\left(\sum_i p_i (<E_i>)\right)_b} \qquad (9)$$

where:

E_i denotes the energy (MeV); and

p_i denotes the fraction of disintegrations which give rise to a photon or beta particle of energy E.

Electron emissions are included within the summation for â emissions. As noted above, absorbed fractions for á emissions are assumed to be zero for bacteria and unity for all other organisms.

The corresponding Dose Per Unit Concentration (DPUC) values then become:

$$
\begin{aligned}
DPUC^{\text{int}}_{lowb} &= 5.77\times10^{-4}\times\overline{F_{lowb}}\times\sum_i\left(p_iE_i\right)_{lowb}\\
DPUC^{\text{int}}_{bg} &= 5.77\times10^{-4}\times\left[\overline{F_b}\times\sum_i\left(p_iE_i\right)_b+\overline{F_g}\times\sum_i\left(p_iE_i\right)_g\right]\\
DPUC^{\text{int}}_{a} &= 5.77\times10^{-4}\times\overline{F_a}\times\sum_i\left(p_iE_i\right)_a\\
DPUC^{\text{int}}_{total} &= DPUC^{\text{int}}_{lowb}+DPUC^{\text{int}}_{bg}+DPUC^{\text{int}}_{a}
\end{aligned}
\qquad (10)
$$

DPUC units are μGy h^{-1} per Bq kg^{-1}, and the constant 5.77×10^{-4} is the conversion factor from MeV s^{-1} to μJ h^{-1}.

The subscript *lowb* denotes â particles and electrons with an average energy less than 10 keV; the subscript *b* refers to all other â particles. These two â particle components are kept separate so that different radiation weighting factors can be applied to them, as described in Section 5.5 below.

Use of these expressions for DPUC results in the calculation of absorbed dose with no weighting factors for radiation type applied, i.e. *unweighted absorbed dose.* The introduction of radiation weighting factors is described in Section 5.5 below.

5.4 Radionuclides in soil, sediment and water

External doses to organisms from radionuclides present in soil, sediment or water are calculated using a variant of the simple formula for a uniformly contaminated isotropic infinite absorbing medium:

$$
\begin{aligned}
DPUC^{ext}_{lowb} &= 5.77\times10^{-4}\times\left(1-\overline{F_{lowb}}\right)\times\left(\sum_i p_iE_i\right)_{lowb}\\
DPUC^{ext}_{bg} &= 5.77\times10^{-4}\times\left[\left(1-\overline{F_b}\right)\times\left(\sum_i p_iE_i\right)_b+\left(1-\overline{F_g}\right)\times\left(\sum_i p_iE_i\right)_g\right]\\
DPUC^{ext}_{a} &= 5.77\times10^{-4}\times\left(1-\overline{F_a}\right)\times\left(\sum_i p_iE_i\right)_a\\
DPUC^{ext}_{total} &= DPUC^{ext}_{lowb}+DPUC^{ext}_{bg}+DPUC^{ext}_{a}
\end{aligned}
\qquad (11)
$$

These equations:

- approximate the dose rate to an organism immersed in an infinite contaminated medium,
- neglect density differences between the organism and the medium,
- allow for self shielding by the organism itself, and
- average the dose rate throughout the volume of the organism.

Equations (11) have been used to calculate external dose for organisms underground, buried in sediment or free swimming in the water column; the relevant concentrations being those in the soil, water or sediment media as appropriate. For the case of an organism exposed on the ground surface or at the sediment/water interface we have taken the dose per unit concentration to be half of that for exposure underground or buried in sediment.

5.5 Relative Biological Effectiveness (RBE)

As discussed above, the system of dosimetry for humans is well defined, however such a system for wildlife has not yet been widely agreed. It cannot be assumed that the Relative Biological Effectiveness (RBE) values applied to α, β or γ radiation, and hence radiation weighting factors (w_r), in human dosimetry are applicable to wildlife due to the vast differences in physiology between, for example, humans and invertebrates. Furthermore, it cannot be assumed that the RBE value for a radiation type will be the same across different biota e.g. fish and mammals or across different

biological endpoints. However, a system using radiation weighting factors, derived from RBE values, is required to calculate a "dose equivalent for flora and fauna" from absorbed doses in parallel to that of human dosimetry, in order to develop a framework to assess the risk posed to wildlife from ionising radiation (Pentreath, 1999).

In addition, little, if any, information is available on the effects of radiation exposure in different body organs of wildlife species (i.e. effective dose). Effective dose is used in human radiobiological protection to deal with situations where the body is not uniformly exposed. This requires knowledge of the distribution of radionuclides within the organism's body. Wildlife dosimetry is limited in this respect so a uniform distribution has to be assumed when calculating dose rates to biota. When considering internal doses from low penetrating radiation with such as α and β this can lead to underestimation of the dose in the immediate vicinity of the deposition site and an overestimation of the dose further away from the deposition site.

This Section describes the need for using RBE of different types of radiation for wildlife. Recommendations for wildlife specific radiation weighting factors for á, â and ã radiations are made. Many studies deriving RBE values, have used cytogenetic and molecular endpoints. The techniques used to assess cytogenetic and molecular endpoints are described in Appendix 2 as they can also be used to evaluate the impact of exposure to environmental contaminants. They are commonly referred to as 'biomarker' techniques.

5.5.1 Relative Biological Effectiveness (RBE) and radiation-weighting factors in human radiation protection

The extent of damage to living cells caused by different types of radiation, for a given absorbed dose, is greatly dependent on their linear energy transfer (LET). The concept of relative biological effectiveness (RBE) provides a quantitative comparison between biological damage produced by radiations of different LET. Experimentally, RBE is the ratio of doses from two different radiations that produce equal levels of biological damage in the same system:

$$RBE = \frac{Absorbed\ dose\ of\ reference\ (X\ or\ g)\ radiation\ to\ produce\ given\ effect}{Absorbed\ dose\ of\ radiation\ of\ type\ in\ question\ to\ produce\ same\ effect}$$

The purpose of this review is to recommend radiation-weighting factors, which might be applied to non-human biota, for the purposes of impact assessment from the effects of exposure to ionising radiation. In order to recommend any radiation-weighting factors, an understanding of RBEs in humans must be first described.

A RBE value must always be referenced to either a particular type or quality of radiation and a particular biological endpoint, i.e. a specific measure of biological damage. The reference radiation is usually 250 kVp X-rays, ^{137}Cs ã rays or ^{60}Co ã rays. Any factor which affects the two-dose response curves differently, e.g. dose, dose rate, LET, endpoint, gender, age, will affect the ratio between any pair of points on the curves and hence alter the calculated RBE.

An important distinction arises between:

- stochastic effects, in which the probability of the effect increases with radiation dose, and
- non-stochastic or deterministic effects in which the effect is only manifested above a particular level of dose.

Stochastic effects, which include carcinogenesis, are of particular importance in human radiation protection. Other effects such as general life shortening, or reduction in fertility, are deterministic in nature and are of less significance in human radiation protection (see Chapter 3).

For many stochastic endpoints RBE values increase as the dose and dose rate decrease. This is due mainly to a decrease in the slope of the dose response curve of the reference radiation (NCRP Report 104, 1990) as illustrated in Figure 5.3. It is therefore vital that experimental data for RBEs be

interpreted from both the endpoint studied and the experimental conditions (in particular levels of dose-rate and dose).

Human radiation protection uses 'quality factors' or 'radiation weighting factors' in calculating radiation doses, to reflect the differing biological effectiveness of different radiation types. These factors are recognised to be a broad interpretation of the underlying RBE values, with most weight being placed on stochastic effects (e.g. carcinogenesis) because of their importance.

Table 5.3 Comparison of LET and radiation weighting factor for humans

LET (keV μm^{-1})	Radiation weighting factor
<10	1
10-100	$0.32 \times LET - 2.2$
>100	$300/\sqrt{LET}$

α particles typically exhibit LET values in the range 175 to 250 keV μm^{-1}, and are assigned a radiation-weighting factor of 20, consistent with this relationship. ICRP do not make any distinction between â radiation emitters of differing energy, although for some β emitters such as tritium, the low energy of the emitted â particles does result in LET values close to, or above, the 10 keV μm^{-1} 'threshold' in Table 5.3, as illustrated in Figure 5.3.

Radiation weighting factors are set to 20 for á and unity for ã and â radiation in human radiation protection.

5.5.2 General considerations in recommending radiation weighting factors for the protection of biota

There is a general consensus that protection of non-human biota should be applied mainly at the level of the population rather than the individual; one implication of this is that deterministic effects, such as reduction of fertility, are likely to be of importance. Stochastic effects on individuals, e.g. malignancy, are of little consequence unless they affect a significant proportion of individuals in the population. However, the accumulation of heritable mutations in a population could have significant effects in the longer term. Stochastic effects should therefore also be taken into account as a precautionary approach.

Compared with the diversity of the non-human biota, there is relatively little experimental data relating to RBEs in organisms other than mammals. Radiation weighting values for use in the protection of biota have been recommended for á and â radiation and for tritium by a number of authors and organisations. While there is a measure of agreement over â emitters, the recommendations vary for á radiation due largely to different interpretations of the same evidence. These issues are explored below to arrive at considered recommendations on radiation weighting factors for interim application in the protection of biota.

- **Weighting factors for á radiation in the protection of biota**

UNSCEAR (1996) suggests that a weighting factor for á radiation of 5 is appropriate for non-human biota. This is on the basis that deterministic effects will be of greater significance than they are for human protection and that a lower factor than that used for humans should therefore apply. Pentreath (1996) advances a similar argument in respect of aquatic organisms, although no specific value is recommended. More recently, Kocher and Trabalka (2000) have argued that experimentally determined RBEs for deterministic effects lie in the region 5 to 10 and weighting factors in this range would therefore be appropriate for use in the protection of biota.

Both Woodhead (1984) and Blaylock et al. (1993) have suggested a weighting factor of 20 for aquatic organisms, on the grounds that this value incorporates the spectrum of effects, including stochastic

effects, which are of relevance in human radiation protection. This value may of course be conservative in respect of deterministic effects.

A Canadian review of RBEs for the protection of biota (Environment Canada, 2000), has taken a different view from UNSCEAR. The review emphasises evidence that the biological damage caused by á particles is fundamentally different to that due to low LET radiation. In particular, it suggests that á irradiation induces a form of genetic instability in human and mouse haemopoietic stem cells, and that this instability persists throughout several generations of daughter cells (Kadhim *et al.* 1992, described in to Appendix 2). Environment Canada now recommends a radiation-weighting factor for á radiation of 40. While this can be seen as a cautious approach in the face of new evidence, the same conclusion has not yet been reached for human protection. Moreover, higher reported RBE values can derive in part from a lower effectiveness of the reference radiation at low dose rates rather than a higher effectiveness of á radiation.

On the basis of the available evidence, this report **recommends a weighting factor for á radiation of 20**. This is based on the judgement that:

- the value for human protection is derived, partly, on data from other mammals, which are the most radiosensitive species, and that
- there is insufficient evidence from other non-human biota to influence this conclusion.
- the value of 20 is likely to be conservative in respect of deterministic effects.

- **Weighting factors for â radiation in the protection of biota**

Particular attention has been paid to the effects of tritium due to its incorporation into water and consequent environmental distribution and bioavailability. Straume and Carsten (1993) reviewed tritium data on a range of species and endpoints and concluded that a radiation-weighting factor of 3 is appropriate for this â emitter as tritiated water (HTO). Higher RBEs were found when exposure was to tritiated nucleotides. Since this review, RBEs for HTO of 2.7-3.1 have been reported for induction of chromosomal aberrations in human lymphocytes and bone marrow cells (Tanaka *et al.* 1994); 1.4 for gene mutation frequency in *Drosophila* (Fosset *et al.* 1994); and 1.2 ± 0.3 for the induction of myeloid leukaemia in mice (Johnson *et al.* 1995).

Figure 5.4 shows the calculated LET values for mono-energetic electrons. The average â energy for tritium, at 6 keV, lies just at the point where decreasing energy produces a significant increase in the LET value. Figure 5.5 shows the corresponding radiation weighting factors calculated from the relationship in Table 5.3, as recommended by ICRP. At 6 keV μm^{-1}, it would appear that the LET for tritium â radiation is too low to account for the experimentally observed RBEs. However this value for LET refers to the <u>average</u> â energy for tritium; as for all other â emitters, tritium emits â particles with a spectrum of energies up to a maximum value, the maximum being about threefold higher than the average energy. The energy distribution is skewed, with more than half the decays emitting â particles with energies less than the average value. Therefore a proportion of the â particles emitted by tritium will have LET values well in excess of 6 keV μm^{-1}. Prestwich and Kwok (1993) have calculated the radiation-weighting factor for tritium âs, according to the ICRP 1991 formulation, by integrating the value across the whole energy spectrum for tritium âs. They report a weighting factor of 1.9 ± 0.2 on this basis.

It is not clear from the literature whether the higher experimental RBEs for tritium are due to its LET (about 6 keV μm^{-1} compared with 0.24 keV μm^{-1} for ã rays and 175 keV μm^{-1} for á particles), or its ability to exchange with hydrogen in biomolecules (such as proteins and DNA). Environment Canada has recommended a radiation-weighting factor of 3 for tritium, based largely on Straume and Carsten (1993). UNSCEAR (1996) only makes a general recommendation for all â emitters of 1.

However, given that calculations based on LET values alone suggest a weighting factor of 2 would be justifiable for tritium â radiation, it would be prudent to assume that the experimental RBEs for tritium reflect the elevated LET values for low energy â particles and electrons, and to apply a weighting

factor greater than unity to all such â particles and electrons, regardless of the radionuclide from which they originate.

A radiation-weighting factor of 3 for mono-energetic electrons, or â particles of average energy, less than 10 keV is recommended.

A weighting factor of 1 is recommended for all other â particles and electrons.

Gamma and x-rays are conventionally the reference radiations so the weighting factor for gamma is always 1 (i.e. referenced to itself).

More experimental data are desirable to increase confidence in the weighting factors for both low energy â radiation and á radiation, and particularly for environmentally relevant deterministic endpoints.

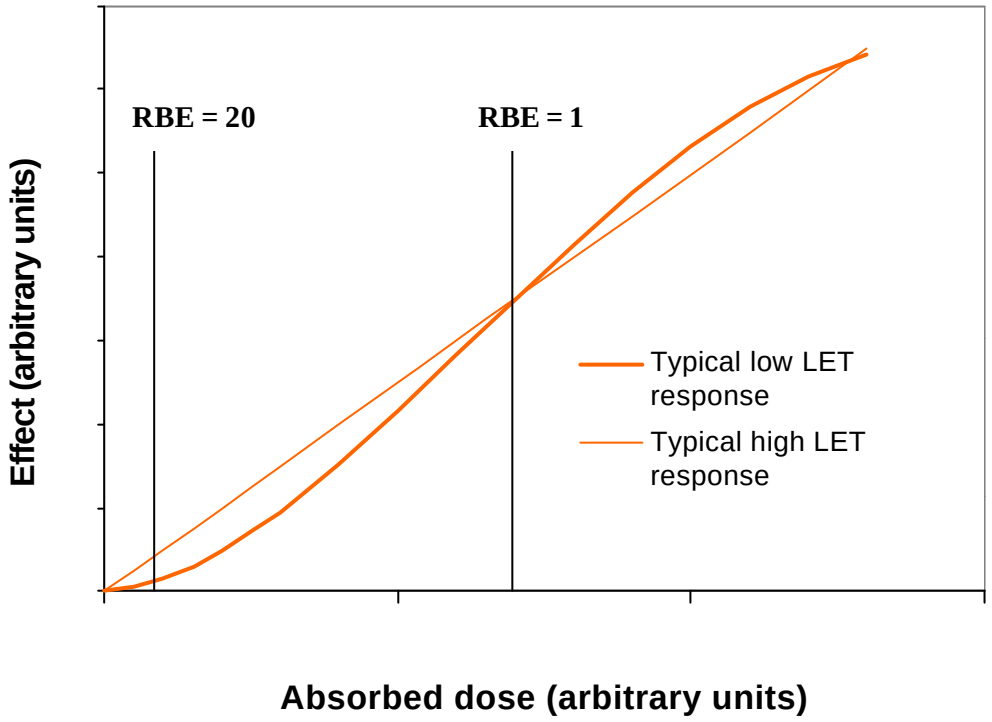

Figure 5.3 Schematic of RBE varying with dose due to differing dose-response of low and high LET radiation

Low LET radiation (e.g. X or ã radiation) often shows a non-linear ('linear-quadratic') dose response, in which the increase in effect with dose is initially quite small, increasing at higher doses and sometimes reducing again at yet higher doses. High LET radiation (e.g. á radiation) commonly shows a uniform linear increase in effect with dose. Consequently, the relative biological effectiveness of the two types of radiation will vary with dose (and often dose rate).

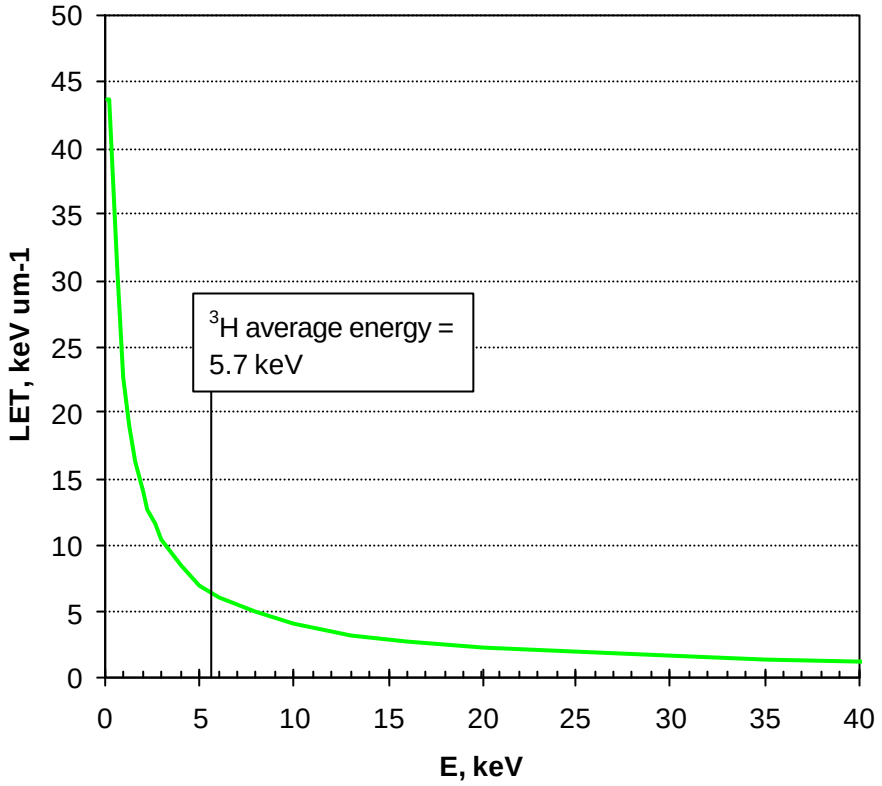

Figure 5.4 LET as a function of energy for mono-energetic electrons

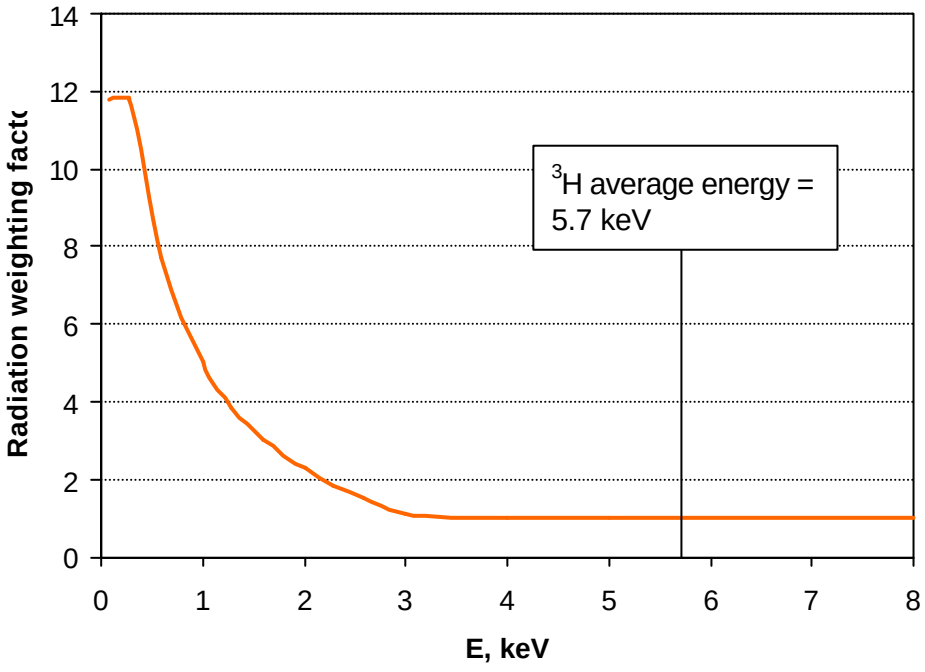

Figure 5.5 Radiation weighting factor (according to ICRP formulation) for mono-energetic electrons, as a function of energy

Based on the review in Section 5.5, a radiation weighting factor of 3 is recommended for interim application to doses arising from low energy â particles and electrons (with energy less than 10 keV). Similarly, a weighting factor of 20 is recommended for interim application to doses arising from á particles. These weighting factors are introduced into the calculation as simple multipliers for the appropriate component of the dose per unit concentration value, e.g.

$$\begin{aligned}\left\lfloor DPUC_{lowb}^{\text{int}} \right\rfloor_w &= DPUC_{lowb}^{\text{int}} \times w_{lowb} \\ \left[DPUC_{a}^{\text{int}}\right]_w &= DPUC_{a}^{\text{int}} \times w_a \\ \left[DPUC_{total}^{\text{int}}\right]_w &= \left[DPUC_{lowb}^{\text{int}}\right]_w + DPUC_{bg}^{\text{int}} + \left[DPUC_{a}^{\text{int}}\right]_w\end{aligned} \qquad (12)$$

where:

w_{lowb} and w_a are the radiation weighting factors of 3 and 20 for low energy â particles and á particles, respectively, as explained above.

The spreadsheets, which accompany this report, have the facility to change these factors.

Use of the DPUC values defined in equations (12) results in the calculation of (mean, whole organism) *weighted absorbed dose.*

5.6 Dose calculations

The dose calculations for the terrestrial and aquatic environments are similar in principle but differ in detail because concentration factors for radionuclides in the terrestrial environment are calculated relative to air or soil, whereas those for the aquatic environment are calculated relative to water.

5.6.1 Terrestrial environment

For the radionuclides ^{3}H, ^{14}C, ^{35}S, ^{90}Sr, ^{129}I, ^{137}Cs, ^{226}Ra, ^{238}U, and $^{239/240}$Pu, concentration factors for biota are referenced to surface soil. Equivalent dose rates $\dot{H}$ to biota are simply calculated as:

$$\dot{H} = \sum_i (C_i^{soil} \times CF_i^{soil} \times DPUC_{total,i}^{\text{int}}) + \left(\left(f_{soil} + 0.5\, f_{surface}\right) C_i^{soil} \times DPUC_{total,i}^{ext}\right) \qquad (13)$$

where:

$\sum_i$ represents summation over all nuclides;

C^{soil} is the concentration of the radionuclide in surface soil;

CF^{soil} is the concentration factor for the organism referenced to soil;

f_{soil} is the fraction of time the organism spends under the soil surface; and

$f_{surface}$ is the fraction of time the organism spends on the ground surface.

It is assumed that organisms receive no external dose during the fraction of their time spent above the ground surface, e.g. birds flying or roosting.

For the radionuclides ^{3}H, ^{14}C and ^{35}S a slightly different approach is used because these radionuclides do not accumulate readily in the soil. Concentration ratios between air on the one hand, and soil and biota on the other, are estimated from the concentration ratios for the stable elements or (for ^{35}S) field studies reported in the literature. The dose rates from these radionuclides are then calculated as:

$$\dot{H} = \sum_i (C_i^{air} \times CF_i^{air} \times DPUC_{total,i}^{\text{int}}) + \left(\left(f_{soil} + 0.5\, f_{surface}\right) C_i^{soil} \times DPUC_{total,i}^{ext}\right) \qquad (14)$$

where:

$\sum_i$ represents summation over all nuclides;

C^{air} is the concentration of the radionuclide in air;

C^{soil} is the concentration of the radionuclide in surface soil, calculated from the air concentration and the relevant concentration ratio;

CF^{air} is the concentration factor for the organism referenced to air;

f_{soil} is the fraction of time the organism spends under the soil surface; and

$f_{surface}$ is the fraction of time the organism spends on the ground surface.

5.6.2 Aquatic environment

For the radionuclides ^{3}H, ^{14}C, ^{99}Tc, ^{90}Sr, ^{129}I, ^{137}Cs, ^{226}Ra, ^{238}U, and $^{239/240}$Pu, concentration factors for biota are referenced to water, and dose rates H are calculated as:

$$\dot{H} = \sum_i (C_i^{water} \times CF_i^{water} \times DPUC_{total,i}^{\text{int}}) + \left(\left(\left(f_{sediment} + 0.5 f_{surface}\right) C_i^{sediment} + \left(f_{water} \times C_i^{water}\right)\right) \times DPUC_{total,i}^{ext}\right) \quad (15)$$

where:

C^{water} and $C^{sediment}$ are the radionuclide concentrations in water and sediment respectively;

CF^{water} is the concentration factor for biota referenced to water;

$f_{sediment}$ is the fraction of time spent buried in sediment;

$f_{surface}$ the fraction of time spent on the sediment surface; and

f_{water} the fraction of time spent free swimming in the water column or on the water surface.

5.7 Calculations of doses to biota using the accompanying spreadsheet applications

Equations (13) to (15) have been programmed into three Microsoft Excel spreadsheets, to allow calculation of dose rates to biota in terrestrial and aquatic environments. These equations have been verified in accordance with a ISO 9001quality system. A full operating guide for these spreadsheets is provided as Appendix 3, and a detailed assessment using the spreadsheets is described in Section 6.5.1. The spreadsheets are provided on a CD-ROM situated at the back of this R&D Publication 128.

5.7.1 Assumptions and applicability of the method

As explained above, the dose calculation method employs a number of inherent assumptions:

- Organisms are represented as ellipsoids
- Concentrations of radionuclides in biota are calculated using simple equilibrium concentration ratios between biota and water, soil or air.
- Radionuclides are considered to be distributed uniformly through all tissues of the animal or plant.
- Resulting absorbed doses, both internal and external, are calculated as an average throughout the volume of the organism.
- Doses are calculated as dose rates from equilibrium concentrations of radionuclides in biota.

- Organisms receive external dose at a reduced rate during the fraction of their time spend above ground surface, e.g. birds flying or roosting
- Absorbed fractions for á emissions are assumed to be zero for bacteria and unity for all other organisms.
- Calculated doses to micro-organisms are equal to the absorbed dose in the soil or sediment in which they are located.

With regard to the applicability of the method, the most important assumption is that concentrations in biota are in equilibrium with concentrations in the surrounding environmental media. The method cannot be used to assess doses to biota in situations where the concentrations of radionuclides in the surrounding environmental media are changing rapidly.

Generally, it is considered that aquatic organisms equilibrate quite rapidly with concentrations of radionuclides in the water column, so that it would be quite reasonable to use annually averaged concentrations of radionuclides in the aquatic environment as the basis of an assessment. However, equilibration can occur much more slowly in terrestrial ecosystems. Further, the simple assumption of equilibration between radionuclide concentrations in soil and biota cannot adequately represent the complex dynamics of the contamination of vegetation, soil and biota whilst there is continuing deposition of radionuclides from the atmosphere.

The calculation method as provided is intended for use in a stable contaminated environment, where radionuclide burdens have accumulated over an extended period of time. For prospective assessments of the effects of proposed discharges of radionuclides to the environment, we recommend that predictive models should be used to estimate radionuclide concentrations in soil, air or water as appropriate after discharge for 50 years at the proposed discharge rates. The concentrations so estimated can then be used as the basis for calculating doses to biota.

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6. Impact Assessment of Ionising Radiation on Wildlife - Approach and Scenarios

6.1 Introduction

As stated there is no international consensus on the approach to impact assessment of ionising radiation on wildlife. Assessment of the risk to all potentially exposed individual biota would pose serious difficulties and thus, a simplified approach is required with the aim of ensuring that risks to wildlife are negligible based on best available knowledge of dosimetry (Chapter 5) and exposure routes (Chapter 2) using conservative assumptions. The approach is proposed for practitioners who carry out impact assessments of ionising radiation, supported by Excel spreadsheets. An operating guide for the spreadsheets is provided in Appendix 3, with a 'colour version' available on the attached CD-ROM.

The approach described has been broadly based on those proposed by Pentreath (1998; 1999) and by Woodhead (2000a, b; 2001) and can be summarised in the following steps:

- Definition of a range of reference ecosystems, populated by 'reference organisms' (see Section 6.2.2);
- Construction of a database of concentration factors for selected radionuclides;
- Construction of dosimetric models as described in Chapter 5.

The assessment itself then follows as:

- Selection of the type of assessment, i.e. prospective (for new nuclear installations or to assess the likely impact of changes to authorisations) or retrospective (assessing the actual impact of authorised discharges into the environment);
- Determination of wildlife/ecosystem at risk;
- Determination of data requirements for the assessment;
- Execution of the assessment using the spreadsheet(s), using site specific data if applicable;
- Interpretation of the results, taking account of uncertainties;
- Evaluation of the assessment based on the output from the dose models comparing the estimated doses to known effects (Chapter 3) and guideline dose limits (Table 3.1).

Given the potential scope of scenarios where authorised discharges of radioactive materials may occur and the wide range of wildlife that may be impacted it is necessary to simplify the problem using an approach which identifies representative radionuclides and wildlife most likely to be impacted by ionising radiation. Therefore the assessment approach adopted within this report is limited to a number of reference species (Section 6.2.2) and radionuclides (Section 6.2.3). Given these limitations, the information contained in the spreadsheets has been obtained from the literature describing a wide range of studies of radionuclide uptake and impact on wildlife. Using the default values provided, it is possible to carry out a generic impact assessment; alternatively, site specific information may be incorporated into the assessment process to derive more realistic dose estimates for a particular situation. Indeed, it is recommended that site specific characteristics be incorporated into any assessment process especially if there are scenarios where rare or endangered species may be present. In these circumstances, the recommended guideline values for dose limits provided in Table 3.1 may need to be revised in line with the sensitivities associated with a particular site or species.

6.2 Prerequisites and assumptions

6.2.1 Selected ecosystems

A scoping exercise considered ecosystem types likely to be significantly impacted by authorised releases of radioactive materials in England and Wales; three ecosystems were identified:

- Freshwater;
- Estuarine/marine;
- Terrestrial (i.e. coastal grassland)

6.2.2 Reference Organisms

By using the reference organism approach, a standard set of models (Chapter 5) and databases of information (Section 6.5.3) can be developed for comparison purposes. This approach has already been successfully adopted in the marine environment (Pentreath and Woodhead, 1988).

The choice of organisms were based on Woodhead (2000a), but modified following the first workshop of the FASSET project (February 2001). Reference organisms were defined as:

> *"a series of imaginary entities that provides a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects. It is important to recognise that they are not a direct representation of any identifiable animal or plant species."*

These reference organisms have been selected based on consideration of ecological-, and radio-, sensitivity. The selected organisms are given in Table 6.1. Tables 6.2 to 6.4 describe the dimensions of the organisms, which were gathered from the literature. These dimensions define the approximate shape of an average animal or plant usually as an ellipsoid. Calculations were made to check that the dimensions obtained were appropriate for the mass of the organism given in the literature.

Radioecological data for species of similar size to the reference organisms can be used depending upon the locality of the site under assessment e.g. tropical, temperate etc. In this way, site specific information can be included in the assessment for different situations. For example, radioecological data on bank voles or meadow voles may be used in assessments within the UK or US respectively.

Table 6.1 Selected reference organisms for each ecosystem

Freshwater	Estuarine/marine	Terrestrial
Bacteria	Bacteria	Bacteria
Macrophyte	Macrophyte	Lichen
Phytoplankton	Phytoplankton	Tree
Zooplankton	Zooplankton	Shrub
Benthic Mollusc	Benthic Mollusc	Herb
Small Benthic Crustacean	Small Benthic Crustacean	Seed
Large Benthic Crustacean	Large Benthic Crustacean	Fungus
Pelagic Fish	Pelagic Fish	Caterpillar
Benthic Fish	Benthic Fish	Ant
Amphibian	Fish Egg	Bee
Duck	Seabird	Woodlouse
Aquatic Mammal	Seal	Earthworm
	Whale	Herbivorous Mammal
		Carnivorous Mammal
		Rodent
		Bird
		Bird Egg
		Reptile

Table 6.2 Freshwater reference organism ellipsoid dimensions

Reference organism	Reference dimension (cm)	Mass (kg)	Reference
Benthic bacteria	5.0E-05 x 5.0E-05 x 5.0E-05 [b]		*Hammer (1986)*
Phytoplankton [a]	0.005 x 0.005 x 0.005	6.50E-11	*IAEA (1976)*
Zooplankton [a]	0.62 x 0.31 x 0.16	1.60E-05	*IAEA (1988); NCRP (1991)*
Macrophyte	10 x 0.2 x 0.2	2.10E-04	*Patton et al. (2001)*
Benthic mollusc [a]	2.5 x 1.2 x 0.62	1.00E-03	*IAEA (1988); NCRP (1991)*
small benthic crustacean [a]	0.62 x 0.31 x 0.16	1.60E-05	*IAEA (1988); NCRP (1991)*
large benthic crustacean [a]	3.1 x 1.6 x 0.78	2.00E-03	*IAEA (1988)*
Benthic fish [a]	45 x 8.7 x 4.9	1.0E+00	*IAEA (1988)*
Pelagic fish [a]	45 x 8.7 x 4.9	1.0E+00	*IAEA (1988)*
Fish egg [a]	diameter 0.08, 0.12 and 0.2	2.7E-07, 9.1E-07 and 4.2E-06	*IAEA (1979)*
Amphibian	10 x 6 x 4	1.25E-02	
small aquatic mammal	10 x 2 x 2	2.10E-03	*Patton et al. (2001)*
Duck [a]	solid tissue at an average density of 0.8 g cm(-3): 15 x 11 x 7.6	Total : 0.6 0.55	*NCRP (1991)*
	Feathers at an average density of 0.33 g cm(-3) and overall dimensions: 21 x 16 x 11	0.05	

unless otherwise specified, organisms are assumed to have a uniform body density of 1 g cm^{-3}

[a] drawn from Woodhead (Technical report P350)

[b] mean of range 5.0E-04 - 5.0E-06

6.2.3 Selection of radionuclides

Table 6.5 identifies the chosen radionuclides for assessment, based on their presence and importance in authorised discharges in England and Wales.

Their properties include:

- biologically mobility;
- released in large quantities;
- shown to accumulate in certain species;
- radiologically significant dose contributors.

For radionuclides that have progeny with a short half-life, the dose models assume that the progeny will be in equilibrium with the parent atom. Of the radionuclides listed in Table 6.5, this is only affects ^{90}Sr.

^{210}Po (aquatic ecosystems) and ^{226}Ra (terrestrial ecosystem) were included to illustrate the significance of naturally occurring radionuclides in any assessment. Naturally occurring radionuclides may be present through technological enhancement and can be significant contributors to dose.

Table 6.3 Estuarine/marine reference organism ellipsoid dimensions

Reference organism	Reference dimensions (cm)	Mass (kg)	Reference
Benthic bacteria	5.0E-05 x 5.0E-05 x 5.0E-05 [b]		*Hammer (1986)*
Phytoplankton [a]	diameter 5E-03	6.50E-11	*IAEA (1976)*
Macrophyte	10 x 0.2 x 0.2	2.10E-04	*Patton et al. (2001)*
Zooplankton a	0.62 x 0.31 x 0.16	1.60E-05	*IAEA (1988); NCRP (1991)*
Benthic mollusc (Mussel) [a]	2.5 x 1.2 x 0.62	1.00E-03	*IAEA (1988); NCRP (1991)*
small benthic crustacean (Shrimp) [a]	0.62 x 0.31 x 0.16	1.60E-05	*IAEA (1988); NCRP (1991)*
large benthic crustacean (Lobster) [a]	3.1 x 1.6 x 0.78	2.00E-03	*IAEA (1988)*
Pelagic fish (e.g. Cod) [a]	45 x 8.7 x 4.9	1.00E+00	*IAEA (1988)*
Benthic fish (e.g. Plaice) [a]	45 x 8.7 x 4.9	1.00E+00	*IAEA (1988)*
fish egg [a]	diameter 0.08, 0.12 and 0.2	2.7E-07, 9.1E-07 and 4.2E-06	*IAEA (1979)*
seal [a]	180 x 35 x 19	5.80E+01	*IAEA (1998)*
whale [a]	450 x 87 x 48	1.00E+04	*IAEA (1998)*
seabird [a]	solid tissue at an average density of 0.8 g cm(-3): 15 x 11 x 7.6	Total : 0.6 0.55	*NCRP (1991)*
	Feathers at an average density of 0.33 g cm(-3) and overall dimensions: 21 x 16 x 11	0.05	

unless otherwise specified, organisms are assumed to have a uniform body density of 1 g cm^{-3}

[a] drawn from Woodhead (Technical report P350, 2000)

[b] mean of range 5.0E-04 - 5.0E-06

For example, Aarkrog *et al.* (1997) estimated the dose to humans from consumption of seafood containing ^{210}Po and ^{137}Cs, and demonstrated that 30,000 man Sv came from ^{210}Po compared with 160 man Sv from ^{137}Cs.

^{210}Po was selected for the aquatic environments, but because of a lack of information in the literature on it for the terrestrial ecosystem ^{226}Ra was selected instead.

Basic information on the typical source, chemical properties and management strategies for the radionuclides selected is given in Table 6.6.

Table 6.4 Terrestrial reference organism ellipsoid dimensions

Reference Organism	Reference dimensions (mm)	Mass (kg) Fresh weight (FW)	Reference
Lichen	100 x 5 x 5	1.31E-03	
Moss	100 x 10 x 5	2.62E-03	
Tree (root)	100 x 2 x 2	2.10E-04	*Patton et al. (2001)*
Shrub (root)	100 x 2 x 2	2.10E-04	*Patton et al. (2001)*
Herb (root)	100 x 2 x 2	2.10E-04	*Patton et al. (2001)*
Germinating Seed	6 x 1 x 1	1.80E-06	*Copplestone, pers comm*
Fungal fruiting body	30 x 15 x 10	2.63E-03	*Isaac, pers comm*
Caterpillar	30 x 7 x 7	7.70E-04	*Copplestone, pers comm*
Social Insect - ants	5 x 3 x 3	2.00E-05	
Social Insect - bee	20 x 15 x 10	2.00E-03	
Wood Louse	15 x 6 x 3	1.00E-03	*Copplestone, pers comm*
Earthworm	100 x 5 x 5	3.50E-03	*Copplestone, pers comm*
Herbivorous Mammal (rabbit)	300 x 150 x 100	8.00E-01	*Mammal Society*
Carnivorous Mammal (fox)	670 x 350 x 180	5.50E+00	*Mammal Society*
Small Burrowing Rodent (mouse)	100 x 20 x 20	2.00E-02	*Copplestone, pers comm*
Woodland Bird (Grouse)	350 x 150 x 150	1.50E+00	*Mullarney et al. (1999)*
Bird egg	40 x 25 x 25	1.30E-03	*Copplestone, pers comm*
Reptile (Grass snake)	1200 x 60 x 60	2.26E+00	*University of Exeter (2001)*

Table 6.5 Radionuclides selected for the assessment, for each chosen ecosystem

Freshwater	Estuarine/marine	Terrestrial
^{14}C	^{14}C	^{14}C
^{3}H	^{3}H	^{3}H
^{90}Sr	^{90}Sr	^{90}Sr
^{137}Cs	^{137}Cs	^{137}Cs
$^{239+240}Pu$	$^{239+240}Pu$	$^{239+240}Pu$
^{238}U	^{238}U	^{238}U
^{129}I	^{129}I	^{129}I
^{99}Tc	^{99}Tc	^{35}S
^{210}Po	^{210}Po	^{226}Ra

Table 6.6 Properties of selected radionuclides (modified from FASSET Workshop, 2001)

Nuclide	*Source*	*Waste Disposal e.g. LLW, ILW, HLW?*	*Has natural analogue?*	*Property*	*Decay Mode*	*Half Life*
^{14}C	Artificial/Natural	Yes	Yes	Highly Mobile	â	5370y
^{3}H	Artificial/Natural	No	Yes	Highly Mobile	â	12.26y
^{90}Sr	Artificial	Yes	Yes	Mobile	â	29y
^{137}Cs	Artificial	Yes	Yes	Mobile	â/ã	30.2y
$^{239+240}Pu$	Artificial	Yes	No	Particle reactive	á	24,100y (^{239}Pu) 6537y (^{240}Pu)
^{238}U	Artificial/Natural	Yes	Yes	Mobile	á	4,460,000,000y
^{129}I	Artificial	Yes	Yes	Mobile	â	16,000,000y
^{99}Tc	Artificial	Yes	No	Highly Mobile	â	213,000y
^{210}Po	Natural	No	Yes	Particle reactive	á	138.4d
^{35}S	Artificial	Yes	Yes	Mobile	â	87.5d
^{226}Ra	Natural	No	Yes		á	1,600y

6.3 Impact assessment approach

Figures 1.1 and 6.1 outline the impact assessment approach, with Figure 6.1 identifying information required for the assessment itself. The pathway of exposure and radionuclide source can be assessed through the derivation of concentration factors (Section 6.4), whilst the ecological parameters are used to determine the external exposure.

The approach may be used to make assessments both prospectively and retrospectively although the data to be assembled will be slightly different i.e. based on predicted or measured concentrations in the environment respectively.

For a prospective assessment, concentrations of radionuclides in water, air or soil should be calculated from assumed rates of discharge over an extended period of time.

For a retrospective assessment, measured concentrations of radionuclides in soil, air or water should be used to initiate the assessment

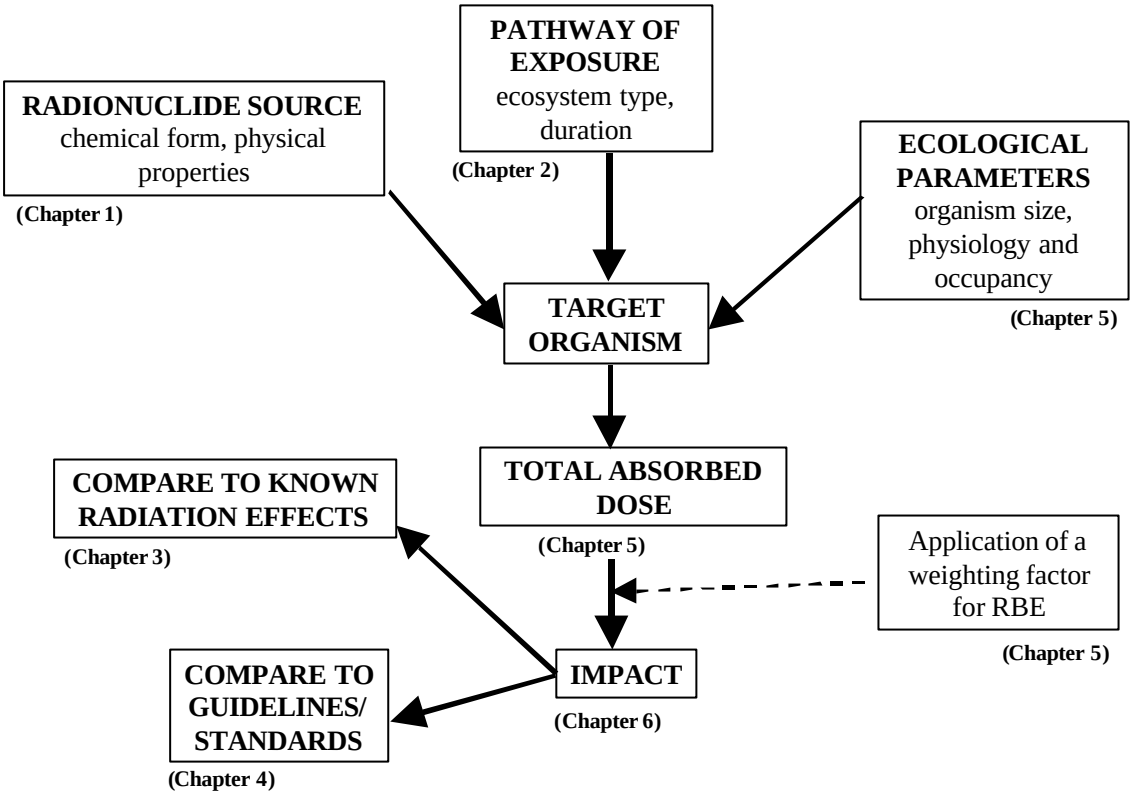

Figure 6.1 Schematic of impact assessment approach

The data required are:

- For the aquatic environment, concentrations of radionuclides in the dissolved phase (filtrate) of the water column (Bq m^{-3}).
- In the terrestrial environment, concentrations of ^{3}H, ^{14}C and ^{35}S in air (Bq m^{-3}).
- For other radionuclides in the terrestrial environment, concentrations of the nuclides in surface soil (Bq kg^{-1} dry weight of soil).

Concentrations should be averaged temporally over a period of at least one-year, and spatially over a scale appropriate to the model being used.

In addition to the calculated concentrations of radionuclides, any existing site-specific data for concentration factors between organisms and water or soil should be assembled. If concentration factors have already been calculated, these may need to be converted into the units required by the spreadsheets, namely:

- For the aquatic environment, Bq kg-1 (fresh weight) of organism per Bq m-3 (dissolved phase) in water.
- For the terrestrial environment, Bq kg-1 (fresh weight) of organism per Bq kg-1 (dry weight) of soil.
- Site-specific data for concentration factors of 3H, 14C or 35S should be in Bq kg-1 (fresh weight) of organism per Bq m^{-3} in air.

Monitoring data for radionuclides in water, soil or air and biota may allow concentration factors to be calculated, bearing in mind that concentrations in biota and soil, air or water should:

- relate to the same location or locations.

- ideally be averaged temporally over a period of at least one year, and spatially over an area appropriate to the assessment being undertaken.

6.4 Derivation of concentration factors (default values)

Concentration factors (CF) relate the concentrations of radionuclides in water, air or soil to the concentrations of radionuclides in biota (and take into account all physiological and physico-chemical properties which may affect radionuclide uptake into biota), allowing the calculation of internal dose. They are intended to provide a 'scoping' order-of-magnitude estimate of doses to biota, and to assist in identifying those radionuclides and/or organisms that are of greatest significance in a particular situation.

Default CF values were derived from the literature for each of the three ecosystems (freshwater, coastal and terrestrial) under study, and are listed in Tables 6.7 to 6.9. The CFs were determined from actual measurements of radionuclide concentrations in different ecosystem compartments which therefore takes into account the different chemical forms of radionuclides and pathways of exposure. However, the values should be used with caution, as information is not available on all possible pathways of exposure.

The default CF values were drawn from an extensive literature search. The principal databases searched were the ISI database and the IAEA's INIS database. Where the literature indicated a range of applicable values, the mean was taken. Values were converted, where necessary, to the following units:

- **Aquatic ecosystems:**
  - Sediment: Bq kg^{-1} (dry weight) of sediment per Bq m^{-3} (dissolved phase) in water;
  - Organisms: Bq kg^{-1} (fresh weight) of organism per Bq m^{-3} (dissolved phase) in water;
- **Terrestrial ecosystem:**

- ^{3}H, ^{14}C and ^{35}S:
  - Soil: Bq kg^{-1} (fresh weight) of soil per Bq m^{-3} in air;
  - Organisms: Bq kg^{-1} (fresh weight) of organism per Bq m^{-3} in air.

- Other radionuclides:
  - Organisms: Bq kg^{-1} (fresh weight) of organism per Bq kg^{-1} (dry weight) of soil.

- **Considerations for organic tritium**

The default CFs provided for tritium in all three ecosystems assume that tritium is present in inorganic form, primarily incorporated as tritiated water (HTO). Furthermore, a specific activity approach was adopted to determine the CF for tritium (see Section 6.4.1) and consequently was referenced as a ratio to the water content of the organism relative to air. In this way, tritium (as tritiated water) within an organism is determined, essentially, by its concentration in water in the environment and the proportion of water in the tissues of the organism, i.e. CFs of around 0.001 m^3 kg^{-1} relative to water. However, if tritium is incorporated into organic compounds, much higher CFs can be observed; for example CFs of around 3 m^3 kg^{-1} are reported in a range of different organisms in the Severn estuary near a radiopharmaceutical facility discharge (FSA, 2000). **Therefore, the form of the tritium needs to be considered during the interpretation, and may influence the output from the assessment.**

6.4.1 Availability of concentration factor (CF) data

CFs are not always cited in the literature, even when concentrations in the relevant environmental media have been measured. It has been possible to derive CFs where relevant parameters are given, or by direct communication with the authors or research groups. Data were, as far as possible, drawn from UK environments. If this was not feasible, data were drawn from areas with broadly comparable

environmental conditions. The literature used to derive the CFs has been reviewed in Chapter 2 and is quoted in the References list at the end of this report.

The literature on natural series radionuclides is much more limited than that for anthropogenic radionuclides. The majority of the information was drawn from the management of uranium mill tailings, natural background radiation in uranium rich regions and laboratory studies. The range of species covered was also limited.

The source of the CF, and the associated potential for variability, should be considered in the interpretation of the results.

- **Freshwater ecosystem**

Data on the freshwater ecosystem are sparse, so it was necessary to consider other geographical regions to seek relevant data. Much work in recent years has concentrated on the potential impacts and accumulation of radiocaesium post-Chernobyl, and radiostrontium following the accident at Mayak in the late 1950s. Consequently, there is a substantial literature on these two radionuclides and their impacts.

Freshwater CFs (Table 6.7) were drawn principally from IAEA (1994) and RWMC (1994). Wherever possible, IAEA (1994) CFs were used, but these were limited in number and related specifically to sediments (not water as required here) and edible portions of fish (not whole body burdens).

Data were extrapolated from organisms of similar dimensions and characteristics to the reference organisms. For example, ^{210}Po and ^{238}U values for benthic molluscs and crustaceans could not be identified from the literature, so a value for a generic invertebrate of similar size was substituted as required.

CFs for zooplankton and small benthic crustaceans were drawn from a study of a lake in Sweden impacted by Chernobyl fallout. It is recognised that this is not ideal, as environmental conditions are likely to be different from those in the UK. CF data for natural series radionuclides were also drawn from studies of catchments around uranium mines and mills in the Canadian literature.

- **Estuarine/marine ecosystem**

The most complete CF dataset was obtained for the estuarine/marine ecosystem (Table 6.8). This reflects the historic focus, as most nuclear installations in the UK and elsewhere discharge to the estuarine/marine environment. Most available CFs are for food species or known bio-accumulators of radionuclides, reflecting potential pathways to man.

Most CF values were recommended by Woodhead (2001) or IAEA (1985). For three radionuclides, alternatives were used: ^{99}Tc (in macrophytes, benthic molluscs, small benthic crustaceans, large benthic crustaceans, benthic fish, pelagic fish); ^{137}Cs (in zooplankton, benthic molluscs, small benthic crustaceans, benthic and pelagic fish, seal and whale); and $^{239+240}$Pu (in phytoplankton, zooplankton, macrophyte, benthic mollusc, small and large benthic crustaceans).

Variation in the CFs from the Woodhead (2001) and IAEA (1985) were made for specific reasons. For example, Steele (1990) reported CFs for species of pelagic and benthic fish common in UK waters (cod, *Gadhus morhua*, haddock, *Merlanogrammus aelefinus*, whiting, *Merlangius merlangius* and plaice, *Pleuronectes platessa*), whereas the Woodhead (2001) and IAEA (1985) values were for a generic fish. Lobsters also show a high concentration factor for ^{99}Tc (Brown *et al.*, 1999), and so this was adopted to ensure that doses are not under-estimated. The generic concentration factors for ^{99}Tc crustaceans given by IAEA (1985) do not reflect this effect.

Data also had to be sought where species CFs were not covered by the Woodhead (2001) or IAEA (1985), e.g. ^{137}Cs in seals and whales. There are few data on marine mammals. CFs have been reported for ^{137}Cs in harbour seals, *Phoca vitulina*, grey seals, *Halicheorus grypus*, and harbour porpoise, *Phocoena phocoena* in UK waters (Berrow *et al.*, 1998; Watson *et al.*, 1999).

No reliable CF data could be found for seabirds and waders.

Table 6.7 Default concentration factors, freshwater ecosystem

Nuclide	Kd	benthic bacteria	phyto-plankton	Zoo-plankton	macro-phyte	benthic mollusc	small benthic crustacea	large benthic crustacea	benthic fish	pelagic fish	amphibian	small aquatic mammal	duck
^{3}H	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03
^{14}C	2.00E+00	2.00E+00			4.55E+00	7.30E+00	7.30E+00	7.28E+00	4.60E+00	4.60E+00			
^{90}Sr	1.00E+00	1.00E+00			1.20E+00	2.52E-01	2.67E-01	2.67E-01	4.27E-02	4.27E-02			
^{99}Tc	5.00E-03	5.00E-03	9.00E-03		1.70E+00	2.40E-02	1.25E-02	1.25E-02	4.51E-02	4.51E-02			
^{129}I	1.00E-02	1.00E-02			4.00E-01	1.70E-01	1.72E-01	1.72E-01	4.00E-02	4.00E-02			
^{137}Cs	1.00E+00	1.00E+00		1.90E+01	2.33E+00	5.80E-01	5.23E+00	6.33E-01	1.09E+01	1.09E+01			
^{210}Po	2.70E+00	2.70E+00			1.40E+00	1.02E+02	1.02E+02	1.02E+02	5.00E-02	5.00E-02			
^{238}U	5.00E-02	5.00E-02			6.50E+00	1.80E-01	1.80E-01	1.80E-01	1.00E-02	1.00E-02			
^{239}Pu	1.00E+02	1.00E+02	3.32E+00	3.32E+00	1.84E+00	8.17E-01	1.37E-01	1.37E-01	6.93E-02	6.93E-02		2.26E-01	2.00E-03

Table 6.8 Default concentration factors, coastal-marine ecosystem

Nuclide	Kd	benthic bacteria	phyto-plankton	zoo-plankton	macro-phyte	benthic mollusc	small benthic crustacea	large benthic crustacea	benthic fish	pelagic fish	waterbird	seal	whale
^{3}H	1.00E-03		1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03	1.00E-03			
^{14}C	2.00E+00		9.00E+00	2.00E+01	1.00E+01	2.00E+01	2.00E+01	2.00E+01	2.00E+01	2.00E+01			
^{90}Sr	1.00E+00		3.00E-03	1.00E-03	5.00E-03	1.00E-02	1.00E-03	2.00E-03	2.00E-03	2.00E-03			
^{99}Tc	1.00E-01		5.00E-03	1.00E-01	1.40E+02	8.31E-01	2.43E-01	8.00E+00	2.72E-02	2.72E-02			
^{129}I	2.00E-02		1.00E+00	3.00E+00	1.00E+00	1.00E-02	1.00E-02	1.00E-02	1.00E-02	1.00E-02			
^{137}Cs	3.00E+00		2.00E-01	2.20E-02	5.00E-02	2.13E-02	1.00E+01	3.00E-02	8.98E-02	8.98E-02		4.88E-01	1.88E-01
^{210}Po	2.00E+02		3.00E+01	3.00E+01	1.00E+00	1.00E+01	5.00E+01	5.00E+01	2.00E+00	2.00E+00			
^{238}U	1.00E+00		2.00E-02	5.00E-03	1.00E-01	3.00E-02	1.00E-02	1.00E-02	1.00E-03	1.00E-03			
^{239}Pu	1.00E+02		1.60E+02	8.00E-01	2.52E+00	2.43E+00	3.00E+01	2.25E-01	4.00E-02	4.00E-02			

Table 6.9 Default concentration factors, terrestrial ecosystem

	Concentration factors, organism : air or organism : soil[7]								
Nuclide	**soil**	**bacteria[8]**	**Lichen**	**tree**	**shrub**	**herb**	**seed**	**fungi**	**caterpillar**
^{3}H	5.36E+01	5.36E+01	1.61E+02	1.07E+02	1.52E+02	1.18E+02	8.93E+00	1.60E+02	1.52E+02
^{14}C	1.88E+03	1.88E+03	3.75E+01	1.25E+03	4.22E+02	5.63E+02	4.75E+03	3.75E+02	4.22E+02
^{35}S	5.00E+01	5.00E+01	1.50E+02	1.50E+02	1.50E+02	1.50E+02	5.00E+01	5.00E+01	5.00E+01
^{90}Sr	1.00E+01	1.00E+01		1.04E+00	1.70E-02			4.76E-03	
^{129}I	1.00E+01	1.00E+01							
^{137}Cs	1.00E+01	1.00E+01	7.73E-01	4.00E-02	1.56E-01	1.43E-01		1.13E+00	
^{226}Ra	1.00E+01	1.00E+01	1.00E-01	1.10E-01	2.20E-01	1.93E-01			
^{238}U	1.00E+01	1.00E+01		1.40E-01		7.90E-01			
^{239}Pu	1.00E+01	1.00E+01	6.60E-01	3.70E-01		4.70E-02			

Nuclide	**ant**	**bee**	**woodlouse**	**earthworm**	**herbivore mammal**	**carnivore mammal**	**rodent**	**bird**	**bird egg**	**reptile**
^{3}H	1.61E+02	1.52E+02	1.43E+02	1.54E+02	1.34E+02	1.38E+02	1.38E+02	1.34E+02	1.52E+02	1.34E+02
^{14}C	2.81E+02	4.22E+02	5.63E+02	3.50E+02	7.50E+02	6.90E+02	6.90E+02	7.03E+02	2.81E+02	7.03E+02
^{35}S	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01	5.00E+01
^{90}Sr							5.00E+00			
^{129}I										
^{137}Cs	1.37E-02	1.60E-03	3.60E-02	1.30E-02	2.16E+00	9.03E+00	1.30E-02	1.60E+00		
^{226}Ra					1.06E+00		2.32E-02	6.00E-02		
^{238}U					4.00E-03		2.00E-03			
^{239}Pu	1.37E-02	1.90E-03	4.50E-02	2.60E-02	1.00E-04		5.00E-04			

[7] Concentration factors are Bq kg^{-1} (fresh weight) per Bq m^{-3} in air for ^{3}H, ^{14}C, and ^{35}S; Bq kg^{-1} (fresh weight) per Bq kg^{-1} (dry weight) in soil for other nuclides.
[8] Bacteria are assumed to have the same composition as soil (dimensions of bacteria are such that internal dose is unimportant).

- **Terrestrial ecosystem (coastal grassland)**

The CFs for the grassland ecosystem were drawn from a wider variety of sources than the aquatic systems and are given in Table 6.9. CFs for ^{137}Cs have been mainly derived studies in the literature which describe the impact of Chernobyl fallout. It was also necessary to collate data for relevant species from different land use types, e.g. forest (e.g. Barci-Funel *et al.*, 1995; Guilitte *et al.*, 1994) or areas contaminated with uranium tailings (e.g. Cloutier *et al.*, 1985; Clulow *et al.*, 1992; Mirka *et al.*, 1996). As with the freshwater ecosystem, data on naturally occurring radionuclides have been obtained from Canadian studies.

Occasionally, a different species had to be substituted due to a lack of CF data on the chosen reference organism. For example, to obtain a CF for $^{239+240}$Pu, it was necessary to substitute the 'reference herbivorous mammal - rabbit' with a field vole sampled from a saltmarsh (Copplestone, 1996). The difference in size and mass , and hence the metabolism, of the two animals may affect the CF, and should be considered when interpreting results.

A specific activity approach was needed to calculate CF values for ^{3}H, ^{14}C and ^{35}S:

- ^{3}H: it was assumed that the nuclide was present as tritiated water and that the CF could be approximated as the ratio of the water content of the organism to that of air. A water content of air of 0.0056 kg m^{3}, corresponding to a relative humidity of 50% at 15º C, was taken as the reference. Typical dry:wet weight ratios of materials and organisms were used to estimate the water content and hence determine the CF relative to air (Copplestone, pers. comm.). ^{3}H was estimated relative to its concentrations in air.
- ^{14}C: carbon content (kg kg^{-1} dry weight) of organisms and soil (Bowen, 1966) were combined with the above dry:wet weight ratios. A reference value for carbon (as CO_2) in air of 0.00016 kg m^{-3} was used to estimate the CF values. ^{14}C was also estimated relative to its concentrations in air.
- ^{35}S: it has a short radioactive half-life (87.4 days), so radioactive decay will alter the relationship between ^{35}S and the stable element in the various ecosystem 'compartments'. Kluczewski *et al.* (1987) have studied the uptake of ^{35}S from air to a variety of plant crops; CF for leafy crops have been taken as a conservative value for all plants, whereas the lower CF for root crops have been taken as an indicative value for other biota.

The seasonality of particular species is also a factor in the interpretation of doses. This is particularly the case with organisms such as fungi where the fruiting bodies are only be present for part of the year.

6.4.2 Predicted radionuclide concentrations in biota using default CFs

A limited (i.e. not extensive) validation was provided for the default CFs in the coastal environment (Tables 6.8). It considered the Cumbria area, which is intensively monitored both by the UK regulatory agencies (Environment Agency, 2001a; Food Standards Agency, 2000) and the operators of the Sellafield nuclear site (BNFL, 2000). Table 6.10 compares the calculated biota levels, using the reference CFs, with the typical measured values.

The majority of the calculated and measured values were not significantly different from each other, i.e. within an order of magnitude providing confidence that the default CF values can provide a reasonable estimate of likely exposure. The actual measured CF will vary with:

- species;
- local environmental conditions (e.g. salinity, suspended sediment load);
- concentration gradients of radionuclides with distance from the point source of discharge; and
- home range or migratory habits of organisms.

This probably explains the difference between calculated and observed values for ^{99}Tc and ^{137}Cs fish in Table 6.10.

Table 6.10 Comparison between calculated and measured biota concentrations in a coastal environment

		Biota concentration (Bq kg^{-1})	
Radionuclide	**Biota**	**Measured**	**Predicted (from Table 6.8)**
^{3}**H**	Benthic molluscs	7.00E+01	2.50E+02
	Benthic fish	2.20E+02	2.50E+02
^{14}**C**	Pelagic fish (cod)	7.00E+01	-
^{90}**Sr**	Molluscs (*M. edulis*)	5.00E+00	-
^{99}**Tc**	Macrophyte (*F. Vesiculosus*)	2.00E+04	7.00E+04
	Molluscs (*M. edulis*)	1.30E+03	4.20E+02
	Large crustacean (lobster)	4.70E+03	4.00E+03
	Pelagic fish (cod)	2.00E+00	1.40E+02
	Benthic fish (plaice)	6.00E+00	1.40E+02
	Seabird (teal)	3.00E+00	-
^{137}**Cs**	Sediment (max)	6.00E+02	9.00E+02
	Macrophyte (*F. vesiculosus*)	6.00E+00	1.50E+02
	Molluscs (*M. edulis*)	7.00E+00	6.40E+00
	Large crustacean (lobster)	7.00E+00	9.00E+00
	Pelagic fish (cod)	7.00E+00	2.70E+02
	Benthic fish (plaice)	5.00E+00	2.70E+02
	Seabird (teal)	1.00E+01	-
^{210}**Po**	Molluscs (*M. edulis*)	3.00E+01	-
^{238}**U**	Molluscs (*m. edulis*)	2.00E+00	-
^{239}**Pu**	Sediment (max)	6.00E+02	5.00E+02
	Macrophyte (*F. vesiculosus*)	2.00E+01	1.30E+02
	Molluscs (*M. edulis*)	1.20E+01	1.20E+02
	Large crustacean (lobster)	5.00E-01	1.10E+00
	Pelagic fish (cod)	2.00E-02	2.00E-01
	Benthic fish (plaice)	5.00E-02	2.00E-01
	Seabird (teal)	1.20E-02	-

6.5 Calculation of doses - methodology

With the help of the devised spreadsheets, it is possible to undertake an impact assessment of ionising radiation on wildlife. This generic impact assessment can provide an indication of the likely scale of risk to wildlife in England and Wales. If site-specific information is available, a more accurate impact assessment can be undertaken.

6.5.1 Steps to undertake the dose calculations

The following dose assessment approach is based on a number of assumptions described in Section 5.7.1 and uses the algorythms described in Chapter 5 to estimate absorbed dose to specified biota (Section 6.2.2.) for given radionuclides (Section 6.2.3). The assessment approach also adopts values for radiation-weighting factors (Section 6.5) in order to account for the effects of different radiation types (e.g. α and γ). The user in the assessment spreadsheets can adjust these values.

The most important assumption is that concentrations in biota are in equilibrium with concentrations in the surrounding environmental media. The method can not be used to assess doses to biota in situations where the concentrations of radionuclides in the surrounding environmental media are changing rapidly. Further information about how this is affected in terrestrial and aquatic ecosystems is given in Section 5.7.1.

Section 6.3 details the data required to run an assessment and its format.

This proposed approach should be applied in conjunction with the operating guide for the spreadsheet application provided as Appendix 3. The steps in calculating doses are as follow:

a) Obtain relevant site-specific information (e.g. radionuclide concentrations in biota, CF values).

For prospective assessment:

b) Run predictive models (not part of this report) to determine concentrations of radionuclides in soil, water and air after a period of at least 50 years.

c) Set all soil, air or water concentrations in the spreadsheet to zero, and restore the default values for concentration factors and radiation weighting factors. This ensures that any alterations made to the spreadsheet input by previous users are cancelled.

d) Enter any site specific concentration factors that have been assembled.

e) Enter the water, air or soil concentrations that have been assembled.

f) Initiate the calculation of concentrations and doses.

For retrospective assessment:

g) Compare the calculated environmental concentrations (from step f above) with the observed values (collected in step e), that have been assembled. If marked differences are found, adjust those concentrations in the spreadsheets.

h) If you do not have measured concentrations of one or more radionuclides in the soil, air or water medium but have measured values of those radionuclides in sediment or biota, the spreadsheet allows you to enter these measured concentrations to estimate concentrations in water, soil or air.

i) Having made any such adjustments (from steps g or h), re-initiate the calculation of concentrations and doses.

For all assessments:

j) Check carefully that all input data are correct, then save the calculation results.

k) Compare the results with the guideline values tabulated in Table 3.1.

l) Compare the doses calculated with the effects observed on wildlife in Tables 3.6-3.19.

Based on the expert opinion of the authors:

if doses are calculated with the generic default concentration factors are in excess of 5% of the IAEA 'benchmark' values (Table 3.1),

or

if doses are calculated with site specific values and exceed 30% of the IAEA 'benchmark' values (Table 3.1), then consideration should be given to further action, e.g:

It is recommended that a more detailed assessment is undertaken, requiring additional measurements of concentrations of key radionuclides in biota and the environment to provide more site-specific concentration factors if either of the two statements above are true.

The calculations will produce estimates of the doses to biota, but a number of points must be borne in mind when interpreting the results, as described below. These are in addition to considerations on concentration factors identified in Section 6.4.

6.6 Sensitivity analyses, and gaps in the data

It is possible to calculate doses to biota using several variants of the input parameters. For example, it is possible to check the sensitivity of the calculated doses to the value of the radiation weighting factors by running the calculations with different w_r values.

For some organisms and some nuclides, it has not been possible to provide default CF values. The internal dose resulting from such organism-nuclide combinations will not be calculated, although the external dose will be calculated. Entering nominal CF values of similar magnitude to other CF values for the same nuclide can assess the possible importance of the 'missing' CF data. In some cases, (e.g. internal contamination by â and ã emitters in small organisms) external radiation dose is likely to be dominant and the 'missing' CF value may have little effect on the dose calculation. In other cases (e.g. internal contamination by á emitters) the total dose will depend directly on the 'missing' CF value. If the nominal CF values entered produce significant doses, this may indicate a need for measurements to be made in biota (for a retrospective assessment), or attempt to determine a concentration factor from field or laboratory studies (for a prospective assessment).

6.7 Interpreting results and taking account of uncertainty

Calculation results can be checked against the IAEA dose rates of 40 ì Gy ħ^{1} for terrestrial biota, and 400 ì Gy ħ^{1} for aquatic biota, where harm to populations and ecosystems is considered unlikely (Table 3.1). In comparing results with these 'benchmarks', uncertainties in the calculation must be considered.

At present, any calculated radiation dose to biota must be regarded as an estimate rather than an accurate value. The main sources of uncertainty may be summarised as follow:

- The weighting factors of 3 for low energy â radiation and 20 for á radiation are likely to be cautious, especially if it proves that non-stochastic effects are most important in determining harm to ecosystems. The 'true' values for these weighting factors may be a factor of 3 to 4 lower, and are most unlikely to be a factor of two higher.
- The calculation of external doses from concentrations of radionuclides in soils or sediments is cautious, mainly because they assume soil or sediment is uniformly contaminated to an infinite depth. External doses may therefore be over-estimates, but should not exceed a factor of two in most circumstances.
- The greatest uncertainty lies in the values of concentration factor used to calculate internal contamination by radionuclides, and hence internal doses. Concentration factors vary considerably between species and also with environmental conditions, such as water chemistry and soil type. The true values for concentration factor could easily differ from the recommended defaults by an order of magnitude or more in either direction.
- For a given level of internal contamination, calculated internal dose are quite accurate, and should produce results for the average dose within the organism within 10% of the true value. Doses to different organs may of course differ from this average value if radionuclides are not distributed uniformly within the organism.

In recognition of these uncertainties, it is recommended that consideration be given to the following points when interpreting results:

- If only default concentration factors are being used and the calculated doses are in excess of 5% of IAEA guideline values, the uncertainties within the result should be considered. For example, are the concentration factors used appropriate to the assessment, e.g., is the tritium present as tritiated water or organically bound? Efforts should be made to acquire site specific concentration factors or direct measurements of radionuclide concentrations in important organisms.
- If site specific concentration factors, or actual environmental measurements, are being used and calculated doses exceed 30% of the IAEA guidelines values then consideration should be given as to what further investigation might be appropriate. This might involve consideration of the radiosensitivity of the organisms receiving the highest calculated doses, e.g., are they amongst the most or least radiosensitive organisms? For a retrospective assessment of an existing contaminated ecosystem, selected biomarker studies and/or ecological investigations may be appropriate.
- If the calculated doses are several orders of magnitude lower than the guideline values provided in Table 3.1 then, subject to verification and assessment of the uncertainties, it should be possible to conclude that the impact on wildlife is likely to be small.

It should also be clear that this methodology only takes into account nine radionuclides (Table 6.5). Any assessment must also consider the dose contribution from other radionuclides that could be discharged from the site under investigation.

Reasoned judgement using the information in this report should allow sensible conclusions to be drawn in most cases.

6.8 Assessment of risk to wildlife in England and Wales

To support the proposed assessment approach, this Section provides a series of realistic (in terms of levels of radionuclide releases and contamination) but hypothetical 'worst case' scenarios. The scenarios have been developed by considering radionuclide measurements in the environment from around a number of nuclear installations within England and Wales, as if the releases all occurred simultaneously in the same location. These scenarios will also demonstrate the likely risks to wildlife in England and Wales.

Real data were compiled from a variety of sources to represent the worst case in terms of anthropogenic radioactivity in each of the three ecosystems (Section 6.2). Doses to organisms in each of these 'worst case' scenarios were then assessed using the assessment spreadsheets. The description that follows therefore provides an indication of the level of risk posed to UK wildlife by anthropogenic radioactivity, as well as a 'worked example' for use of these assessment spreadsheets

Scenario 1: Freshwater ecosystem

In this scenario it was assumed that a nuclear power station, a uranium enrichment facility, and a radio-pharmaceutical plant discharge into the same freshwater body.

Measured water concentrations (Table 6.11) were used with the default[9] CF values to generate biota concentrations (Table 6.12). These were then compared against measured concentrations (Table 6.12). Measured concentrations of biota were also used to validate/adjust the concentration factors and to generate water concentrations where none were available.

It was not possible to carry out the assessment for some of the reference organisms. This was due to the lack of published CFs, i.e.: for:

- phytoplankton (^{14}C, ^{90}Sr, ^{129}I, ^{137}Cs, ^{210}Po, ^{238}U),

[9] A non-default CF value of 3 for tritium in all fauna was used, to reflect the possible presence of organically bound tritium. No measurements of ^{3}H in biota in the freshwater environment were available; however, high CF values were observed in the estuarine environment affected by discharges from the same facility, due to the presence of ^{3}H in organic form (FSA, 2000).

- zooplankton (^{14}C, ^{90}Sr, ^{99}Tc, ^{129}I, ^{137}Cs, ^{210}Po),
- amphibians (all radionuclides, except ^{3}H),
- aquatic mammals (all radionuclides except ^{3}H, ^{239}Pu), and
- duck (all radionuclides, except ^{3}H and ^{239}Pu).

As a result, internal dose to reference organisms could not be assessed for those organisms. In most cases an external dose could be estimated.

The dose rates to biota in the freshwater ecosystem were assessed using the spreadsheets. Due to the lack of information, these doses do not include contributions from ^{129}I and ^{210}Po. Weighted dose rates (Figure 6.2) showed that macrophytes and bacteria would receive the maximum dose rates (~23 $\mu Gy\ h^{-1}$ and ~22 $\mu Gy\ h^{-1}$, respectively), with the main contribution from ^{238}U. The remaining reference organisms would all receive a dose of <6 $\mu Gy\ h^{-1}$. Unweighted dose rates (Figure 6.3) gave a maximum of ~1.4 $\mu Gy\ h^{-1}$ to macrophytes and ~1.3 $\mu Gy\ h^{-1}$ to bacteria (both with a main contribution from ^{238}U). The rest of the reference organisms would receive doses of ~0.7 $\mu Gy\ h^{-1}$ or less.

All organisms showed doses substantially lower than the IAEA recommended maximum dose rate to biota in freshwater ecosystems is 400 $\mu Gy\ h^{-1}$ and below the dose rates where effects were observed in Chapter 3. The impact on biota in the freshwater ecosystem, based on this 'worst case' scenario, would therefore appear to be low, especially as the organisms receiving the highest doses are amongst the least radio-sensitive.

Table 6.11 Concentrations of radionuclides in freshwater

Radionuclide	Concentration (dissolved phase) Bq m^{-3}	
	Measured	**Calculated[d]**
^{3}H [a]	50, 000	
^{14}C	-	6.52
^{90}Sr [b]	-	234
^{99}Tc [c]	1, 000	
^{129}I	-	-
^{137}Cs [b]	40	
^{210}Po	-	-
^{238}U [c]	180	
$^{239/240}Pu$ [b]	-	0.0115

[a] data from Nycomed Amersham, Glamorganshire Canal

[b] data from Llyn Trawsfynyydd, Wales

[c] data from Capenhurst, Riveracre Brook.

[d] calculated from selected measurements in biota and default CF values

Table 6.12 Freshwater ecosystem scenario - biota concentrations

		Biota concentration (Bq kg^{-1})	
Radionuclide	**Biota**	**Measured**	**Predicted**
^{14}C [a]	Pelagic fish	3.00E+01	-
^{90}Sr [a]	Pelagic fish	1.00E+01	-
^{99}Tc [c]	Silt	5.00E+02	5.00E+00
	Macrophyte	1.10E+02	1.70E+03
^{137}Cs [b]	Pelagic fish	1.00E+02	4.36E+02
^{238}U [c]	Silt	5.40E+02	9.00E+00
	Macrophyte	2.20E+02	1.17E+03
^{239}Pu [a]	Pelagic fish	8.00E-04	-

[a] data from Nycomed Amersham, Glamorganshire Canal

[b] data from Llyn Trawsfynydd, Wales

[c] data from Capenhurst, Riveracre Brook.

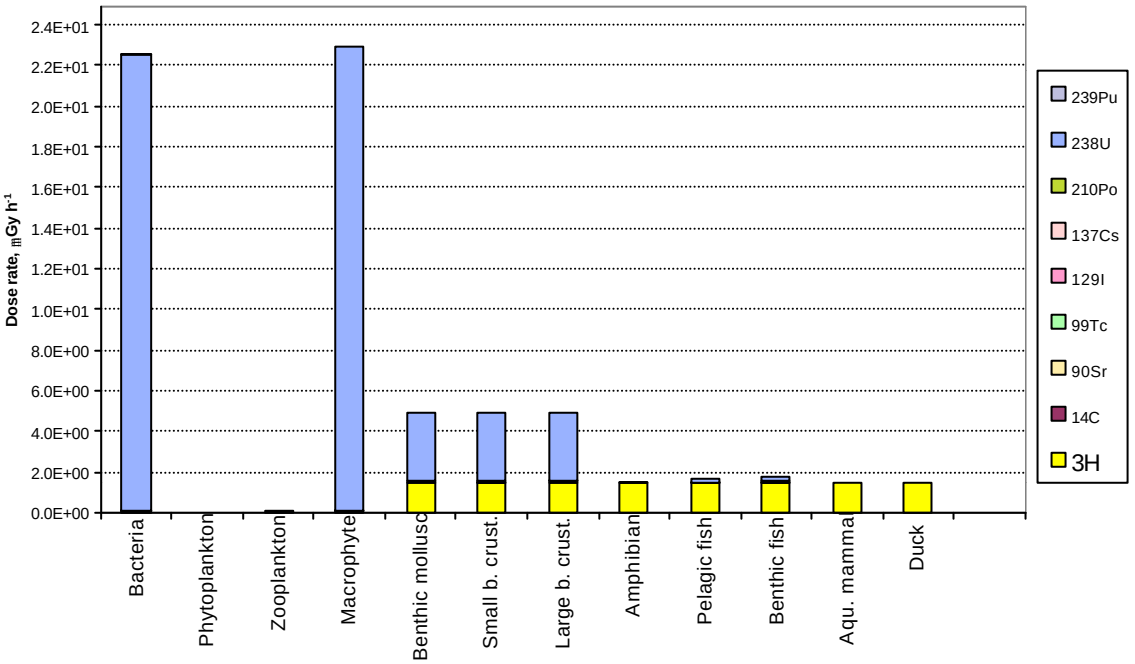

Figure 6.2 Weighted dose rates, freshwater ecosystems

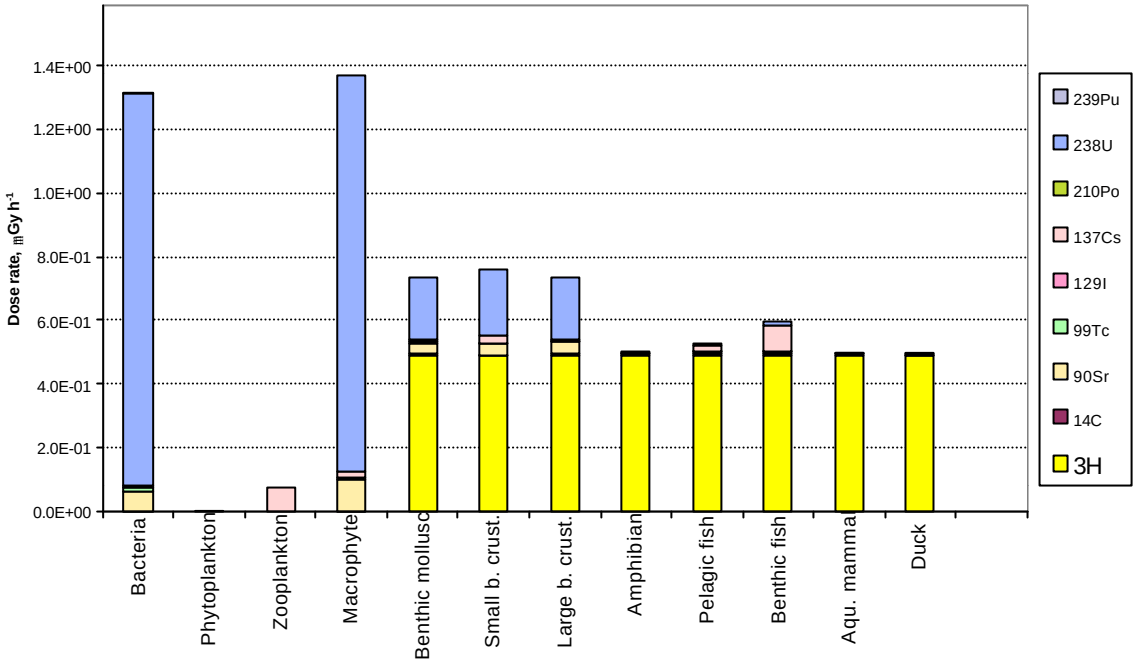

Figure 6.3 Unweighted dose rates, freshwater ecosystems

Scenario 2: Estuarine/marine ecosystem

The scenario was created using data from the Cumbrian coastal environment. Extensive data were obtained from the Environment Agency, FSA-RIFE and BNFL annual surveillance reports (EA, 2001a; FSA, 2000; BNFL, 2000).

Water concentrations (Table 6.13) and default concentration factor values were used to generate initial biota concentrations in the reference organisms. These calculated results were then compared with typical measured biota concentrations taken from the north-west Cumbrian coast (Table 6.14).

Table 6.13 Measured concentrations of radionuclides in seawater

	Concentration (dissolved phase) Bq m^{-3}	
Radionuclide	**Measured**	**calculated[a]**
^{3}H	25, 000	
^{14}C	-	3.5
^{90}Sr	-	500
^{99}Tc	500	
^{129}I	-	-
^{137}Cs	300	
^{210}Po	-	3
^{238}U	-	66.7
$^{239/240}Pu$	5	

[a] calculated from selected measurements in biota and default CF values

Table 6.14 Estuarine/marine ecosystem scenario - biota concentrations

		Biota concentration (Bq kg^{-1})	
Radionuclide	**Biota**	**Measured**	**Predicted**
^{3}H	Benthic molluscs	7.00E+01	2.50E+01
	Benthic fish	2.20E+02	2.50E+01
^{14}C	Pelagic fish (cod)	7.00E+01	-
^{90}Sr	Molluscs (*M. Edulis*)	5.00E+00	-
^{99}Tc	Macrophyte (*F. Vesiculosus*)	2.00E+04	7.00E+04
	Molluscs (*M. Edulis*)	1.30E+03	4.15E+02
	Large crustacean (lobster)	4.70E+03	4.00E+03
	Pelagic fish (cod)	2.00E+00	1.36E+01
	Benthic fish (plaice)	6.00E+00	1.36E+01
	Seabird (teal)	3.00E+00	-
^{137}Cs	Sediment (max)	6.00E+02	9.00E+02
	Macrophyte (*F. Vesiculosus*)	6.00E+00	1.50E+02
	Molluscs (*M. Edulis*)	7.00E+00	6.39E+00
	Large crustacean (lobster)	7.00E+00	9.00E+00
	Pelagic fish (cod)	7.00E+00	2.69E+02
	Benthic fish (plaice)	5.00E+00	2.69E+02
	Seabird (teal)	1.00E+01	-
^{210}Po	Molluscs (*M. Edulis*)	3.00E+01	-
^{238}U	Molluscs (*M. Edulis*)	2.00E+00	-
^{239}Pu	Sediment (max)	6.00E+02	5.00E+02
	Macrophyte (*F. Vesiculosus*)	2.00E+01	1.26E+02
	Molluscs (*M. Edulis*)	1.20E+01	1.21E+01
	Large crustacean (lobster)	5.00E-01	1.13E+00
	Pelagic fish (cod)	2.00E-02	2.00E-01
	Benthic fish (plaice)	5.00E-02	2.00E-01
	Seabird (teal)	1.20E-02	-

Where significant differences between measured and calculated concentrations were observed, measured concentrations were used to adjust the corresponding CF values. On this basis, alterations were made to the concentration factors of:

- ^{3}H (benthic fish)
- ^{99}Tc (pelagic fish, benthic fish, seabirds)
- ^{137}Cs (pelagic fish, benthic fish, seabirds)
- ^{239}Pu (pelagic fish, benthic fish, seabirds)

In all cases, except ^{3}H, the initial calculated concentrations were higher than the measured concentrations, giving confidence that the default CFs would not under-estimate doses. In the case of

seabirds no default CF is available; however, the monitoring programme measurements allow values to be estimated.

Concentrations of ^{14}C, ^{90}Sr and ^{210}Po in the water column were estimated from the measured concentrations in pelagic fish and molluscs respectively. Re-running the calculations then generated concentrations of these nuclides, and corresponding internal doses. ^{210}Po of natural origin may have been somewhat enhanced by anthropogenic inputs from a phosphate plant in the region (FSA, 2000).

No ^{129}I concentration data are available in the water column and in the biota. ^{129}I is a conservative radionuclide i.e. it does not bind in the sediments and remains in the water column. For the purposes of this impact assessment, its concentration in water was estimated using tritium as a tracer for the discharges from the Sellafield reprocessing plant:

^{129}I in water = ^{3}H in water x (^{129}I discharge / ^{3}H discharge) = 5 Bq m^{-3}.

The assessment spreadsheets were used to calculate both weighted (Figure 6.4) and unweighted (Figure 6.5) dose rates to biota. Weighted doses showed that phytoplankton received the maximum dose (~53 μGy h^{-1}) with the main contribution from ^{239}Pu. Bacteria in sediments would receive a dose of ~30 μGy h^{-1}, mainly from ^{239}Pu and ^{210}Po. For the remaining reference organisms, dose rates would be <10 μGy h^{-1}. The maximum unweighted dose was received by macrophytes (~4 μGy h^{-1}), with ^{99}Tc as the main contributor. Phytoplankton (~2.5 μGy h^{-1}) and bacteria (~1.7 μGy h^{-1}) followed, with main contributions from ^{239}Pu and ^{239}Pu plus ^{210}Po, respectively. For the remainder of the reference organisms, dose rates were <10 μGy h^{-1}.

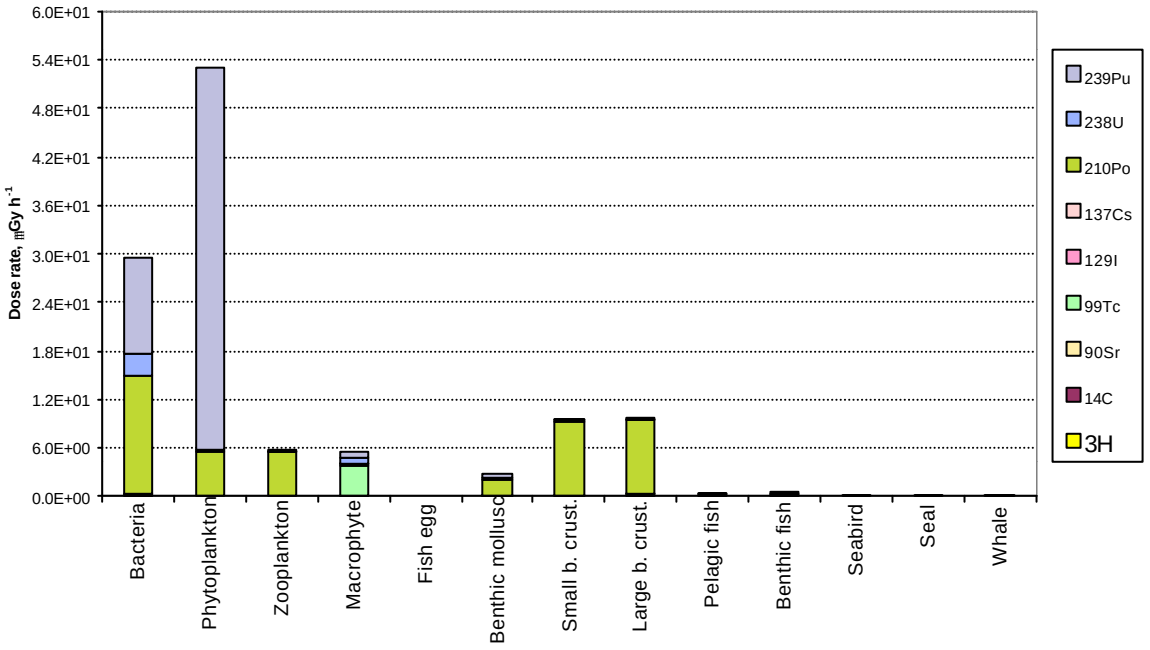

Figure 6.4 Weighted dose rates, estuarine/marine ecosystem

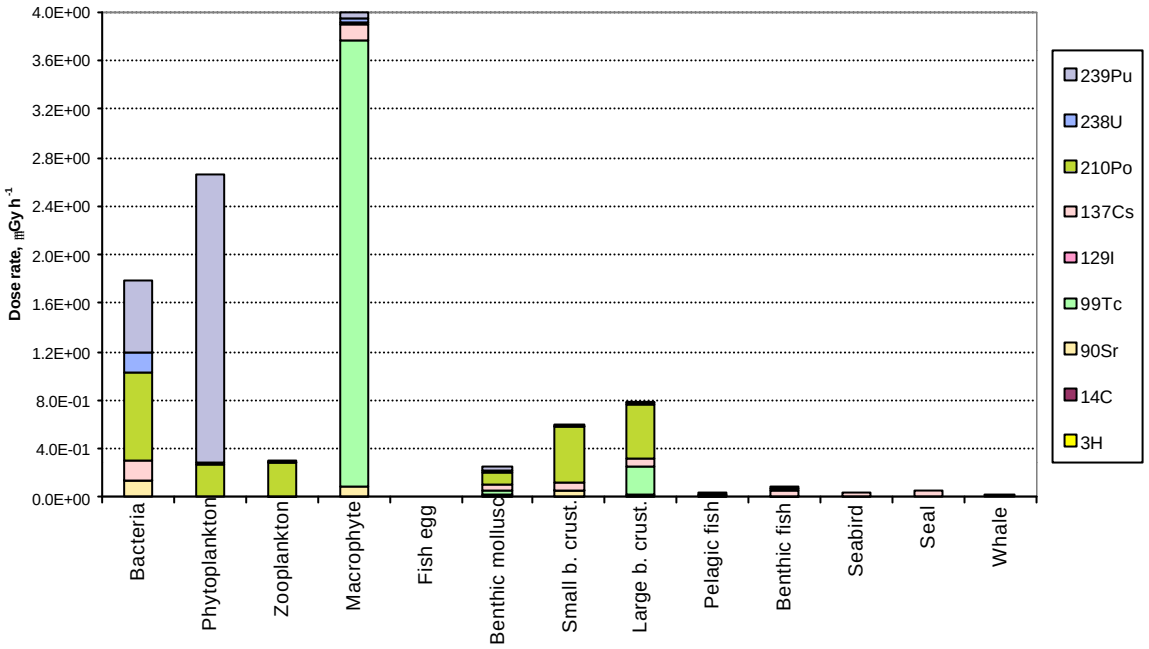

Figure 6.5 Unweighted dose rates, estuarine/marine ecosystem

The recommended chronic dose rate of 400 μGy h^{-1} (Table 3.1) was not exceeded for marine organisms and below the dose rates at which effects were observed in Chapter 3 by two orders of magnitude. The impact on biota in the coastal ecosystem, based on this 'worst case' scenario, would therefore appear to be low, especially as the organisms receiving the highest doses are amongst the least radiosensitive.

Scenario 3: Terrestrial ecosystem

A composite scenario was created for the terrestrial grassland ecosystem. In this case it was assumed that the area was one of high rainfall with a Magnox power station discharging ^{14}C, ^{35}S and other radionuclides and a tritium plant situated nearby. In addition, the area would have received the maximum cumulative deposit of ^{90}Sr, ^{137}Cs and $^{239/240}Pu$ from weapons testing fallout and the Chernobyl accident.

The air (^{3}H, ^{14}C, ^{35}S) and soil (^{90}Sr, ^{137}Cs, ^{226}Ra, ^{238}U, $^{239/240}Pu$) concentrations given in Table 6.15 were used to 'drive' the assessment. These air and soil concentrations, along with the default CFs were used to generate predicted biota concentrations (Table 6.16). Measured concentrations were used to validate or adjust CF values to reflect the 'real world' situation (Table 6.16).

The predicted concentrations generally agreed with the measured concentrations. There are over-predictions for ^{137}Cs in grass and birds. The CF for ^{137}Cs in herbs was based on a mean of three measurements of ^{137}Cs in grass, ranging between 0.03 and 0.23. Factors contributing to the over-prediction may include the species of grass or the part of the plant assessed.

The CF used in calculating the concentration in birds is was derived from a vegetation-to-flesh concentration in grouse. Different feeding, nesting and roosting habits could account for the over-prediction. In addition, soil to plant transfer of ^{137}Cs in the upland mineral-deficient soils, typical of grouse habitat, is likely to be high, leading to a correspondingly high transfer to grouse. The default CF is therefore 'conservative', but not unrealistic.

Table 6.15 Concentrations of radionuclides in air or soil

	Radionuclide	Concentration
Air (Bq m^{-1})	^{3}H [a]	20
	^{14}C [b]	0.1
	^{35}S [b]	0.1
Soil (Bq kg^{-1})	^{90}Sr [c]	10
	^{137}Cs [c]	130
	^{226}Ra [d]	30
	^{238}U [d]	30
	$^{239/240}Pu$ [c]	1

[a] data from Chapelcross
[b] data from Hinkley Point
[c] data from Cumbria, cumulative fallout
[d] typical natural concentration

Table 6.16 Terrestrial ecosystem scenario - biota concentrations

		Biota concentration (Bq kg^{-1})	
Radionuclide	**Biota**	**Measured**	**Predicted**
^{14}C [a]	Leafy vegetables	2.00E+01	5.63E+01
	Wheat	7.70E+01	4.75E+02
	Rabbit	2.60E+01	7.50E+01
^{35}S [a]	Leafy vegetables	2.00E+01	1.50E+01
	Wheat	<8.00E-1	5.00E+00
	Grass	6.00E+00	1.50E+01
^{90}Sr [b]	Grass	2.00E-01	-
^{137}Cs [c,d]	Grass	<5.00E-1	1.86E+01
	Chicken	<4.00E-1	2.08E+02
^{226}Ra [e]	Rapeseed	2.00E-01	5.77E+00
^{239}Pu [d]	Chicken	4.00E-04	-

[a] data from Hinkley Point
[b] data from Chapelcross
[c] data from Wylfa
[d] data from Trawsfynydd
[e] data from Somerset

The dose rates do not include contributions from ^{129}I, where no data were identified to initiate the assessment. Weighted (Figure 6.6) and unweighted (Figure 6.7) dose rates were calculated and compared with the recommended dose limit of 40 μGy h^{-1} (Table 3.1). Weighted doses showed that bacteria received the highest dose of ~ 14 μGy h^{-1} (main contribution from ^{238}U and ^{226}Ra), followed by herbivorous mammals at ~ 10 μGy h^{-1} (main contribution from ^{226}Ra). The remainder of the

reference organisms would receive dose rates < ~ 5 μGy h^{-1}. The same pattern was observed in unweighted doses, with bacteria at ~0.8 μGy h^{-1}, followed by herbivorous mammals at ~ 0.7 μGy h^{-1}. The remaining terrestrial biota would receive doses in the region of <0.4 μGy h^{-1}.

A large proportion of the dose to organisms therefore appears to derive from naturally occurring radionuclides in the terrestrial ecosystem. It should, however, be noted that these calculated doses from ^{226}Ra and ^{238}U may be an over-estimate from 'natural background'. Naturally occurring uranium and thorium series elements will largely be incorporated within the mineral matrix in soil, and be relatively unavailable for uptake by biota. However, default concentration factors for these radionuclides reflect results from uranium mining areas with a technologically enhanced input. Concentration factors in this situation are likely to be higher because nuclides may be present in different forms, e.g. adsorbed onto the surface of soil particles rather than being incorporated within the mineral matrix itself.

Calculated doses are well below the recommended dose rate guidelines and below the dose rates at which effects were observed in Chapter 3 particularly considering the criteria for assessment in Section 6.7. Thus the impact on biota in the terrestrial ecosystem, based on this 'worst case' scenario, would therefore appear to be low according to this assessment methodology.

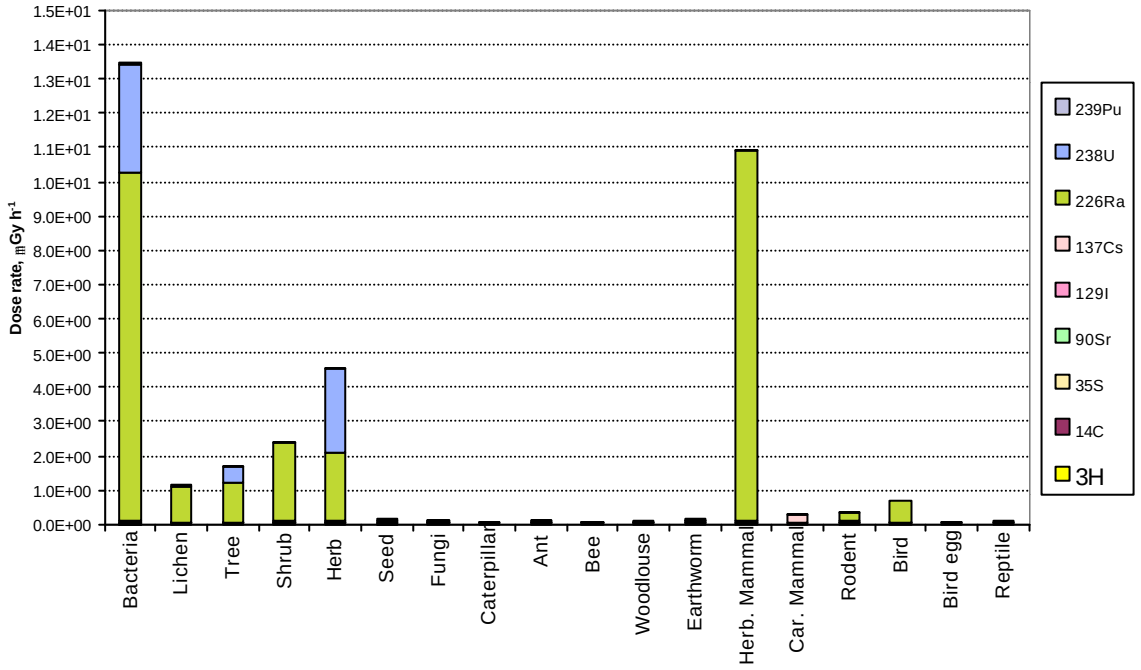

Figure 6.6 Weighted dose rates, terrestrial ecosystem

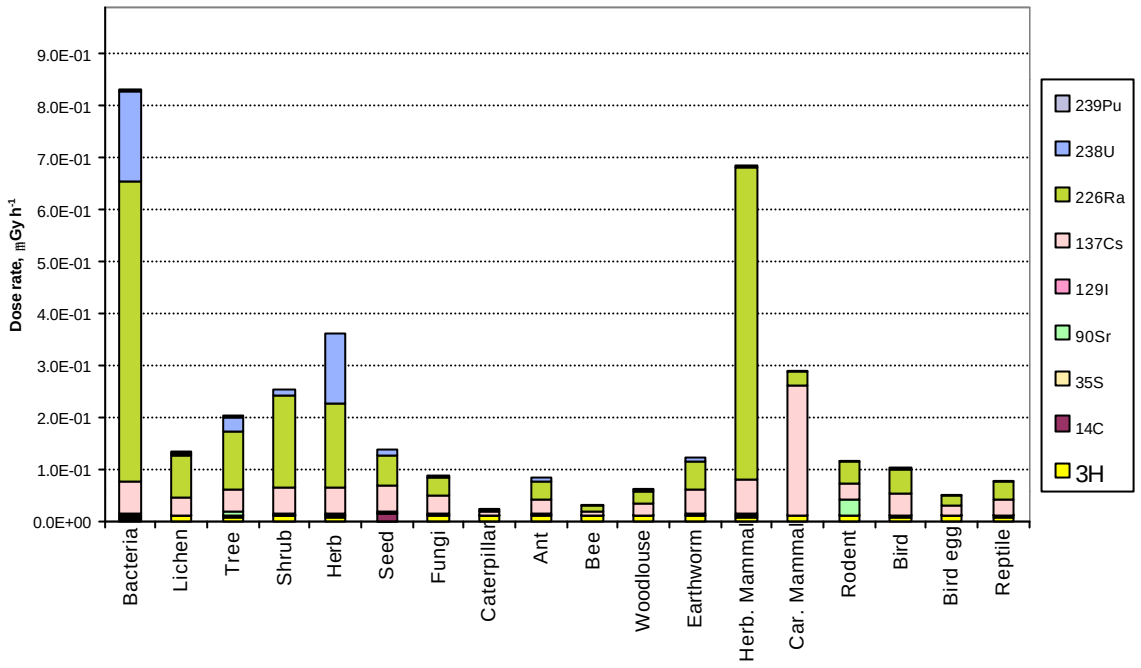

Figure 6.7 Unweighted dose rates, terrestrial ecosystems

6.9 Summary

This Chapter describes an approach to assessing the impact of ionising radiation from authorised discharges on wildlife, supported by Excel spreadsheets. The approach is illustrated in Figure 6.1.

Three ecosystems are assessed for a selected range of radionuclides (6.1) in order to produce a generic assessment that can incorporate site specific considerations when required.

The approach relies upon a number of assumptions (also listed in Chapter 5):

- Organisms are represented as ellipsoids
- Concentrations of radionuclides in biota are calculated using simple equilibrium concentration ratios between biota and water, soil or air.
- Radionuclides are considered to be distributed uniformly through all tissues of the animal or plant.
- Resulting absorbed doses, both internal and external, are calculated as an average throughout the volume of the organism.
- Doses are calculated as dose rates from equilibrium concentrations of radionuclides in biota.
- Organisms receive external dose at a reduced rate during the fraction of their time spend above ground surface, e.g. birds flying or roosting
- Absorbed fractions for á emissions are assumed to be zero for bacteria and unity for all other organisms.
- Calculated doses to micro-organisms are equal to the absorbed dose in the soil or sediment in which they are located.
- There are gaps in the data for the concentration factors for some reference organism/radionuclide combinations.

- There are limitations with the modelled approach for ^{3}H and ^{35}S, which should be taken into account when considering the calculated dose results. The chemical form of 3H is important and should be assessed. The short half life of ^{35}S means that the modelled approach may have limitations, although the significance of any impact on the calculated doses is likely to be small.

The selected default CFs appear appropriate for use in England and Wales and the limited validation exercise provides confidence in the calculated results. Provisions have been made in the assessment spreadsheets to allow user-defined, site specific CFs and habit data to be added to allow more accurate predictions to be made. The inclusion of site specific information is recommended. Although there are gaps in the available data for the concentration factors, the dose assessment spreadsheets do calculate an external dose. In the majority of cases, the external dose will give a reasonable estimate of the total dose to the organism. The internal dose becomes significant when radiation-weighting factors are applied to the alpha and low energy beta emitting radionuclides.

Step methodology is described (Section 6.5.1) and an operating guide for the spreadsheets is provided in Appendix 3 and on the CD ROM.

There are gaps in radioecological knowledge that should be addressed in future research and monitoring programmes. The most significant data gaps are on the transfer of radionuclides to:

- marine mammals (other than radiocaesium);
- seabirds and waders;
- in UK freshwater environments, with particular gaps in respect of plankton, amphibians, aquatic mammals and waterbirds; and
- sparse data on concentration factors of radionuclides to organisms in the terrestrial environment, particularly for bird eggs, reptiles and the larger mammals.

The assessment of the dose to biota using the 'worst case' scenarios provided in Section 6.8 demonstrate that the current impact of ionising radiation from authorised discharges on wildlife in England and Wales is low. The estimated dose rates to wildlife in all three ecosystem scenarios are well below the recommended ‘dose guidelines’ (Table 3.1) and the dose rates at which significant effects that may affect a population have been reported (Chapter 3). Site specific information should always be incorporated into the assessment where available, and further comparisons with the effect levels in Chapter 3 may be appropriate when considering risks to particularly sensitive species and ecosystems (Section 6.7).

It is however essential to recognise that any assessment of authorised discharges considers also the potential impact of other radionuclides that have not been incorporated into this assessment (i.e. those not included in Table 6.5). However, the application of reasoned judgement using the information in this report should allow sensible conclusions to be drawn in most cases.

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7. Conclusions and Recommendations

This report reviews the latest research on the pathways and effects of radionuclides on wildlife as part of the development of an impact assessment approach to determine the impact of ionising radiation on wildlife from authorised discharges in England and Wales. Figure 1.1 outlines the impact assessment process identifying the information and steps required. The report provides a series of Tables in Chapter 3 which outlines the effects of ionising radiation at different doses or dose rates. The outputs from the assessment spreadsheets, which are included with the report, can be compared with these Tables to assist in judging the level of impact on wildlife.

More specifically the conclusions are:

- **Pathways of exposure of wildlife from ionising radiation**

Radionuclides can enter ecosystems by many routes and become widely dispersed within their component parts. The behaviour of radionuclides in soil and sediment determines the impact of ionising radiation on biota in both terrestrial and aquatic ecosystems. Some aspects of the behaviour of radionuclides in soils are still poorly understood, particularly with respect to chemical form and bioavailability for uptake. As a result wildlife can be exposed to ionising radiation though a number of different routes including:

- External irradiation;
- Plant root uptake from soil;
- Foliar absorption;
- Inhalation of:
  - resuspended material;
  - gaseous radionuclides;
- Ingestion of:
  - plant material;
  - animal material;
  - microbial material;
  - soil;
  - water.

However, most of the studies reviewed demonstrate that the transfer of radionuclides through successive trophic levels is limited, with ^{137}Cs and ^{90}Sr being the most biologically mobile. However, only a relatively small number of radionuclides have been studied in terms of their environmental behaviour with respect to the possible radiation exposure of wildlife. This is mainly because releases of some radionuclides are low, and/or because analytical techniques are difficult and costly. This lack of information on specific radionuclides is a limitation in our ability to understand and account for the risks to wildlife associated with exposure to ionising radiation.

When undertaking an impact assessment of exposure to ionising radiation, it is necessary to consider the importance of seasonal and spatial variation in radionuclide concentrations.

Data are sparse on the behaviour and pathways of naturally occurring radionuclides to wildlife, particularly for the terrestrial ecosystem. Most studies have investigated the impact of uranium mine discharges to aquatic ecosystems. Most information is available for ^{40}K, ^{210}Po, ^{226}Ra, ^{238}U and ^{232}Th and assesses the geochemistry rather than biological uptake. The uptake of ^{222}Rn has also been assessed but mainly from the human perspective.

Naturally occurring radionuclides can give rise to high doses compared with anthropogenic sources. This is an area, which requires further research to establish the consequences and impact of natural

exposure. This is supported by the impact assessment scenarios carried out, in which it is clear that a significant contribution to the dose arises from naturally occurring radionuclides.

- **Effects of ionising radiation on wildlife**

The effects of radiation can be divided into two categories: stochastic and deterministic effects. The probability of inducing a stochastic effect increases with dose, but the severity of the effect is unrelated to dose. Deterministic effects are induced at doses above a threshold, above which the severity of effect is linked to dose received. It is generally thought that for wildlife populations deterministic effects are likely to be more important.

Damage induced at the molecular level of an individual may be propagated to successively higher levels of biological organisation; cell, tissue, organs, individual, population, community and ecosystem. The effects of ionising radiation are most easily observed at an individual level. Mortality, fertility, fecundity and genetic mutations are all individual end-points that may induce a significant impact at the population level.

Past research on wildlife has centred on the effects of acute radiation at an individual level, predominantly from external γ irradiation rather than internal exposure from mixed radiation types. Radioactive discharges in the environment generally result in chronic low level irradiation, thus studies on chronic irradiation are considered to be the most useful in investigating the impact of ionising radiation on wildlife.

Research into the biological effects of ionising radiation has focused on mammals, often in laboratory experiments. It is difficult to extrapolate these data to assess effects on wildlife in natural systems because of the lack of consideration for other stressors that may be present in the natural environment. Nuclear accidents, such as Kyshtym and Chernobyl, have contributed to our understanding of the effects of ionising radiation on biota as a result of the field experiments that have taken place.

Sensitivity to ionising radiation varies between taxa/species, stage of development and endpoint examined. Radiosensitivity increases as the biological complexity of the taxa increases. Developing stages are considered to be more radiosensitive than adult stages. Generally reproduction is the most radiosensitive endpoint, with the reproductive system of females considered to be more radiosensitive than that of males.

Higher plants are more radiosensitive than lower plants. The order of sensitivity in plants is accepted to be coniferous trees> deciduous trees> shrubs> herbaceous plants> lichen> bryophytes and fungi.

Invertebrates including insects, soil and litter fauna are less radiosensitive than birds and mammals. Earthworms are considered to be one of the most radiosensitive terrestrial invertebrates, possibly as a result of their sedentary nature and potentially high exposure pathway. Mammals are generally considered to be the most radiosensitive taxa. The impact of ionising radiation on birds, reptiles and amphibians has been studied to a lesser degree than mammals or plants. Birds appear to be slightly less impacted by radiation exposure than mammals, possibly as a result of their greater mobility, which reduces their exposure. Radiosensitivity of reptiles and amphibians to acute exposure is lower than that of birds and mammals.

There is no reported research into the impact of ionising radiation on aquatic mammals. Fish are considered to be one of the most radiosensitive of aquatic organisms, although aquatic mammals may also enter this category if more was known. Fish require a longer period of observation compared with terrestrial mammals before mortality due to acute exposure is apparent. Aquatic invertebrates are less radiosensitive than fish. No research has been conducted into the impact of ionising radiation on aquatic macrophytes, but lower plants, such as blue green algae, are less radiosensitive than aquatic animals.

- **Legislation**

The protection of humans from ionising radiation is well developed, with legislation in place to limit an individual's exposure. An internationally agreed framework to protect the environment from ionising radiation does not exist. Protection mainly relies on ICRP recommendations issued in 1977,

and modified in 1990, which effectively state that standards resulting in the protection of man will be sufficient to ensure protection of the environment. The ICRP recommendation has been criticised in terms of environmental protection, and international regulatory and scientific opinion has now recognised the need to protect the environment in its own right.

Stringent legislation concerning the use of nuclear materials, containment of radiation sources and discharges of radioactive waste exists in the UK. The Environment Agency in England and Wales has the responsibility to issue authorisations, which stipulate discharge limits and methods of disposal.

Legislation arising from the European Commission has increased the need to consider the impacts of discharges on the environment when issuing discharge consents for both radioactive and non-radioactive substances. Implementation of the Wild Birds and Habitats Directives has led to the creation of conservation areas collectively referred to as Natura 2000 sites. English Nature is a statutory consultee when considering applications for radioactive discharge consents that may impact these sites.

Individual countries have adopted different approaches to protect the environment from ionising radiation. Some approaches are based on the ICRP statement, some with specific requirements for environmental protection and some have implemented actual dose limits. The European Commission believes that understanding of radiation impacts on the environment is insufficient to permit the introduction of new legislation at a community level. As a result, a research programme (FASSET) has been funded to develop a framework for the protection of the environment from ionising radiation in Europe. The framework will provide a tool to assess environmental impact and judge compliance against environmental quality objectives, but not set standards.

- **Dosimetry methodology and assumptions**

Dose calculations recommended for impact assessment require information or estimates to be made of: the organism's dimensions; concentrations factors for the radionuclide under consideration; distribution of internal contamination; and the location of the organism relative to soil or sediment.

The proposed dosimetry model represents organisms as ellipsoids. Radionuclides are assumed to be uniformly distributed throughout the organism, thus the resulting internal dose is calculated as an average for the whole organism. For calculation of external doses, the fractional occupancy of key organisms is considered, whether underground, on the soil/sediment surface, or fully immersed in air or water.

External dose rates are evaluated using radionuclide concentrations in soil, sediment, and water. Density differences between the organism and the medium are neglected, and it is assumed that the external dose is evenly distributed within the organism. The dose to each organism per unit concentration of internally incorporated radionuclides is also determined. Concentration factors specific to each radionuclide and organism (relative to soil, water or air) are applied in order to estimate internal concentration.

As the damage induced by radiation is dependent upon the LET of each radiation type, recommendations on appropriate weighting factors have been made.

Radiation weighting factors recommended are:

- 20 for á radiation;
- 3 for low energy â radiation (<10 keV);
- 1 for â radiation greater than 10 keV and ã radiation.

These values are applied by default in the assessment spreadsheets provided with this report, but may be changed by the user.

There are a number of important assumptions/caveats, which must be considered when using the spreadsheets however. These are:

- Organisms are represented as ellipsoids

- Concentrations of radionuclides in biota are calculated using simple equilibrium concentration ratios between biota and water, soil or air.
- Radionuclides are considered to be distributed uniformly through all tissues of the animal or plant.
- Resulting absorbed doses, both internal and external, are calculated as an average throughout the volume of the organism.
- Doses are calculated as dose rates from equilibrium concentrations of radionuclides in biota.
- Organisms receive external dose at a reduced rate during the fraction of their time spend above ground surface, e.g. birds flying or roosting
- Absorbed fractions for á emissions are assumed to be zero for bacteria and unity for all other organisms.
- Calculated doses to micro-organisms are equal to the absorbed dose in the soil or sediment in which they are located.
- There are gaps in the data for the concentration factors for some reference organism/radionuclide combinations.
- There are limitations with the modelled approach for ^{3}H and ^{35}S, which should be taken into account when considering the calculated dose results. The chemical form of 3H is important and should be assessed. The short half life of ^{35}S means that the modelled approach may have limitations, although the significance of any impact on the calculated doses is likely to be small.

- **Impact Assessment**

The dose models described above have been subject to limited validation within this report and it is recommended that they be used in impact assessments to evaluate the impact of ionising radiation on wildlife from authorised discharges in England and Wales. The models have been produced as standalone spreadsheets for estimating doses to biota in the following scenarios:

- Estuarine/marine and freshwater ecosystems: ^{3}H, ^{14}C, ^{99}Tc, ^{90}Sr, ^{137}Cs, $^{239+240}Pu$, ^{238}U, ^{129}I and ^{210}Po
- Coastal grassland ecosystem: ^{3}H, ^{14}C, ^{35}S, ^{90}Sr, ^{137}Cs, $^{239+240}Pu$, ^{238}U, ^{129}I and ^{210}Po

for a representative range of biota within each ecosystem.

- For freshwater ecosystem: bacteria, macrophyte, phytoplankton, zooplankton, benthic mollusc, small benthic crustacean, large benthic crustacean, pelagic fish, benthic fish, amphibian, duck, aquatic mammal.
- For estuarine/marine ecosystem: bacteria, macrophyte, phytoplankton, zooplankton, benthic mollusc, small benthic crustacean, large benthic crustacean, pelagic fish, benthic fish, fish egg, seabird, seal, whale.
- For terrestrial ecosystem: bacteria, lichen, tree, shrub, herb, seed, fungus, caterpillar, ant, bee, wood louse, earthworm, herbivorous mammal, carnivorous mammal, rodent, bird, bird egg, reptile.

A database of concentration factors for these organisms and radionuclides has been developed (Tables 6.7 to 6.9) for use in the impact assessment process. Tests on the validity of these concentration factors have proved successful, giving confidence that a generic assessment performed using the concentration factors identified will provide a result which is likely to be, if anything, over cautious. It is however recommended that wherever possible site-specific information should be used in the dose calculation spreadsheets to improve the assessment.

The concentration factors adopted have been tested under scenarios considered to be realistic for the situations encountered in England and Wales. The results have been compared with known concentrations of radionuclides in biota for given discharges thus providing further evidence that the information underlying the impact assessment is valid.

Guidelines on how the impact assessment should be undertaken are provided along with operating instructions for the spreadsheets included with this report. Provided these guidelines are followed any impact assessments undertaken will be in line with current knowledge and, broadly, and also in line with approaches being adopted in other countries. Whilst there are no hard and fast rules for interpreting the dose results obtained from the spreadsheets, reasoned judgement using the information contained within this report should allow sensible conclusions to be drawn in most cases about the impact of authorised discharges of radioactive materials on wildlife

The results from realistic but hypothetical worst case scenarios in England and Wales indicate that the doses received by wildlife are considerably lower than the current guideline values suggested by the IAEA (1992) which this report has adopted. Therefore it can be concluded from the assessments carried out by the authors that the current impact of ionising radiation from authorised discharges on wildlife in England and Wales is low. There are however a number of caveats which must be considered in drawing such a conclusion and these are described in greater detail in Chapter 6. This can however be summarised as follows:

- being limited to the radionuclides included in the assessment;
- the adopted values for radiation weighting factors may change in the future as more information becomes available;
- there may be other scenarios, for example, with vulnerable ecosystems in which wildlife may be exposed to higher levels of ionising radiation;
- general lack of information on certain radionuclides;
- concentration factors may not be available for all species and situations.

- **Recommendations for future research**

More research is required to develop a database of the concentration factors necessary to estimate internal radionuclide concentrations for reference organisms. Much of the data that exist for concentration factors are not applicable in its current form. There is also little information available on the transfer and uptake of naturally occurring radionuclides to wildlife, particularly in terrestrial ecosystems.

- Appropriate assessment methods for spatial, temporal and averaging of doses is required.
- The role of chemical speciation in determining radionuclide availability and studies on less frequently studied radionuclides (e.g. ^{129}I because of the analytical difficulties involved) are required.
- The relevance of biomarker techniques to the long-term health of individuals and populations needs to be determined.
- Assessment of the impact of major accidental releases on wildlife populations will provide an insight into the likely consequences of exposure to ionising radiation at the level of the individual, population and community. The Chernobyl's exclusion zone represents a unique opportunity as a field laboratory.
- Assessment of the impact of ionising radiation on specific species is required. This may include rare and endangered species inhabiting areas around point source releases, and long-lived species (particularly marine).
- Additional experiments to determine the effects of low level, continuous exposure to ionising radiation on different plant and animal taxa to fill in the gaps in Tables 3.6 to 3.19, particularly

at the lower (chronic) doses typical of environmental conditions resulting from authorised discharges.

- Further radioecological research is required to identify concentration factors, which can be used to estimate the internal concentrations of radionuclides in relevant wildlife species. As a starting point the gaps identified in Tables 6.7 to 6.9 should be addressed.
- Models to predict radionuclide transfer and behaviour in the environment should be developed further, and rigorously tested. By linking such models to the assessment process, the risks associated with authorised discharges (from the past, present and future) can be determined.
- Assessment of the impact of ionising radiation in conjunction with exposure to other non-radioactive pollutants is required. In a regulatory framework, greater emphasis should be placed on the interaction of pollutants at any one site.

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A1 Ecological Consequences of the Chernobyl Accident

Content

Appendix 1

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A1.1 Introduction

The Chernobyl accident in April 1986 has been described as the one of the greatest technological disasters in the world (Savchenko, 1995). The explosions resulted in an uncontrolled release of radioactivity into the environment.

Experiments to test the turbine generators' supply of energy for a limited period, operator mishandling, violations of operating procedures and inherent design faults contributed to the accident. The explosions caused movement of the reactor cover plate, exposing the nuclear material in the core to the environment and emitting radioactive material. Uncontrolled releases continued for several days before a nitrogen coolant was pumped through tunnels specially constructed underneath the core and led to the cessation of emissions.

This Chapter reviews the deposition of Chernobyl derived fallout and updates the information reported on the ecological effects of the accident. A large number of field studies investigating the ecological impact of the Chernobyl derived fallout have been conducted, particularly within close proximity to the accident site but also in other countries that received Chernobyl fallout e.g. Sweden and Belarus. No recent studies could be found that specifically relate to the impact of radioactive material deposited from Chernobyl on biota within the UK.

A1.2 Deposition and accumulation of Chernobyl derived radionuclides

Some 2 x 10^{18} Bq of activity was released representing between 3 and 4% of the core inventory. Much of the material released into the atmosphere consisted of spent fuel, noble gases and volatile radionuclides such as isotopes of caesium, iodine and tellurium. The composition of the radionuclides released resembled those in the fuel but with preferential release of the more volatile radionuclides. For example, 20% of the iodine available in the core was released (6.7 x 10^{17} Bq), 10% of the caesium (1.9 x 10^{16} Bq ^{134}Cs, 3.7 x 10^{16} Bq ^{137}Cs) and about 3% of the rare earths and actinides (Table A1.1).

Table A1.1 Radionuclides of radiological significance released during the Chernobyl accident (Fry, 1987)

Radionuclide	*Activity, 10^{15} Bq*	
	By day 1	*By day 10*
^{131}I	170	440
^{134}Cs	5	25
^{137}Cs	10	50
^{90}Sr	0.5	9
$^{2309+240}$Pu	0.1	0.7

Most of the radionuclides from the fallout were short lived (<1 year) with the exception of ^{134}Cs and ^{137}Cs which have half lives of 2 and 30 years respectively. Their ^{137}Cs:^{134}Cs ratio approximated 2:1 (Table A1.1). As ^{134}Cs was not present in weapons testing fallout, it has been extensively used as an indicator of Chernobyl derived fallout.

In addition to the 30 km exclusion zone, an area of 240,000 km^2 was contaminated with ^{137}Cs at a deposition density of greater than 200 kBqm^{-2}, 5,710 km^2 at a density greater than 600 kBqm^{-2} and 1,360 km^2 at a density greater than 1.5 MBqm^{-2} (IAEA, 1986).

A1.2.1 The exclusion zone

Two plumes carried the majority of radionuclides released by the accident; one to the north-west and the other to the south-west of the Chernobyl nuclear power plant (NPP). This resulted in patchy radionuclide deposition. Based on the deposition patterns around the reactor at the time of the accident an exclusion zone of 30km radius was imposed to limit human activities and thus reduce exposure of humans to ionising radiation (Figure A1.1). A second inner exclusion zone (10km in radius) was further implemented within which movement is more strictly controlled by border guards.

Within weeks of the accident, it became evident that the exclusion area could be divided into three zones based on acute doses received (Kozubov *et al.*, 1990):

- Zone 1, approximately 500-600 hectares, nearest the reactor experienced a dose of between 80 and 100 Gy,
- Zone 2, approximately 3,000 hectares, received an estimated 8-10 Gy, and
- Zone 3, approximately 12,000 hectares, received 3.5-4 Gy

Much of the contamination inside the exclusion zone comprises of particles of irradiated fuel. These fuel fragments are generally insoluble and therefore not biologically available. ^{134}Cs, ^{137}Cs, and ^{131}I were also deposited in large quantities (Konoplev and Bobovnikova, 1990). Radiation levels have since declined as a result of decay, dispersion and remediation practices.

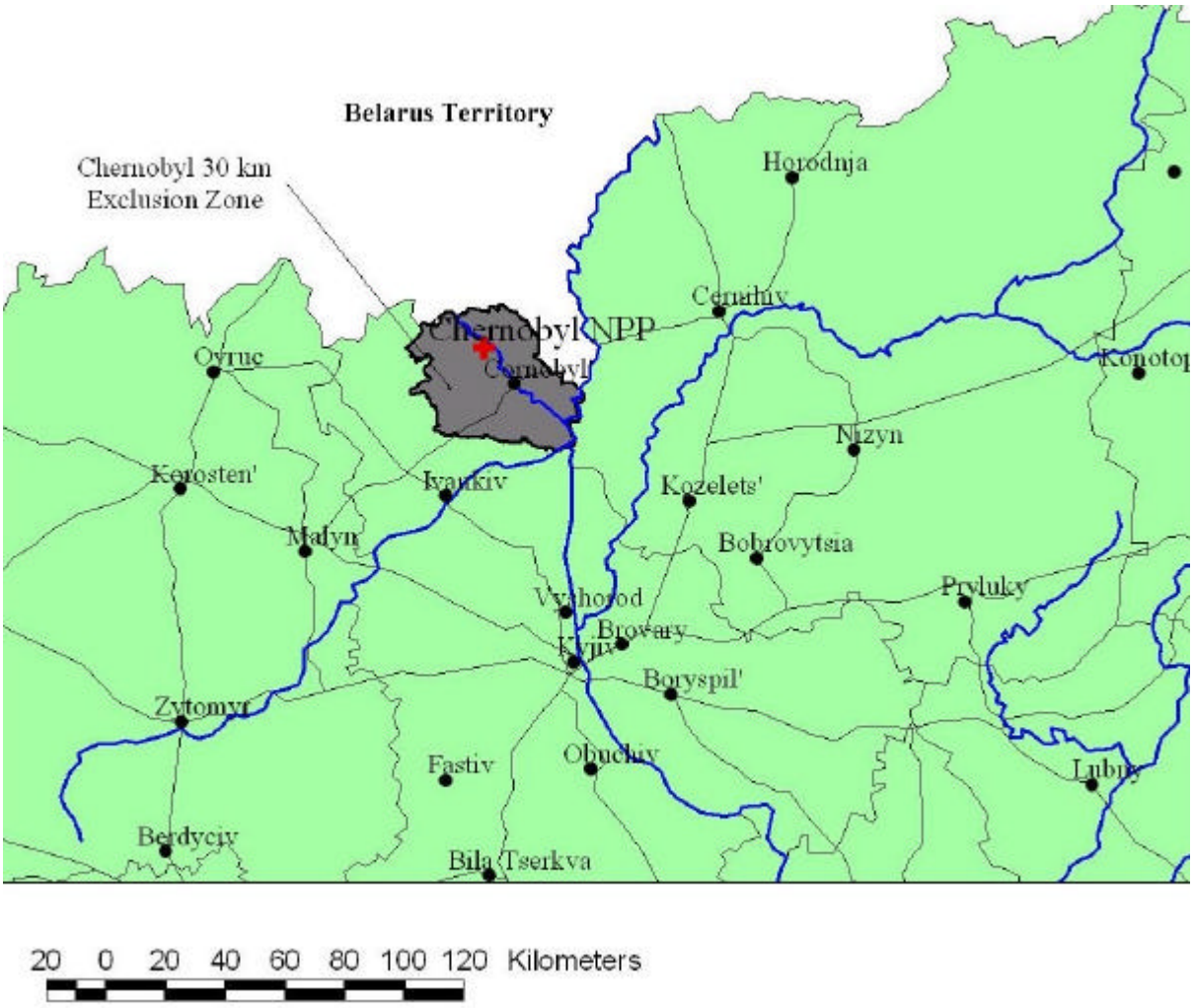

Figure A1.1 Sketch map of the Chernobyl exclusion zone and surrounding territories. Data provided by Wright and Arkhipov (pers. comm.)

- **Soil**

  Following the accident, the radioactivity concentrations in the soil within the 30 km zone varied as a consequence of the release from the accident. The concentration of the radionuclides present decreased in the following order $^{137}Cs>^{90}Sr>^{144}Ce>^{134}Cs>^{241}Am>^{125}Sb>^{154}Eu>^{155}Eu$ (Askbrant *et al.*, 1996). The soil has since stabilised primarily as a result of deactivation measures, land improvement and disintegration of short-lived radionuclides. By 1990 the major radionuclides remaining were ^{137}Cs and ^{90}Sr. Much of the ^{137}Cs and ^{90}Sr deposited in the 30km zone has been retained in the superficial layers of the soil and is likely to remain there for a long time. Furthermore, within the inner (10 km) exclusion zone levels of ^{239}Pu and ^{240}Pu are still present (Sokolov *et al.*, 1993).

  Surface ^{137}Cs in soil samples collected in 1992 was 1,100-1,500 $kBqm^{-2}$ at areas of 'high' contamination and 180-700 $kBqm^{-2}$ in areas of low contamination. Similar activity concentrations were observed in spring 1995 (Eriksson *et al.*, 1996).

- **Vegetation**

  Levels of vegetation contamination decreased with distance from the site. In 1987, ^{137}Cs concentrations in birch catkins was 113 $kBqkg^{-1}$ at 1.5 km from the accident site, decreasing to 65 and 25 $kBqkg^{-1}$ at 6 and 18 km respectively. Over the next two years the ^{137}Cs content of seeds (catkins) and leaves increased up to 9-10 fold in trees near the Chernobyl NPP (Yushkov *et al.*, 1990).

  ^{90}Sr contamination of birch catkins and leaves followed a similar pattern to ^{137}Cs. ^{90}Sr concentrations in the vegetation also increased within 2 years of the accident, to a lesser extent than ^{137}Cs. The build up of ^{137}Cs and ^{90}Sr in reproductive organs and leaves of birch three years post accident was attributed to the uptake of radionuclides from the soil (Yushkov *et al.*, 1992).

  Average ^{137}Cs levels in evening primrose collected in 1992 were 1-5 $kBqkg^{-1}$ dry weight and 15-30 $kBqkg^{-1}$ dry weight in higher contamination areas. Concentrations peaked at 74 $kBqkg^{-1}$ dry weight in samples collected from just north of the power plant (Eriksson *et al.*, 1996). Similar activity concentrations were observed in samples collected in spring 1995 (Eriksson *et al.*, 1996).

- **Animals**

  In 1992 the minimum level of ^{137}Cs in wild boar muscle was 2 $kBqkg^{-1}$ wet weight (in spring), with a maximum of 20 $kBqkg^{-1}$ wet weight (in winter). ^{137}Cs levels in roe deer ranged from 6-10 $kBqkg^{-1}$ wet weight. Similar concentrations were still observed in spring 1995 (Eriksson *et al.*, 1996).

  ^{137}Cs muscle concentrations of free-ranging small mammals sampled between 1994-1996 from the 10 km exclusion zone was 3,200 $kBqkg^{-1}$, whilst ^{90}Sr in bone was 297 $kBqkg^{-1}$. ^{137}Cs in muscle of small mammals captured 30 km south-east of the reactor averaged only 2 $kBqkg^{-1}$ (Chesser *et al.*, 2000).

- **Aquatic environment**

  The Chernobyl cooling pond is considered to be the most contaminated water body in the exclusion zone. In May 1986, the ^{137}Cs deposited into its sediment was estimated to be 110×10^9 kBq, and 60×10^9 kBq in its water. ^{90}Sr deposition was estimated as 50×10^9 kBq into the sediment and 6×10^9 kBq in water (Kryshev, 1995). ^{137}Cs concentrations in sediments were recorded as greater than 100 $kBqkg^{-1}$ in 1993 (Jagoe *et al.*, 1998).

  ^{137}Cs concentrations in the muscle of fish collected from ponds within 10 km of the power plant ranged between 6 and 192 $kBqkg^{-1}$ in 1993 and correlated with sediment concentrations (Jagoe *et al.*, 1998). The Kiev reservoir located to the south-east of the Chernobyl NPP lies outside of the 30 km exclusion zone and, comparatively, received less fallout than areas to the north and west. Concentrations of ^{137}Cs in crucian carp collected from the Kiev reservoir in 1993 were 0.55 $kBqkg^{-1}$ (Jagoe *et al.*, 1998).

A1.2.2 The United Kingdom

Following the accident, a radioactive plume travelled a considerable distance from the site as a result of the ambient meteorological conditions at the time, depositing radionuclides throughout many countries of Europe. Where the plume's passage coincided with periods of heavy rainfall, deposition levels were 30-40 times that of normal background as far as 1500 km from the Chernobyl NPP (Hohenemer *et al.*, 1986; ApSimon *et al.*, 1988; Clark and Smith, 1988).

The radioactive plume from Chernobyl passed over the UK on 2nd May 1986 and coincided with a period of heavy rainfall. Patchy deposition patterns were observed particularly in upland areas such as Cumbria, Gwynedd, areas of Yorkshire and south-east Scotland. The highest deposition was 6.7 $kBqm^{-2}$ at sea level near Barrow in Furness, Cumbria, whilst in excess of 3.0 $kBqm^{-2}$ was reported on the western fells (Kennedy *et al.*, 1990). The highest deposition in north Wales was 1.8 $kBqm^{-2}$ and around 2.0 $kBqm^{-2}$ near East Kilbride, Scotland (Kennedy *et al.*, 1990).

- **Soil**

  ^{137}Cs in soil samples collected pre and post Chernobyl are shown in Table A1.2. Over half of ^{137}Cs pre-Chernobyl soil samples were attributed to atmospheric discharges from Sellafield, with relatively little ^{134}Cs (Rudge *et al.*, 1993a). Post Chernobyl, several γ emitting radionuclides were present in the soil, ^{137}Cs, ^{134}Cs and ^{40}K being the most abundant. Lower levels of ^{137}Cs were deposited in Cheshire consistent with rainfall pattern (Table A1.2). Between 30-50% of the ^{137}Cs in the top 15cm of soil could be attributed to Chernobyl, with the percentage varying proportionally with the amount of rainfall received after the deposition occurred.

Table A1.2 Concentrations of ^{137}Cs in soil cores (14 cm) collected from Cumbria and Cheshire pre and post Chernobyl (based on Rudge et al., 1993a)

Location	*Pre Chernobyl (Bqm^{-2})*	*Chernobyl (Bqm^{-2})*	*Chernobyl ^{137}Cs (%)*
Drigg	15 170 ± 80	7330 ± 60	32.6 ± 0.3
Cheshire	>1,030	<230	<18.3

  In addition to ^{137}Cs, minor nuclides deposited following the Chernobyl accident included; ^{110m}Ag, ^{141}Ce, ^{103}Ru and ^{106}Ru, and were detected almost exclusively in the upper 4 cm of the soil (Rudge *et al.*, 1993a).

- **Vegetation**

  Contamination of plant material can occur via two main processes: direct contamination of the plant surface and root uptake. The direct contamination of foliage is generally less significant in areas of little or no atmospheric discharges of radionuclides. Root uptake is therefore the most important transfer pathway from soil contaminated with radionuclides.

  Following the Chernobyl accident direct contamination of plant foliage provides an immediate and significant transfer pathway for plants as well as to higher trophic levels. This is particularly true for ^{137}Cs (Rudge *et al.*, 1993a,b)

  Soil type affects the retention and recycling of radionuclides in the environment. A large proportion of the contaminated upland areas in Wales, Cumbria and Scotland consists of organically rich acidic soil which maintained ^{134}Cs and ^{137}Cs in a bioavailable form for root uptake (Kennedy *et al.*, 1990).

  A number of studies on the transfer and uptake of radionuclides following the accident were undertaken. These provided further evidence that radionuclide accumulation differs between plant species. For example, lower plants, bryophytes and lichens tend to accumulate radiocaesium to a greater extent than higher plants (Livens *et al.*, 1991). The Ericaceaea family (heathers) are

particularly efficient at accumulating radiocaesium although this is related also to the retention and recycling of caesium in the organic rich soils on which heathers thrive (Horrill *et al.*, 1990).

Concentrations of ^{137}Cs in grass collected pre and post Chernobyl in the UK are shown in Table A1.3. Post Chernobyl, there was a rapid increase in ^{137}Cs levels followed by a general decrease until late 1987. The decrease was attributed to the loss of radiocaesium from the plant tissues to the soil and to an increase in plant material through seasonal growth which diluted the radionuclides present. A second peak in ^{137}Cs concentrations was observed in the autumn of 1987, due to translocation of ^{137}Cs from old to new tissues, particularly storage organs, prior to the senescence of the plants or increased ^{137}Cs availability as a result of bacterial activity in the soil. Grasses sampled in the early spring of 1988 had reduced ^{137}Cs concentrations but still much higher than pre-Chernobyl levels (Rudge *et al.*, 1993a).

Table A1.3 Concentrations of ^{137}Cs in grasses from Drigg and Cumbria (based on Rudge et al., 1993a)

Location	*Pre Chernobyl (Bqkg^{-1})*	*Post Chernobyl (Bqkg^{-1})*
Drigg	5.9-12.2	110-1,600
Cheshire		4.1-13

Sandalls & Bennett (1992) reported that ^{137}Cs in grasses collected from upland areas in West Cumbria varied to a greater extent than soil levels, with concentrations ranging from 400-4,000 kBqkg^{-1}. Soils contained both pre-Chernobyl and Chernobyl depositions of ^{137}Cs, whilst levels in vegetation were considered to arise almost entirely from the Chernobyl fallout. This was attributed to the greater contribution of Chernobyl derived ^{137}Cs to total ^{137}Cs within the vegetation rooting zone. Older ^{137}Cs from to weapons testing fallout was thought to have had migrated through the soil profile to below the rooting zone. It was concluded that uptake of Chernobyl derived ^{137}Cs into vegetation would similarly decrease. However in upland organic soils reduction in plant ^{137}Cs activity concentration is likely to be slow as radiocaesium is recycled (Beresford *et al.*, 1992).

- **Animals**

Levels of ^{137}Cs in worm tissues collected from Drigg, Cumbria, were comparable with that in soil samples, but 3-4 fold higher in intact worms (tissue and gut). ^{137}Cs concentrations in whole earthworms decreased from 1.5 Bqkg^{-1} in early summer 1986 to 0.25 Bqkg^{-1} in the same period of 1987. This pattern was consistent with a reduction in mean concentrations observed for vegetation sampled at the same time (Rudge *et al.*, 1993a).

Herbivorous species, such as slugs and weevils, had ^{137}Cs body burdens lower than vegetation and detritus feeders (e.g. earthworms) from the same location (Rudge *et al.*, 1993a). Radionuclides incorporated into plant material are reportedly more available to herbivores than those deposited on plant surfaces. The concentration of ^{137}Cs in herbivores decreased between May 1986 and July 1987 at a rate consistent with that for radioactivity in vegetation from the same location. Predatory invertebrates like spiders also displayed lower body burdens of radionuclides than detritus-feeders, but with levels similar to herbivores (Rudge *et al.*, 1993a). Consequently, this study demonstrated that the accumulation of radionuclides is dependent upon diet and trophic level and varies between species.

The mean ^{137}Cs concentrations in rodents captured at Drigg pre-Chernobyl ranged from 7.2 Bqkg^{-1} dry weight in field voles to 26 Bqkg^{-1} in shrews, and increased by at least an order of magnitude in June 1986 following fallout deposition, e.g. shrews contained around 840 Bqkg^{-1}. Body burdens of ^{137}Cs also increased significantly at the Cheshire site post Chernobyl but the increase was smaller than that observed at Drigg with a maximum of 40 Bqkg^{-1} in shrews sampled in June 1986. Secondary autumn peaks in ^{137}Cs observed in field voles and shrews were attributed to seasonal increases in vegetation and detritus ^{137}Cs concentrations (Rudge *et al.*, 1993b).

A1.3 Impact on plants and animals

The UNSCEAR review (1996) described the impact of Chernobyl on plants and animals, concentrating on mammals living within the 30 km exclusion zone. The lack of dosimetry data makes it difficult to compare field investigations with reported laboratory results which used clearly defined experimental approaches with known radionuclide concentrations or doses.

A1.3.1 Exclusion zone

The 30 km exclusion zone is characterised mainly by low flat land and includes the flood plains of the river Pripyat and its tributaries. Agriculture accounted for approximately 50% of the land use and was abandoned after the accident. The remaining land was predominantly forest and swamp. Since the accident, this area has been reverting back to its natural state (Eriksson *et al.*, 1996).

- **Animals**

– Vertebrates

Within the 30 km exclusion zone acute doses to small rodents until mid May 1986 were estimated to be 20 Gy from γ and 880 Gy from β radiation (Testov and Taskayev 1990).

No evidence of mortality or migration of vertebrates under the direct effect of ionising radiation was recorded immediately after the accident (Sokolov *et al.*, 1993). The total number of a range of small mammal species and measures of species diversity did not differ between contaminated and uncontaminated sites (Sokolov and Krivolutsky 1998). However, the proportion of mature rodents present was higher at the contaminated site.

Over several generations, increased radiation background failed to affect the libido or reproductive capacity of northern redback voles. However the state of some female reproductive organs and embryogenesis, such as resorption, were affected along with increased deformation of extremities (Sokolov and Krivolutsky 1998). The radiation exposure did not influence the sex structure of the rodent population with the male :female ratio remaining 1:1 (Sokolov *et al.*, 1993).

Biomarker techniques have been used to monitor radiation induced molecular damage within the exclusion zone. Research has investigated the genetic impacts of the Chernobyl accident using bank voles because of their higher internal dose of ^{134}Cs, ^{137}Cs and ^{90}Sr compared with other rodents in the region. Genetic diversity in the bank vole population collected from the Red Forest (within the 10 km exclusion zone), estimated to receive 3.6 mGy h^{-1}, was compared with that of a reference site where doses did not significantly differ from background. Genetic diversity, measured as DNA mutation rates, in the Red Forest population (0.722 ± 0.024) is significantly higher than that of the reference population (0.615 ± 0.068) (Matson *et al.*, 2000).

Studies on northern redback voles from contaminated and uncontaminated sites did not show any differences in growth or development between the two populations (Sokolov and Krivolutsky 1998). Increases in liver and thymus masses were observed in bank voles from contaminated sites, whilst sub-adult shrews had larger body masses, and heavier spleens, kidneys and livers (Tsiperson and Soloviev, 1997). These changes reportedly reflect the physiological and immunological pressure exerted on small mammals as a result of chronic radiation.

– Reptiles and Amphibians

Brown frogs close to the Chernobyl NPP were found to suffer from reduced fertility (dose estimates unavailable), which was the only effect attributed to increased exposure within the exclusion zone reported for reptiles or amphibians (Cherdantsev *et al.*, 1993).

– Birds

The occurrence of genetic mutations giving rise to albinism in barn swallows close to the Chernobyl site were reported to be between two and ten times higher than in birds from control areas in Ukraine and Italy. Albinism is a morphological aberration associated with a loss of fitness (Ellegren *et al.*, 1997).

– Invertebrates

Radioactive fallout from Chernobyl induced marked reductions in the number of species in the litter microarthropod community of the forests within the 30 km exclusion zone. The impact was less pronounced on soil microarthropods and larger invertebrates. Populations generally recovered within 2-3 years (Krivolutsky and Pokarzhevsky, 1990).

Beetle populations 8 km away from the Chernobyl power plant studied between 1993-1996 were found to have an increased number of dwarfish beetles, with biomass negatively correlated to radionuclide deposition in the soil (Ipatyev, 1999).

The impacts of acute irradiation on animals and plants following the Kyshtym and Chernobyl accidents are summarised in Table A1.4.

- **Plants**

The Chernobyl accident occurred in spring as plants enter their period of accelerated growth and reproductive phase, the most radiosensitive phases of their life cycle. The above ground part of plants receive the highest doses of radiation resulting from atmospheric deposition. It has been estimated that the trees in the zone intercepted between 70 and 80% of the total radioactive material, with interception by pines being two to three times greater than that of deciduous trees (Ipatyev *et al.*, 1999). 63% of the species in the affected forests were coniferous, the remaining being deciduous species such as birch, aspen, alder and oak (Sokolov *et al.*, 1993).

By 1998, dose rates ranged between 100-5,000 μGy h-1 in the Red Forest, located within the 10 km exclusion zone (Matson et al., 2000). Table A1.5 shows the absorbed doses received by plants and invertebrates within 1 year of the Chernobyl accident.

Within three weeks of the accident lethal effects on pine trees in zone 1 were visible. Deciduous trees exhibited only partial damage (Table A1.6). It is estimated that 400 hectares of pine forest died, with an area the same size suffering damage (Izrael et al., 1988).

In 1987 morphological changes in birch trees were evident including abnormal coloration of leaves and twisted branches. These modifications were not permanent, and trees regained their normal foliage by 1988: 2 years after the accident (Sokolov et al., 1993) (Table A1.7).

Studies conducted 8 years after the Chernobyl accident reported higher than normal rates of seed chromosome aberrations in birches and pines growing in contaminated regions (Cherezhanova, 1998). Furthermore, the germinating capacity of herbace with that of seeds from control plots (Shevchenko *et al.*, 1998).

- **Aquatic biota**

Most of the work on radionuclides near Chernobyl has concentrated on terrestrial systems because of the potential impacts on human health, agriculture and forestry resources. However the region contains a large freshwater ecosystem including the Pripyat Marshes, one of the largest freshwater wetlands in Europe and the Pripyat river that drains into the Kiev reservoir.

Table A1.4 Observed effects within the first 15-60 days after the accidents at Chernobyl and Kyshtym (from Whicker 1997, based on Tikhomirov & Shcheglov 1994, Alexakhin 1993, Arkhipov et al. 1994, Kryshev 1992, Skuterud et al. 1994, Smirnov 1993)

Dose range (Gy)	*Impacts on plants*	*Impacts on animals*
<0.1	No visible damage, chromosome damage measured in spiderwort stamens	No data available
0.1-0.3	Minor reduction in growth and reproduction in pines Chromosome damage in pines measured	Impaired reproduction of rodents, chromosomal damage in mice
3-5	Growth inhibition and histological changes in pines	No visible change in fish populations
5-10	Severe growth reduction, needle damage, morphological change and sterility in pines Threshold for ecosystem level disruption	Decreased numbers of soil and litter fauna Physiological changes in rodents
10-25	Growth cessation and severe crown damage	Reductions in mouse populations
25-100	Severe mortality of pines, morphological damage Delayed sprouting and early leaf fall of deciduous trees Significant ecosystem disruption	Mortality of juvenile invertebrates
>100	Complete mortality of pines Severe crown damage in deciduous trees	No data available
>200	Lethality to deciduous trees	Lethality in rodents
>700	Damage to herbaceous communities	No data available

Table A1.5 Absorbed dose rates received by biological structures from external b and g radiations located in the forests of the Chernobyl zone, in the post accident period (Tikhomirov and Shcheglov, 1994)

Biological structure	*Absorbed Dose rate* [a]		
	Average 0-10 days	After 30 days	After 1 year
Leaves, needles	100 (90 + 10) [b]	20 (16 +4) [b]	1 (0.5 + 0.5) [b]
Populations of forest litter	20 (10 +10) [b]	12 (8+4) [b]	3 (2.5 + 0.5) [b]
Plant roots and soil animals	3	1	0.02

[a] 100 is the initial absorbed dose rate in leaves and needles

[b] Figures in parenthesis are ($\beta + \gamma$)

Table A1.6 Distribution of radiation damage in the forest around the Chernobyl nuclear power plant (from UNSCEAR (1996) compiled from the data of Kryshev et al. (1992) and Kozubov et al. (1990)

Damage area	*Type of damage*	*Absorbed dose from external g radiation (Gy)*	*Absorbed dose rate on 1 October 1986 (mGy h^{-1})*	*Absorbed dose in needles.*
Lethal Area 4km^2	Complete death of pine trees Partial damage of deciduous tress	>80-100	>5000	>100
Sub lethal Area 38km^2	Death of most growth points Partial die back of coniferous trees Morphological changes in deciduous trees	10-20	2000-5000	50-100
Area of medium damage 120 km^2	Suppressed reproductive ability Desiccated needles Morphological changes	4-5	500-2000	20-50
Area of minor damage	Disturbances in growth and reproduction Morphological disturbance in coniferous trees	0.5-1.2	<200	<10

Maximum dose rates to aquatic animals in the Chernobyl NPP cooling pond from external irradiation were estimated to range from 4,200-8,400 μGy h^{-1} (100,000 times greater than the natural background rate) (Kryshev and Sazykina, 1986). By June 1988, there was a 200-fold decrease in external irradiation from water attributed to the precipitation of radionuclides into the sediment and radioactive decay (Sokolov *et al.*, 1993).

The effects of irradiation on aquatic biota have also been studied in water bodies in the Southern Urals (in the vicinity of the Kyshtym accident), and around Chernobyl and the impacts are summarised in Table A1.8.

^{137}Cs in non-predatory fish collected from the Chernobyl NPP cooling pond was higher in bottom fish (silver bream) and plankton-eaters (silver carp) than in predatory fish in the first year following the accident, with doses estimated at 10 Gy. In later years an increase in ^{137}Cs accumulation in predatory fish (e.g. perch, pike and chub) in muscles ranged between 3 and 10 times higher than non-predatory fish (Kryshev *et al.*, 1993). A relationship between fish size (length or weight) and muscle concentrations of ^{137}Cs has been reported, with muscle concentrations for ^{137}Cs being correlated positively with size for perch and tench; and negatively correlated for carp (Jagoe *et al.*, 1998).

Radiation induced abnormalities in the gonads of silver carp located within the Chernobyl cooling pond were also observed in subsequent generations (Table A1.9) (Belova *et al.*, 1993).

Table A1.7 Temporal dynamics of the conditions for injured pine stands (from Arkhipov et al., 1994)

Stand injury	*Year*					
(dose, Gy)	1986	1987	1988	1989	1990	1991
No injury (<0.1)	Normal growth	Normal	Normal	Normal	Normal	Normal
Low (0.1-1.0)	Depression of growth	Occasional changes in morphology	Normal	Normal	Normal	Normal
Medium (1-10)	Strong growth depression, modified morphology, occasional death of trees	Partial forest restoration, modified morphology, absence of flowering	Rehabilitation of timber output, modified morphology	Normal	Normal	Normal
High (10-60)	Interrupted timber output, browning of needles, death of individual stands	Rehabilitation of individual sites	Rehabilitation of timber output, modified foliar morphology	Growth of foliage Under-growth of grass	Growth of foliage	Under-growth of grass
Acute (>60)	Total forest destruction	Needles fall splintering of bark	Bark fall creation of foliage and undergrowth of grass	Collapsing of stems, creation of a new plant community	Collapsing of stems, creation of a new plant community	Creation of a new plant community

Table A1.8 Radioecological effects in water bodies exposed to radioactive contamination – following investigations of water bodies surrounding Southern Urals, and Chernobyl NPP (Kryshev and Sazykina, 1998)

Dose (mGy h^{-1})	*Effect*
8-125	Increased anomalies in the reproductive system and disturbances in the state of sexual cells to 47-98% and sterility of gonads
200-400	12 fold increase in development anomalies of immature larvae of pike
80-4200	Death of fry and carp
(1.25 – 83.3) $x10^3$	Mass death of fish
(12.5-33.3)$x10^6$	Total death of a lake ecosystem

Table A1.9 Effects of radiation on the gonads of caged silver carp fish surviving in the Chernobyl nuclear power plant cooling pond after the accident and in subsequent generations (Belova et al. (1993), in UNSCEAR (1996))

Year of sampling	*Number of fish analysed*				*Proportion of fish with abnormalities in generative cells (%)*		
	Females	Males	Sterile	Total	Females	Males	Total
1989	17	8	2	27	0	25	-
1990	11	6	3	20	55	33	47
1991	9	7	0	16	78	57	69
1992	3	4	0	7	33	100	71
Total	40	25	5	70	35	48	44

External radiation from water and bottom sediments is considered to be an important factor in the dose received by aquatic biota. In 1987 dose rates to mollusc gonads was 0.9 mGy h^{-1} (Sokolov *et al.*, 1993). External radiation arising from the Chernobyl accident exerted a strong impact on benthic animals such as molluscs and bivalves with disturbances in reproduction and deterioration of colonies within the exclusion zone reported (Sokolov *et al.*, 1993). By the early 1990s the mollusc population was beginning to recover (Sokolov *et al.*, 1993).

A1.4 Assessment of overall impact of the Chernobyl incident

On the whole, the effects of irradiation on pines were mostly manifested within 2-5 km of the reactor source. Ionising radiation has, to date, failed to exert a direct impact on the bulk of the terrestrial vertebrates (Sokolov *et al.*, 1993) and although deterioration of mollusc populations were observed in the cooling pond immediately after the accident, these populations are now recovering (Sokolov *et al.*, 1993). Fish populations in the cooling pond have continued to breed successfully and continued to grow.

Despite death of pine trees in the Red Forest (located within the 10km exclusion zone) it has been suggested that the sum of the effect on flora and fauna in the highly reactive zone favoured biodiversity and individual abundance. Moose, roe deer, Russian wild boar, river otter and rabbits are all present within the 10 km exclusion zone; in contrast only rabbits were observed outside the 30 km exclusion zone (Baker and Chesser, 2000). Rare species such as wolves and the endangered black stork are all more abundant within the 30 km exclusion zone (Baker and Chesser, 2000), and rodent trapping is more successful within the 10 km exclusion zone than in uncontaminated areas (Baker *et al.*, 1996). Plant diversity within the most highly contaminated regions is similar to that in protected areas outside the zone (Baker and Chesser, 2000).

Benefits of excluding humans from the highly contaminated region appear to outweigh the radiation impact exerted on forests located near to the nuclear plant. Baker and Chesser (2000) concluded that detailed long-term studies on genetic load, population genetics, mutation rate, life expectancy, fertility, fitness and the development of radioresistance are needed to understand how populations exposed to chronic irradiation differ from unexposed populations. Such studies will also establish whether such dramatic increases in observed mutation rates can continue to be sustained by the populations in the long-term.

Kennedy *et al.* (1990) concluded that there was no evidence to indicate that Chernobyl derived fallout has had any negative impacts on ecosystems and wildlife within the UK. No more recent studies have attempted to relate effects of the impact of the Chernobyl accident within the UK's wildlife.

Internationally, studies have been carried out in Sweden and Belarus. Sweden was one of the countries most affected by the Chernobyl accident and ^{137}Cs levels in mammals correlated with

deposition levels. For example, two years after the accident maximum ground deposition of ^{137}Cs ranged from 22-145 kBqkg^{-1} whilst that at uncontaminated sites was 1.8 kBqkg^{-1}. ^{137}Cs in voles and shrews from the contaminated site were 7,800 Bqkg^{-1} and 6,300 Bqkg^{-1} respectively, compared with 40 Bqkg^{-1} and 50 Bqkg^{-1} respectively at the uncontaminated site (Mascanzoni *et al.*, 1990).

^{137}Cs in reindeer from the most contaminated areas of Sweden were around 100,000 Bqkg^{-1} in the first winter after the Chernobyl fallout. In 1996 the highest concentration level in reindeer was 24,000 Bqkg^{-1} (Ahman, 1996).

Chronic irradiation increased the occurrence of micronucleated polychromatic erythrocytes in bone marrow (cytogenetic damage) of voles collected from sites contaminated with ^{134}Cs and ^{137}Cs. Concentrations ranged from 1.8 kBqm^{-2} (approximate dose rate of 0.18 μGy h^{-1}) at a control site with negligible fallout to 22, 90 and 145 kBqm^{-2} (approximate absorbed dose rates of 0.37, 1.11 and 1.64 μGy h^{-1} respectively) at contaminated sites. The frequency of micronuclei per 1000 cells positively correlated with mean radiation dose received, ranging from 1.3±0.3 at the control site to 2.6±0.2 at the most contaminated site (Mascanzoni *et al.*, 1990). Although an increase in genetic damage has been observed as yet the ecological significance has not been evaluated.

In Belarus, the radiation exposure of 12-18 generations of bank voles collected from sites with ^{137}Cs deposition of 90 and 1,500 kBqm^{-2} resulted in high levels of chromosome aberrations, approximately 3-5 fold higher than measured in pre-Chernobyl frequency (Goncharova and Ryabokon 1995). The frequency of chromosome aberrations at both sites continued to increase from 1986-1991. Although the γ radiation load in bank voles significantly decreased over the 5 years post Chernobyl, no adaptation to the mutagenic effect of low level radiation has been observed. Bank vole γ activity concentrations estimated to be less than 1,000 Bqkg^{-1} also failed to significantly modify the frequency of chromosome aberrations (Goncharova and Ryabokon 1995).

Studies of frogs receiving estimated β dose rates of 7.1 μGy h^{-1} had a rate of chromosome aberrations in red bone marrow 2-10 times higher in 1986-1989 implying radiation induced genetic damage (Eliseyev *et al.*, 1990).

A1.5 Future research

The Chernobyl incident emphasised the lack of knowledge on the behaviour of radionuclides within ecosystems as a whole. It focussed attention on the effects of environmental releases of radionuclides following catastrophic events or through the routine operations of nuclear establishments. Further work recommended includes:

- Fine scale mapping of radionuclides and radiation doses in the exclusion zone surrounding the Chernobyl nuclear reactor. Measurements should be collected from soils, animals, and plants. This will permit a greater understanding of how wildlife interacts with a spatially varying contaminated environment. As such this will improve our knowledge of how wildlife is apparently thriving in what is considered to be a hazardous area.
- Investigations into the ecological half lives (EHL) of some commonly occurring radionuclides in the Chernobyl zone are also required. The EHL can be defined as the rate at which the contamination levels decline by 50% in a biological species at a contaminated site. It includes the effects of physical half life as well as ecological processes that alter the availability of contaminants at particular sites. Such studies will highlight the capacity of ecosystems to retain radionuclides following small and large-scale radionuclide release.
- Investigations into the loss of internally deposited radionuclides within the bodies of mammals from the Chernobyl exclusion zone. Knowledge of those rates will allow the determination of intake and turnover rates of radionuclides in the food chain.
- Work is needed to investigate the transfer of radionuclides from mother to progeny in the Chernobyl region to enable assessments of radiation doses received by embryos.
- Work to assess changes in morphology and mutation rates of plants and animals is required. Long-term changes in the asymmetry of plants and animals and in the distribution of radioactive

contaminants should be assessed. Collection of data on the relative fitness, reproductive success, longevity and movement of species trapped in areas of high radioactive backgrounds will enable the construction of dose response curves for genetic and morphological variables. The use of these variables as indicators of radioactive stress in wildlife populations could then be evaluated.

To further research into the impact of ionising radiation on wildlife resulting from the Chernobyl accident, the International Radioecology Laboratory (IRL) has been established. This has been a natural and logical consequence of the scientific programs in radioecology carried out, particularly by US Scientists, since the accident. The facility aims to foster international collaboration by acting as a co-ordination centre for information, research programmes and by providing the necessary logistical support in terms of equipment and labour. Scientists from various countries including the US and the UK are currently utilising the IRL with the aim of improving radioecological information including for example:

- risk assessments;
- modelling transfer and behaviour of radionuclides in the environment;
- evaluating decontamination methods;
- dose assessment and dose response measurements;
- determining the chronic effects of ionising radiation on wildlife.

A1.6 Summary

A large number of field studies were conducted following the Chernobyl accident particularly investigating the effects on terrestrial plants and mammals. However, the lack of dosimetry makes it difficult to compare field investigations with laboratory results. The area suffering greatest impact was the 30km exclusion zone, which received doses of 80-100 Gy within the first few weeks of the accident.

The following key points on the effects of ionising radiation on wildlife can be summarised from the studies conducted within the Chernobyl exclusion zone:

- No evidence of increased vertebrate mortality immediately following the accident;
- No effect on growth or development of voles, however radiation induced increases in organ size reflecting physiological and immunological pressure were observed;
- Genetic diversity increased in mammals trapped in 1997 that were exposed to 3.6 mGy h^{-1};
- Increased genetic mutations associated with loss of fitness in birds observed close to the Chernobyl site.
- Reduced species diversity of invertebrates within 3 km of the reactor following the accident, particularly litter fauna but populations recovered within 2-3 years as a result of migration from other zones.
- External radiation exerted a strong impact on benthic animals, such as molluscs and bivalves, within the freshwater ecosystems around the site. Disturbances in reproduction and deterioration of colonies within the exclusion zone were reported. However, by the early 1990s the mollusc population was beginning to recover.
- Temporary reductions in growth of trees were induced at doses of 0.1 Gy, whilst doses ranging from 1-10 Gy induced temporary modifications in morphology, 10-60 Gy induced death of individual tree stands and greater than 60 Gy induced total forest destruction.

Despite the mass death of pine forests and cytogenetic effects reported in mammals and birds in the immediate aftermath of the accident, it has been suggested that the sum effect of it on flora and fauna has been positive with observed increases in biodiversity and species abundance. A number of studies

are now reporting that many species of mammal thrive in the 10 km exclusion zone where they are absent outside it. Plant diversity within the exclusion zone is similar to that in protected areas outside the zone.

These studies indicate that the benefits of excluding man far outweigh the impact of ionising radiation. However, detailed long-term studies on genetic load, population genetics, mutation rate, life expectancy, fertility and radioresistance are required to evaluate the long-term ecological impact of the accident.

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A2 Biomarker Techniques

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A2.0 Introduction

In wild populations, natural selection operates to adapt the gene pool transmitted from generation to generation to the local environment. There is a dynamic equilibrium between the genetic variability and selection pressure that affects each generation. Exposure to environmental pollutants may exert selective pressure on the genetic structure of populations, through for example loss of fertility or increased mortality of individuals, and so modify the composition of the genetic pool (Dixon and Wilson, 2000; Woodhead, 1993).

Modifications to genetic material occur primarily by inflicting damage to DNA i.e. mutation (Dixon and Wilson, 2000). The effects of ionising radiation are dependent on both the quantities of the radiation as well as the biological tissues impacted. For example, irradiation of germ cells may produce both gene mutations and chromosome aberrations. Such mutations are produced by exposure to radiation from natural as well as artificial sources, but the amount and proportion of different types induced by irradiation may differ (UNSCEAR, 1977).

A biomarker is defined as a biological response to an environmental pollutant, which gives a measure of exposure. Responses may be at the molecular, cellular, biochemical or whole organism level. This report only focuses on the molecular impact in part due to the wide range of techniques in use and in part due to the fact that the types of biological damage caused by ionising radiation result initially in damage at the molecular level. Biomarkers may be used singly or in conjunction with others to quantify the biological impact of exposure to both radioactive and non-radioactive substances. They can provide a number of advantages in assessing the impact of pollution on wildlife (Copplestone *et al.*, 2000), such as:

- to provide an indication of the overall exposure to whatever environmental pollutants are present and from whichever uptake or transfer pathway;
- to account automatically for the bioavailability and transfer of pollutants;
- to account for any effects of exposure to mixtures of pollutants (e.g. synergistic or antagonistic effects); the effective contaminant(s) also does not need to be identified immediately;
- to assess biological information on the immediate and subsequent long-term impact; and
- to indicate whether any observed biological damage is due to the organism being exposed to levels of contamination which exceed its capacities for detoxification and repair.

It is unlikely that biomarker techniques will be used to assess compliance with any standards set for the protection of wildlife from exposure to environmental pollutants, as more research into the consequences, for the individual organisms and population, of the observed damage is required. They can, however, be used to provide useful information on the chronic exposure of wildlife to pollutants. They can also act as biodosimeters of exposure and act as an early warning that damage is occurring.

It has been recognised that genetic and cytogenetic damage can be sensitive indicators of radiation exposure (IAEA, 1976). There are two main problems when attempting to study the effects of ionising radiation on the genome of wildlife species:

- many species have karyotypes consisting of large numbers of small chromosomes, making it difficult to observe gross structural changes; and
- most tissues of mature organisms have low mitotic activity and slow cell cycling.

Despite this, biological techniques are being developed for use in detecting genetic alterations in wildlife species exposed to a range of environmental pollutants, including ionising radiation. These techniques are known as 'biomarkers' and a number of these techniques are described below.

Methods used for the detection of DNA damage can be divided into two categories based on target size:

- Cytogenetic - with effects visible at the level of the chromosome;
- Molecular or DNA scale - with effects operating at the level of the gene.

The distinction between cytogenetic and molecular scale effects is artificial and used only to group the techniques together.

A2.1 Cytogenetic effects - structural chromosomal aberrations

Structural chromosome aberrations have been used extensively as end points in mammalian studies. Consequences are often severe and exposure to many different environmental agents can induce chromosome aberrations due to chromosome breakage. Breaks may be rejoined, repaired to their original state, remain un-joined or be mis-repaired. Structural chromosomal aberrations are produced when gross changes occur. These are often lethal to the cell.

Structural chromosome changes can be detected using a number of tests:

- **Metaphase analysis**

  A range of structural effects can be identified using this analysis. The damage arises from loss or relocation of DNA material along the chromosome. If this occurs at points coding for the production of essential genes, the genes will not be expressed correctly within the cell, and the cell's function will be affected.

  This approach is laborious and requires a high level of expertise. Limitations to using this technique for wildlife include:

  - inadequate information on species' karyotype (or the karyotype is not suitable);
  - not being able to achieve the right conditions to induce cell division;
  - sample processing can artificially induce chromosome loss, which can affect results.

  Despite these difficulties, metaphase analysis has now been applied with some degree of success in marine invertebrates (Dixon and Flavell, 1986; Jha *et al.*, 1996; Martinez-Lage *et al.*, 1994).

  Metaphase analysis for chromosome aberrations is likely to be superseded as new molecular methods are introduced, e.g. FISH (Fluorescence *in situ* Hybridisation) explained below.

- **Sister Chromatid Exchange (SCE)**

  The Sister Chromatid Exchange test (SCEs) detects DNA breakage in both chromatids followed by an exchange of whole DNA duplexes. The induction of SCEs has been correlated with the induction of point mutations (Carrano *et al.*, 1978) and cytotoxicity (Natarajan *et al.*, 1983).

  There are practical and conceptual difficulties associated with the application of this technique to wildlife studies (Dixon and Wilson, 2000; Natarjan *et al.*, 1986), so it is best applied in addition to less equivocal methods of assessing DNA damage, such as metaphase analysis.

  A wide range of mutagenic and carcinogenic agents, which interfere directly or indirectly with DNA replication, can induce the SCE response. It is reportedly suitable for application to aquatic invertebrates and fish (Dixon & Clarke, 1982; Dixon and Prosser, 1986; Jha *et al.*, 1996; Jones and Harrison, 1987; Zakour *et al.*, 1984).

- **Numerical chromosome aberrations (aneuploidy)**

  Damage inflicted on microtubular apparatus results in a loss or gain in chromosomes during cell division (Parry and Parry, 1989). Aneuploidy, defined as chromosome deviations from the normal condition and number in a cell, is widely suggested as an important endpoint. It has been linked to a wide range of birth defects and certain types of cancer in mammalian cells (Tsutsui *et al.*, 1983).

  Direct counting of chromosomes is time consuming and laborious to perform (Dixon and Wilson, 2000). There is also a risk of artificially inducing variation if cells that are not intact are counted. For some applications, particularly with embryo or larval cells, it is possible to stain cells with a DNA specific stain before screening for chromosome aberrations by anaphase analysis (Anderson *et al.*, 1994; Dixon *et al.*, 1999). As with other cytogenetic techniques, aneuploidy can be induced by a number of agents.

- **Micronucleus test**

  Micronuclei are small membrane bound cellular particles containing DNA but separated from the cell nucleus. Two main types of mechanisms give rise to micronuclei: interference with chromosome segregation; and chromosome breakage. Radiation tends to induce the latter type of micronuclei as it causes DNA double strand breaks (Fenech, 2000).

  The technique is simple to perform, inexpensive and sensitive to a wide range of environmental pollutants. It also has the advantage of being applicable to a wide range of species without requiring a detailed knowledge of the species' karyotype.

  There are two main problems with the assay: it is laborious and time consuming; and it is difficult to distinguish between divided and non-divided cells which is required in order to use the technique quantitatively. Fenech and Morley in 1985 overcame this latter problem by using cell division inhibitors for mammalian studies. The use of centromeric and telomeric probes has also aided the interpretation of the different types of chromosome damage leading to micronucleus formation in mammalian studies (Russo *et al.*, 1996). It is still however difficult to culture cells of other wildlife species, limiting its application.

  The micronucleus test (MN) is widely established in the field of genetic toxicology. Evans *et al.* (1959) first described the micronucleus test for *in vitro* studies of radiation effects. The test was then developed in mammalian genetic toxicology (Heddle, 1973; Schmid, 1977). The micronucleus test has been applied to a wide range of mammal, plant and invertebrate species from both the terrestrial and aquatic environments (Hose and Puffer, 1983; Wrisberg and Vandergaag 1992; Burgeot and Galgani, 1995; Fenech and Neville, 1992; Adler, 1990; Parry *et al.*, 1988). A number of studies have demonstrated the potential for automation of the micronucleus test using flow cytometry or automated image analysis (Schreiber *et al.*, 1992; Hayashi *et al.*, 1992).

- **Fluorescence *in situ* hybridisation (FISH)**

  This technique allows specific nucleic acid sequences to be detected in metaphase spreads or interphase nuclei using chromosome specific paints. These paints are fluorescently labelled complementary DNA sequences which identify specific chromosomes and detect interchanges between labelled and non-labelled chromosomes.

  The technique was first developed independently by Pardue and Gall (1969) and John *et al.* (1969) using radiolabelled probes. Bauman *et al.* modified the method in 1980, by using fluorescently labelled probes. This has greatly increased its use in radiobiology and ecotoxicology due to improved efficiency, speed, clarity, increased safety and ability to use multiple labelled probes (Hofler, 1990).

  The technique is of value because it can detect stable structural chromosome aberrations, which are potentially of great biological significance. The aberrations may result in inappropriate gene expression, which is unlikely to be repaired and may be transmitted to future generations. Being stable, they also have the capability to reflect cumulative or historical damage (Pascoe *et al.*, 1995; Ellard *et al.*, 1995).

  FISH is a powerful technique for assessing and quantifying the impact of exposure to a wide range of pollutants with mutagenic properties, including ionising radiation (Pascoe *et al.*, 1995; Mitelman *et al.*, 1991; Parry and Parry, 1995).

  The application of FISH to the analysis of chromosome damage has opened up a wide field of research into the underlying mechanisms producing chromosome damage of all types. With suitable probes, FISH can be applied to nuclear material from both dividing and non-dividing cells making it applicable to wildlife studies. So many types of chromosome aberration have been detected using FISH that new methods of classifying the damage have been proposed (Savage and Simpson, 1994; Tucker *et al.*, 1995).

  FISH has two major limitations:

- the cost of the chromosome paints and the equipment to visualise them; and
- the fact that paints only exist for a limited number of species.

Chromosome paints are only available for humans, mice (Breneman *et al.*, 1993; Hoebee *et al.*, 1994) and for the slider turtle (Ulsh *et al.*, 2000). The cost of producing new chromosome paints is likely to decrease in the future as new *in situ* polymerase chain reaction kits become more widely available. This will widen the application of FISH for wildlife studies. FISH techniques have the potential to become biodosimeters with which to quantify chronic exposure to agents such as ionising radiation (Eastmond *et al*, 1995).

A2.2 Molecular approaches to the detection of DNA damage

New molecular tools for mutation testing have been developed in mammalian genotoxicity research over the past decade, and are now being extended to non-mammalian species.

It is now possible to detect individual DNA base changes allowing gene mutations to be identified directly. The development of automated DNA sequencers allows rapid screening for known and unknown mutations, although its application to wildlife studies has, to date, been limited.

Assays to quantify DNA strand breaks are more common. These DNA strand breaks are representative of genotoxic insult. Whilst DNA strand breaks do occur under natural conditions, exposure to genotoxins will increase their frequency. The most common technique in use is:

- **Single Cell Gel Electrophoresis Assay (SCGE) or Comet assay**

  The Comet assay was developed as a convenient and sensitive way to detect single or double strand breaks in individual cells (Ostling and Johnason, 1984; Singh *et al.*, 1988). The technique can be used on any isolated cell nuclei and does not require cell culturing (Fairbairn *et al.*, 1995). These features make the technique widely applicable in environmental studies, where any agent that causes DNA strand breaks can be assessed in many different wildlife species. In circumstances where there are multiple pollutants present, the technique will measure cumulative damage from all sources although the technique cannot distinguish the impact from the different causative agents.

  One advantage of the technique is that semi-automated image analysis techniques have been developed which permits rapid and objective determination of the levels of DNA damage. However, inter-laboratory comparisons are difficult due to variability in the application of the technique itself (i.e. different protocol steps), and the choice of image analysis parameters (Fairbairn *et al.*, 1995; Hellman *et al.*, 1995; Kent *et al.*, 1995).

  Despite these difficulties, the technique has wide application in radiation biology, assessment of DNA damage and crosslinks, oxidative damage, genetic toxicology, apoptosis and DNA repair mechanism studies (Fairbairn *et al.*, 1995). It is becoming a major tool for environmental biomonitoring because of its advantages in determining the impact from the interactions of complex mixtures of pollutants in the environment and the influence of environmental availability and pathways to wildlife (Tice, 1995a). The Comet assay is being used in particular as a tool to assess genotoxic damage in sentinel organisms in the environment (Tice, 1995b; Table A2.4).

There are a number of other assays being developed for environmental biomonitoring applications. For example, ^{32}P-postlabelling for the identification of DNA adducts tends to be specific to certain groups of pollutants, e.g. persistent organic compounds, but not relevant to ionising radiation (Jones and Parry, 1992; Gupta and Randerath, 1988; Phillips and Farmer, 1994). These will not be considered further in this report.

A2.3 Application of biomarker studies to quantify exposure to ionising radiation

Biomarker techniques have been applied to a wide range of potential chemical and ionising radiation sources. Examples include: X-rays; ^{60}Co; ^{137}Cs; á particles; UV; hydrogen peroxide; bleomycin; morphine; caffeine; acrylamide; benzene; smoking.

Most studies have been carried out in the laboratory, either on cultured cell lines or on tissue samples taken from laboratory animals exposed to a given chemical. Newer experiments have been carried out on animal samples collected from contaminated sites. Tables A2.1 to A2.4 provide examples of such studies for each of the major techniques described above.

Although emphasis has been on chromosome and molecular techniques in this Chapter, biomarker techniques can measure effects at different levels of biological organisation - molecular, cellular, whole organism. Biochemical, cytological and physiological biomarkers, for example, are available for use in environmental biomonitoring programmes (Nicholson, 1999; Walker, 1995; Peakall, 1992). These may be used in addition to, or instead of, the chromosome and molecular techniques. Ideally, biomarker techniques need:

- to be simple, sensitive and stable (and therefore widely available);
- to measure an impact which correlates with the level of exposure; and
- where possible, to provide an objective measurement to remove any bias from the results.

Biomarkers have the ability to measure the direct effects of environmental chemicals including any interactive effects from complex mixtures (Nicholson, 1999) but are unlikely at present to replace existing approaches to protect the environment because of their limited application. They do provide a measure of environmental health and can therefore provide an early warning of the environmental impact of chemicals on individuals (Peakall, 1992; Peakall and Shugart, 1993).

Table A2.1 Examples of recent chromosome aberration studies

Species	*Stressor*	*Description*	*Reference*
Humans	0.12 Gyd^{-1}	No significant difference in damage induced by low doses of chronic/acute radiation	*Zaichkina et al., 1997*
Rat	Cyclophosphamide	Increased chromosome aberrations in erythrocytes following exposure to chemical stressors	*Krishna et al., 1991*
Plant	Ionising radiation	Demonstrated increased incidence of chromosome aberrations in seeds and meristems collected from areas with different contamination levels around Chernobyl and Kysthym	*Shevchenko et al., 1990*
Aquatic invertebrates	10 µGy/h	Increase in radiation induced chromosome aberrations in the vicinity of Chernobyl	*Tsytsugina, 1998*
Fish	^{137}Cs – 842 $kBqkg^{-1}$ ^{90}Sr – 1879 $kBqkg^{-1}$	Increased variation in DNA content observed at some sites with a history of genotoxic pollutants	*Lingenfelser et al., 1997*
Slider Turtles	Ionising radiation	Increased variation in DNA content in red blood cells of turtles inhabiting a nuclear power plant cooling reservoir	*Lamb et al., 1991*

Table A2.2 Examples of recent micronucleus test studies

Species	*Stressor*	*Description*	*Reference*
Humans	2.4 - 3.6 Gy	Increase in micronucleus frequencies observed in red blood cells of radiation exposed individuals	*Almassy et al., 1987*
"	Radon (0.6 Gy)	Modification of the proportion of cells within each phase of the cell cycle following radiation exposure	*Johnson et al., 1997*
"	1.2 Gy h^{-1}	No difference in the degree of damage induced in cultured human lymphocytes by low doses of chronic and acute radiation	*Zaichkina et al., 1997*
Mice	γ rays	Erythrocyte monitoring at chronic low dose rates	*Garriott and Grahn, 1982*
"	γ rays 20 μGy h^{-1} for 26 days	Increased frequency of micronucleated erythrocytes following exposure	*Grawe et al., 1993*
"	Dichloroacetic acid	Micronuclei used to detect dose related increases in chromosome damage of erythrocytes	*Fuscoe et al., 1996*
"	Misonidazole	Increased frequency of micronuclei and chromosomal aberration in bone marrow following exposure	*Bisht and Devi, 1994*
"	Methyl methanesulfonate Ethyl methane sulphonate Mitomycin C	Dose response relationship established with chemical stressor induced chromosomal damage in bone marrow cells	*Matter and Grauwiler, 1974*
Small mammal	^{90}Sr and γ rays	Genetic damage detected	*Cristaldi et al., 1985*
"	Chernobyl derived ^{137}Cs	Positive correlation between genetic damage and radiation exposure	*Mascanzoni et al., 1990*
Rat	Radon	DNA damage following radiation exposure	*Johnson et al., 1997*
"	Cyclophosphamide	Increased micronuclei in erythrocytes following exposure to chemical stressor	*Krishna et al., 1991*
"	Natural radiation	Evidence of changes in dental and skeletal measurements to backup evidence of genetic effects	*Parida et al., 1987*
Fungi	X ray (0.5, 1,2 Gy) Colchicine Chloral hydrate	Micronuclei used to detect induced chromosomal damage	*Degrassi and Tanzarella, 1988*
Aquatic invertebrates	Hg Cd	Increased DNA damage, single strand breaks following exposure to heavy metals.	*Bolognesi et al., 1999*
Catfish	Chernobyl cooling ponds: ^{137}Cs	Genetic damage primarily in the form of DNA strand breaks	*Sugg et al., 1996*

Table A2.3 Examples of recent fluorescence in situ hybridisation studies

Species	*Stressor*	*Description*	*Reference*
Humans	0.56 Sv	Reciprocal translocations found in workers exposed to radiation within acceptable dose limits	*Lucas, 1997*
"	0.1-0.4 Gy	No increase in unstable chromosome translocations reported in populations living within the vicinity of Chernobyl compared with controls, however an increase in numerical chromosome aberrations observed	*Darroudi and Natarajan, 1996*
Mice	X ray. (2 Gy)	Increased frequency of chromosome translocations following exposure	*Hande et al., 1996*
"	X ray. (2 Gy)	Loss of chromosomes and increase in micronuclei in splenocytes	*Hande et al., 1997*
"	^{137}Cs (4 Gy)	Increase in both reciprocal and non-reciprocal translocations in bone marrow following exposure	*Spruill et al., 1996*
Slider Turtles	4-10 Gy	Biological damage induced by cumulative radiation exposure; dose response curve established	*Ulsh et al., 2000*

Table A2.4 Examples of recent comet assay studies

Species	*Stressor*	*Description*	*Reference*
Humans	Irradiation from radioiodine at 1Gy	Detection of DNA damage in human blood cells following irradiation	*Plappert et al., 1997*
"	X ray (<0.05 Gy)	Dose relationship between exposure and DNA damage	*Plappert et al., 1995*
"	γ rays (4 Gy min^{-1}) H_2O_2	DNA damage in lymphocytes. γ irradiation induced damage was more homogeneous than that from H_2O_2	*Visvardis et al., 1997*
"	X ray (8 and 35 Gy)	Induction of single strand and double strand breaks in peripheral white blood cells	*Banath et al., 1998*
"	Alkylating agent, intercalating agent and oxidative stress	Detection of DNA damage induced by chemical stressors	*Henderson et al., 1998*
Dog	X ray (3.9 Gy)	Exposure of peripheral blood and bone marrow cells induced DNA damage	*Kreja et al., 1996*
Plants	Methyl methanesulfonate Ethyl methane sulphonate	Detection of DNA damage, DNA migration and leaf nuclei damage, induced by chemical stressors	*Cotelle and Ferard, 1998*
Worms	Mitomycin C	Dose response relationship between DNA damage and concentration of chemical stressor established.	*Cotelle and Ferard, 1998*
"	PAH	Increase in single and double strand breaks following chemical exposure.	*De Boeck et al., 1997*
Fish	Ethyl methane sulphonate Methyl methanesulfonate	Detection of DNA damage, single and double strand breaks induced by chemical stressors.	*Belpaeme et al., 1998* *Deventer, 1996*
"	Oxidative stress	Detection of DNA damage in erythrocytes.	*Villarini et al., 1998*
Aquatic invertebrates	Hydrogen peroxide, Ethyl nitrosourea (ENU), PAHs,	Detection of chemical induced DNA strand breaks in mussels and shellfish following chemical exposure	*Cotelle and Ferard, 1998*

A2.4 Advances in human radiobiology

A2.4.1 Alternative mutational mechanisms

Underlying the assessment of stochastic radiation risk to both human beings and non-human biota is the assumption that damage to DNA by energy deposition within the gene at the time of irradiation (UNSCEAR 2000). However, there are reports of radiation effects that have been induced indirectly, with effects being manifested in different genes or cells from those originally damaged. While the biological significance of these phenomena remains unclear, they must be considered.

- **Chromosomal Instability**

  Since the original observations by Kadhim *et al.* (1992), there have been reports describing the induction of structural damage to chromosomes in the daughter cells of haemopoietic stem cells exposed to á particles (Nagasawa and Little, 1999). Low LET radiation has also been reported to induce the same chromosomal instability but at much higher doses. Chromosomal instability has also been reported in the descendants of unirradiated cells, referred to as the bystander effect (Lorimore *et al.*, 1998). Most experiments on chromosomal instability have involved *in vitro* irradiation and culture; however Watson *et al.* (1996) reported that instability induced by *in vitro* irradiation persisted after the haemopoietic cells were transplanted into mice. Wright (1998) claimed that these observations could not be explained as the direct result of radiation induced DNA damage, and must be the result of an epigenetic mechanism, perhaps involving persistent oxy-radical activity.

  Despite the interest in á particle induced chromosomal instability as a potential novel mutational mechanism, doubt persists over whether it is transmissible *in vivo.* Bouffler *et al* (2001) failed to find evidence of *in vivo* transmissible instability after á irradiation *in vitro* or *in vivo*. The authors noted, however, that transplantation of unirradiated bone marrow cells into mice whose own bone marrow had been previously ablated, could increase the yield of chromosome aberrations.

- **Minisatellites and Microsatellites**

  Two types of hyper-variable sequences have been implicated in effects on human health, mini- and microsatellite. Both consist of short sequences of DNA which are repeated. The number of repeats varies between individuals leading to differences in the overall length of the satellites. The overall biological significance of both mini- and micro-satellites is unclear but there is some evidence of association with human health. Instability of micro-satellite regions has been linked with inaccurate DNA repair and susceptibility to certain types of cancer (Karran *et al.*, 1996) and an association has been claimed between micro-satellite mutations and spontaneous abortion (Spandidos *et al.*, 1998).

  The evidence for an effect of radiation on the mutation rate of mini- and micro-satellites in non-human species comes from both laboratory experiments and field observations. Working with different strains of male laboratory mice, Sadamoto et al. (1994) and Dubrova et al. (2000) have reported induction of mini-satellite mutations following both high and low LET irradiation. In both cases, the frequency of induction was too high to be explained by the direct induction of damage at the mutated locus. Dubrova proposed an alternative mechanism involving structural damage in other parts of the genome, or in sensor molecules, resulting in indirect induction of mutation at the mini-satellite loci.

  Micro-satellite mutations have been reported to be two to ten fold higher in barn swallows breeding close (25-50 km) to the Chernobyl power plant than in birds outside the contaminated zone (Ellergren *et al.*, 1997). The radiation dose was not, however, reported. Similarly, two genetically identical populations of wheat, one grown on contaminated soil near Chernobyl and the other 30 km distant, were examined for mutations at 13 micro-satellite loci (Kovalchuk *et al.*, 2000). The mutation rate for the exposed plants (6.6×10^{-3}) was about 6 times higher than that for the controls. According to classical risk estimates, the estimated dose to the exposed plants of about 0.3 Gy was too low to account for this difference, implying that the mutations were induced by an indirect mechanism.

A2.4.2 Implications for impact assessment

There remains significant doubt over the substance of claims for alternative mechanisms of mutation induction. Chromosomal instability has not definitively been shown to be transmissible *in vivo*. Both mini- and micro-satellite sequences have naturally high spontaneous mutation rates, and the indirect mechanism of radiation mutation remains theoretical.

There is a further consideration to be made when assessing the potential impact of alternative mechanisms of impact assessment. There is currently little or no human epidemiological evidence of a genetic effect of radiation. It follows that, even if alternative mechanisms of mutation induction operate, the overall effect on health is not significantly greater than that predicted by the direct mutation mechanism. A similar argument can be applied to non-human species. Until an association can be demonstrated between the putative alternative mechanisms and the fitness of either individuals or populations, greater weight should be given to evidence that directly address these endpoints.

The existence of radiation-induced cancer tumours in wildlife exposed in the environment has also not been evidenced. Tumour formation in wildlife has generally been associated with chemical toxicants. This may be partly because the exposed individuals may become less fit, be more prone to predation, and so do not survive long enough for the tumour to develop. It may also be speculated that certain wildlife species (e.g. invertebrates) do not have a molecular mechanism for the generation of tumours (Dixon and Wilson, 2000). If the latter proves to be correct, then great care must be used when extrapolating the somatic genetic consequences of pollution exposure from individuals to populations.

A2.5 Future developments

The consequence of genotoxic exposure of marine species requires attention in the future. Many marine species over-produce gametes, leading to a potentially vast number of offspring, e.g. *Mytilus edulis* which releases an average of eight million eggs each time it spawns (Bayne, 1976). The excess of gametes is produced to combat the pressures of predation. As a consequence, it is difficult to predict how an increase in the number of genetically defective offspring due to pollution exposure will impact on population fitness (Dixon, 1982). Also, only a small proportion of the individuals in a population may actually contribute to the reproductive output (Li and Hedgecock, 1998). These points mean that much work is required to determine the relevance of any observed genetic or cellular damage on, firstly, the health and survival of the individual affected and, secondly, the impact on the population as a whole.

Our understanding of the mechanisms of mutation will need to be advanced in order to answer these questions (including research described in Section A2.2.3). Only then will it be possible to use biomarker techniques to demonstrate that biological damage is occurring at a particular site, identify the causative agent from potentially complex mixtures, and predict the consequence of the impact on both the individual and population affected.

With molecular techniques developing rapidly, these approaches to identify genetic damage may take over from cytological methods for routine analysis and testing of mutagenesis. There is therefore a need to overcome problems in the interpretation of the results observed. Furthermore, the sensitivity of some of the techniques needs to be improved, and/or the biological materials sampled need to be collected in a non-invasive manner.

Biomonitoring assays frequently require the culling of individuals from the population of interest. This is not sustainable, for example, it is not possible to assess directly the impact of environmental contamination on rare or endangered species. In addition, biomonitoring is likely to cause more disruption and harm to a population suspected of being at risk from environmental contamination than from the contamination itself. Consequently there is a need to develop more sensitive assays that for example, only require a small volume of blood to be collected. Non-destructive sampling procedures are also more acceptable on ethical grounds (Fossi, 1998).

The molecular and cytogenetic techniques described may help to evaluate the RBE values for non-human species by determining the impact of exposure to different forms of ionising radiation. Furthermore, they may be used to compare the effects of both radioactive and non-radioactive pollutants. Progress in genome mapping of non-human species may open new studies of different genetic endpoints caused by exposure to environmental pollutants, including ionising radiation.

A2.6 Conclusions

A biomarker is a biological response to an environmental pollutant, which gives a measure of exposure. The biomarker response may be at a molecular, cellular or whole organism level, however those at the molecular level are the most extensively investigated.

Molecular damage occurs at the level of DNA structure, and cytogenetic damage at the level of the chromosome. Ionising radiation can induce both molecular and cytogenetic damage. Chromosome aberrations, such as changes in chromosome number and structure, are the most extensively investigated. Modifications in chromosome structure can be detected by a number of techniques, but differences in the size and number of chromosomes of different species complicates the use of these aberrations as a marker for genetic damage.

Many techniques for investigating chromosome aberrations are laborious and time consuming and it is difficult to distinguish between divided and non-divided cells (a critical factor for some assays).

Techniques, such as Fluorescence In Situ Hybridisation (FISH), are being suggested as potential biodosimeters, which could be used to assess the impact of ionising radiation as part of a regulatory system. The technique uses fluorescently labelled probes that allow the detection of specific nucleic acid sequences that form a chromosome. The major limitation to the use of FISH at present is the limited number of fluorescently labelled probes that exist for non-human species.

Over the last decade developments in molecular techniques have provided new methods for assaying DNA damage, such as single strand breaks, double strand breaks and individual base changes, leading to gene mutations. Assays for DNA strand breaks such as the Single Cell Gel Electrophoresis Assay (SCGE), known as the comet assay, are widely used.

Cytogenetic and molecular damage are not specific to ionising radiation, with many hazardous pollutants also reported to induce damage. Interactions within a mixture of pollutants will be detected in biomarker assays, highlighting their usefulness as a tool for environmental biomonitoring in situations where there is a complex mixture of pollutants. Furthermore, the automation of a number of techniques will simplify the process, provide greater objectivity and higher throughput. Few studies have investigated the relevance of the observed biological damage to the individual and still less to the population level using biomarker techniques. This is a major limitation of application of biomarker studies to wildlife exposed to environmental pollutants.

Developments in human radiobiology have shown that indirect effects of ionising radiation on different genes or cells, to those originally irradiated can be detected. For example, chromosome instability and the bystander effect. The relevance of these effects in humans is still under debate, but maybe relevant in the future to wildlife studies.

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A3 Wildlife Dose Assessment Spreadsheets: Operating Guide

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A3.1 Introduction

This operating guide provides instructions for the use of the spreadsheets released with this Environment Agency R&D Publication 128 (2001): "Impact Assessment of Ionising Radiation on Wildlife".

The guide outlines the steps to be taken to undertake an assessment, detailing the approach and operation of the spreadsheets. The spreadsheets and this guide should ideally be used in conjunction with the report described above as details such as the size and choice of organisms and the selection of the radionuclides contained within the assessment spreadsheets are given in the text of the report. Chapter 6 also details the algorithms used in the spreadsheets.

There are three spreadsheets, two for aquatic and one for terrestrial ecosystems. This guide describes the approach using, as an example, the aquatic ecosystem spreadsheets. The differences between the aquatic and terrestrial spreadsheets are also described in this guide (Section A3.2).

In order to use these spreadsheets you will require:

- A personal computer running the Windows 95 or later versions of the Windows operating system
- Microsoft Word 97 and Microsoft Excel 97 or later versions of these programmes, with Microsoft Visual Basic for Applications loaded (as per the standard installation of Microsoft Office 97 or later)

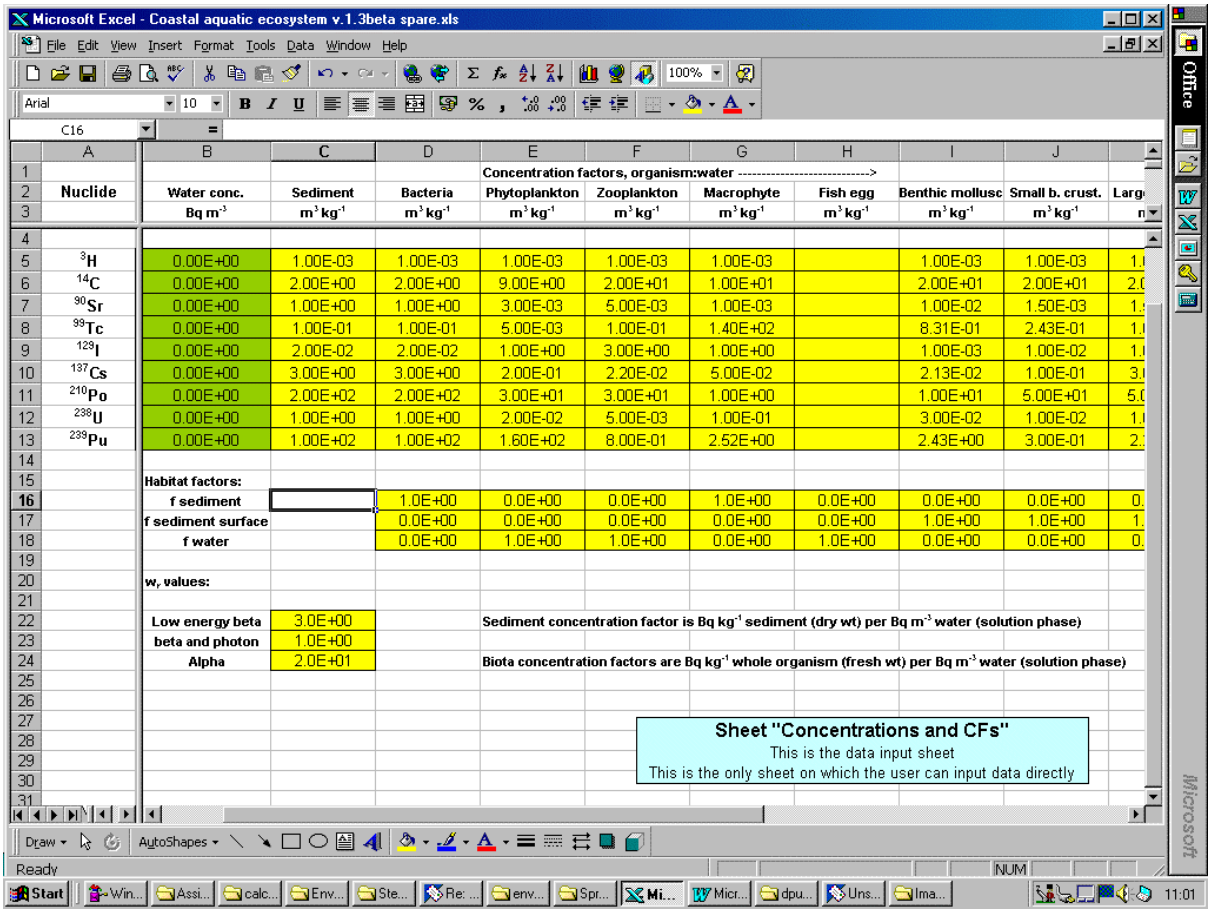
For optimum performance your system should have a processor at least equivalent in speed to a 100 MHz Pentium II and should have at least 64 Mb of RAM.

A3.1.1 Aquatic ecosystems

A3.1.1.1 Description of worksheets

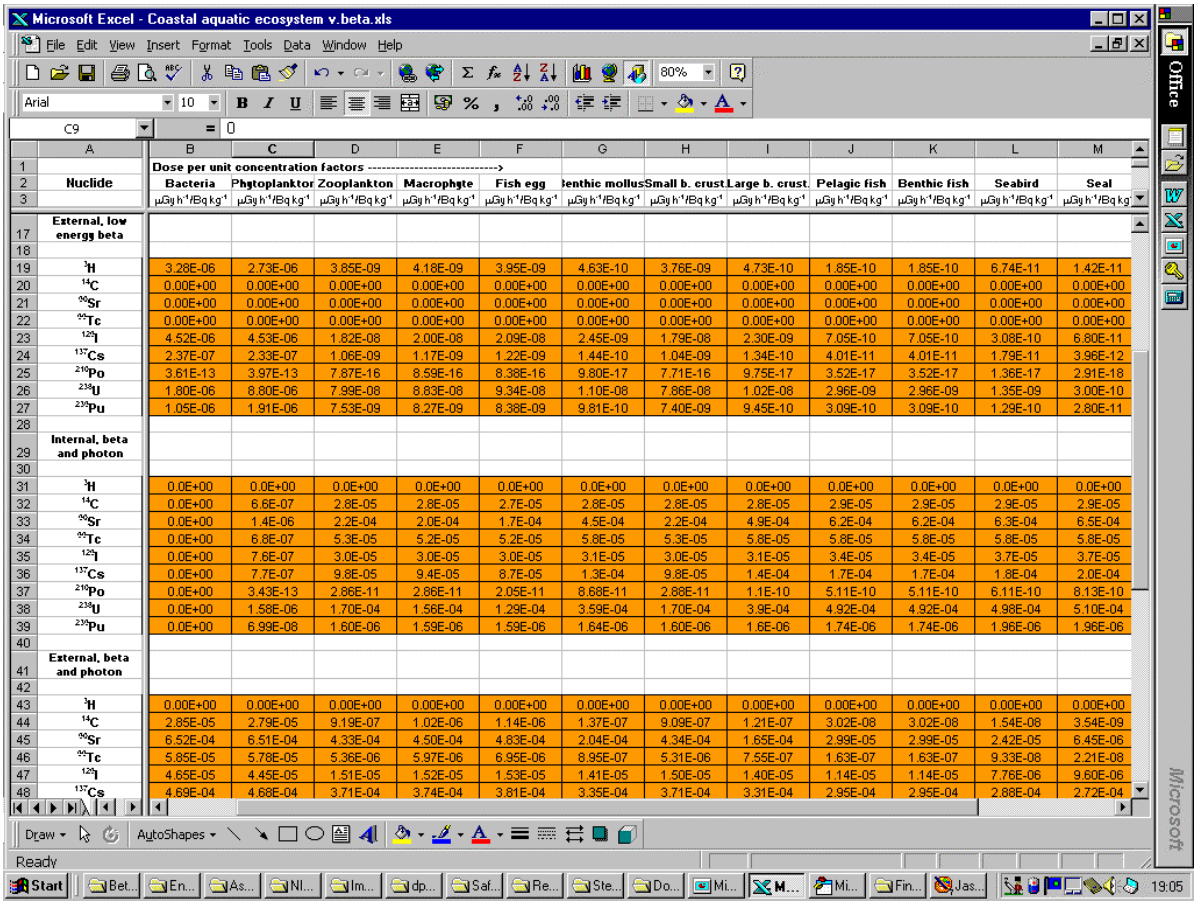
The workbooks for the aquatic environment contains the following worksheets:

"Concentrations and CFs"



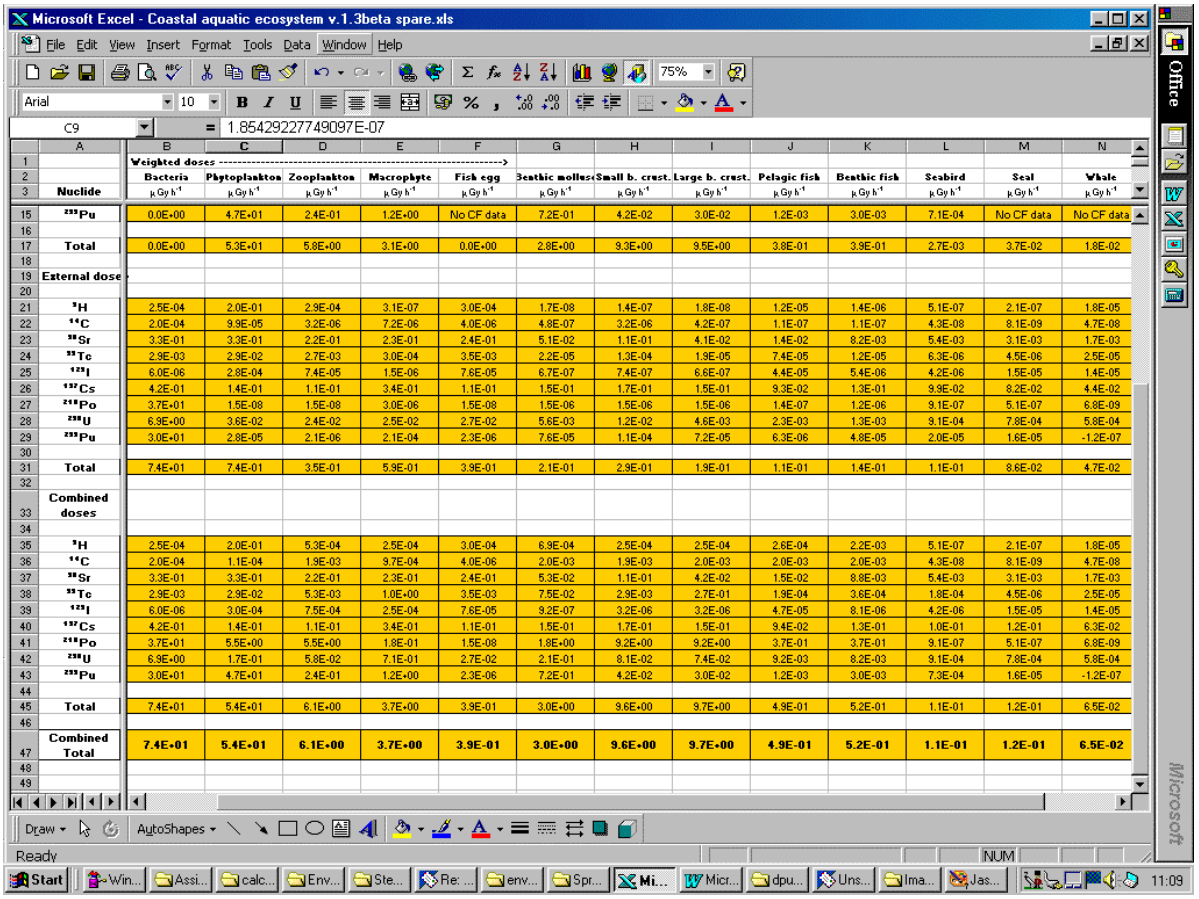
This worksheet contains the input data for the assessment. It is the only worksheet in which the user is allowed to enter data directly. This includes the habit factors which is the fraction of time an organism spends in different parts of the ecosystem. Concentration factors for the different radionuclides/organisms included in the assessment.

"DPUCs"



This worksheet contains the dose per unit concentration factors used in the dose calculations. This sheet cannot be modified by the user.

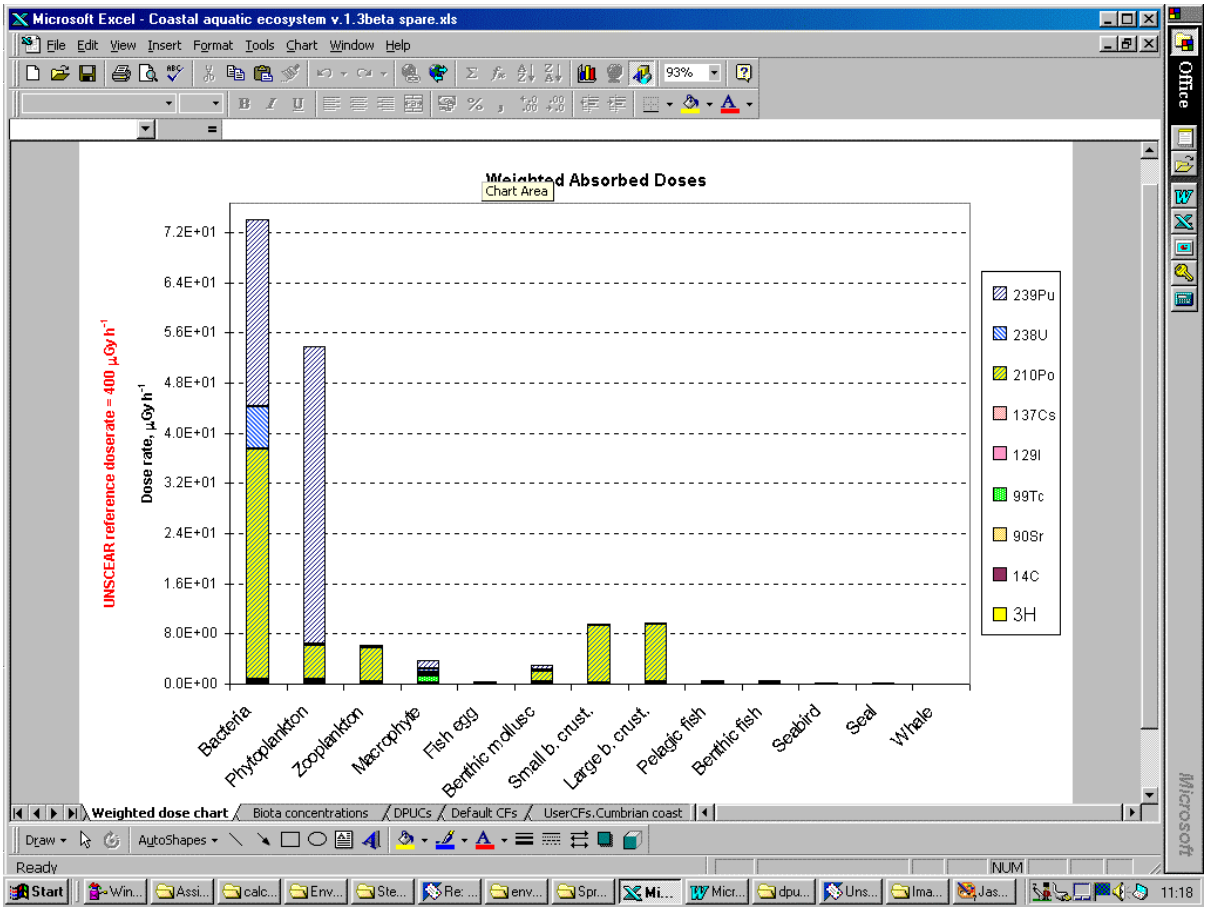
"Weighted doses" and "Unweighted doses"



This sheet contains the results of the dose calculations in terms of absorbed dose rates (μGy h^{-1}) weighted by the radiation weighting factors specified on sheet "Concentrations and CFs". Results are broken down by nuclide and by source of exposure (internal or external). The user cannot edit the sheet.

The similar sheet titled "Unweighted doses" contains the results of the dose calculation in terms of absorbed dose rates (μGy h^{-1}) with no weighting factors applied.

"Weighted dose chart" and "Unweighted dose chart"

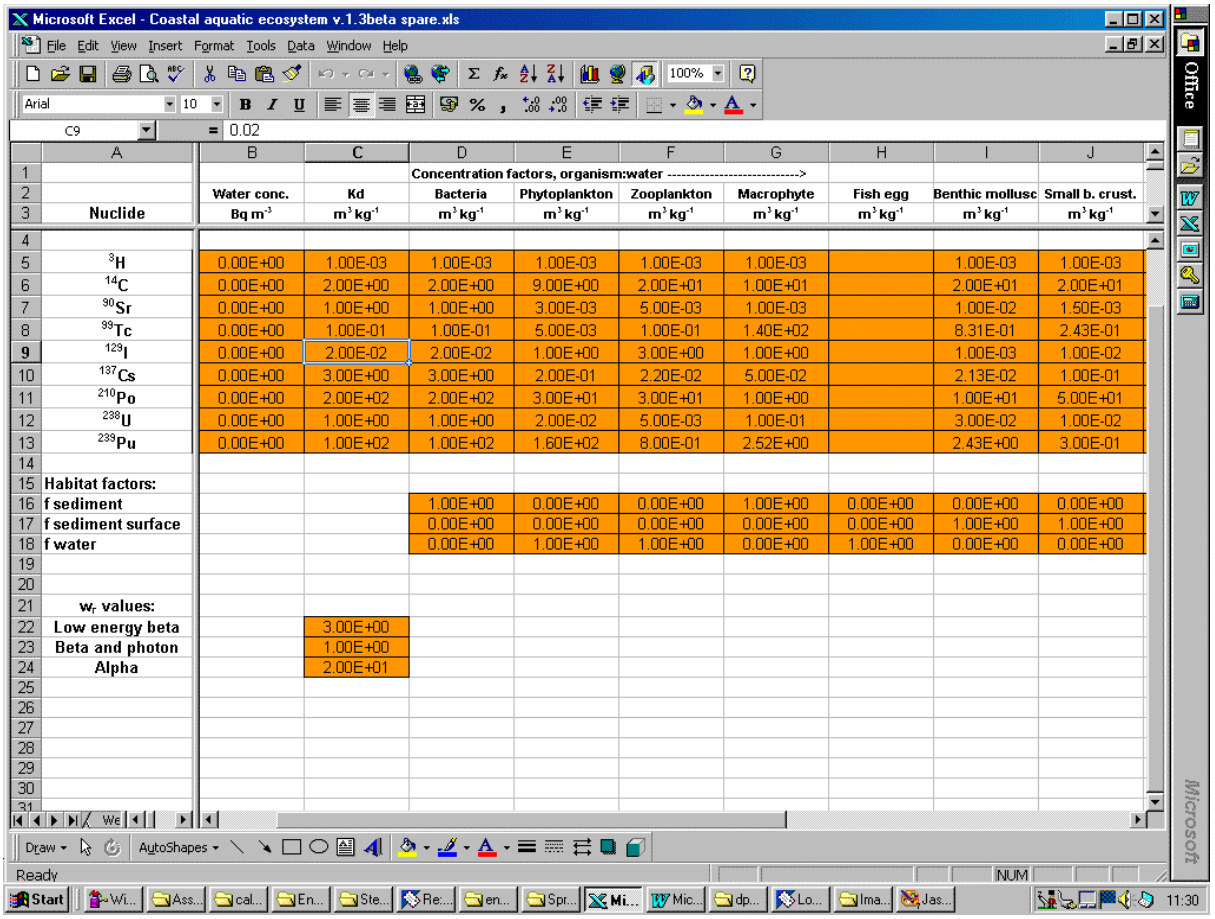


This sheet provides a graphical representation of the results for weighted absorbed dose. The similar sheet "Unweighted Dose Chart" provides a graphical representation of the results for unweighted absorbed dose.

It is recommended that dose charts are printed in colour as it is difficult to distinguish between 9 keys using just black and white. Charts can be printed in black and white but you should be advised that it may be difficult to distinguish between contributions of different radionuclides. If the graphs must be printed in black and white, you are advised to save the results of the dose calculations as a new worksheet and copy and past the graph into a new Excel workbook. The user can then edit the keys on the graph for the different radionuclides in accordance with their requirements by selecting their preferred shading options. The user can then print the graph as required.

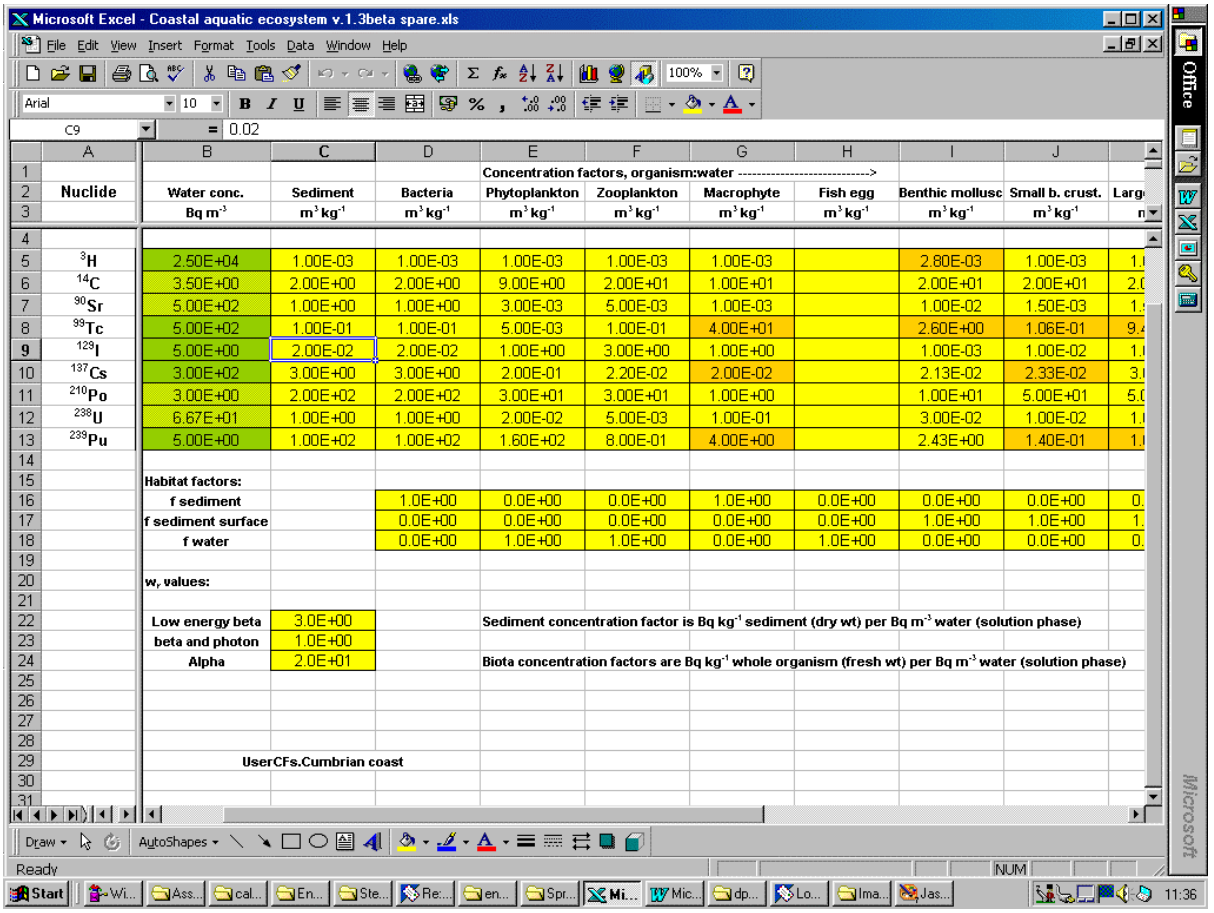
Refer to Section A3.1.2.3, 'reviewing dose results', for adjusting the scales of the charts.

"Default CFs"



This sheet contains the recommended default values for concentration factors, habit/location factors, and radiation weighting factors for use in the dose assessment. Values entered by the user during specific assessments may be recalled into the data input sheet "Concentrations and CFs" at any time (see below).

"UserCFs.xxxx"

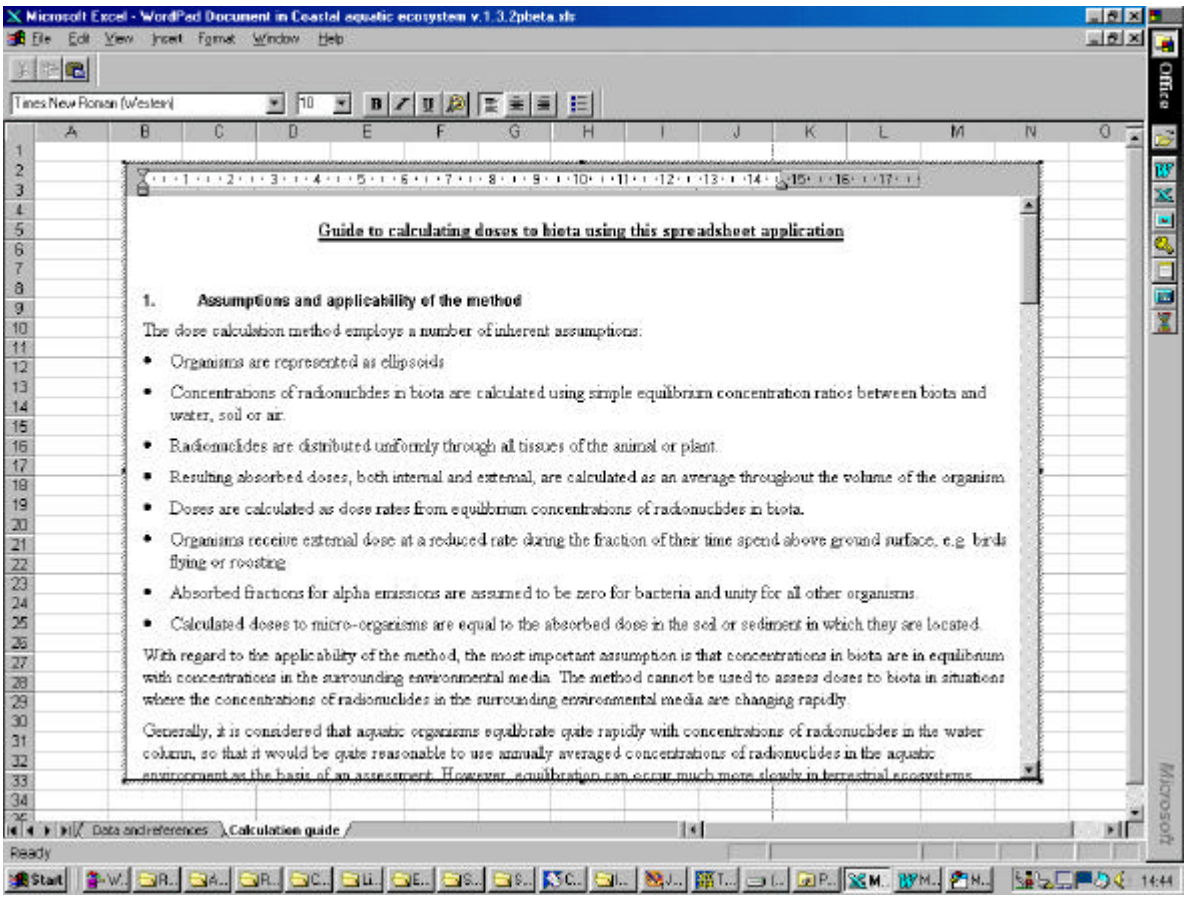


Sheets with this title have been saved by the user to preserve concentration factors and other data which have been developed for a specific site or scenario. Orange shaded cells indicate values for CF or other factors which have been changed from the default values, either by direct entry on the data input sheet or during the assessment sequence (see below). Light green shaded concentration cells denote values which have been derived from biota measurements during the assessment (see below).

Values from sheets of this type may be recalled into the data input sheet at any time, either *in toto* or selectively (see below).

"Calculation Guide" and "Data and references"

These worksheets contain embedded Wordpad documents. "Calculation Guide" contains a summary of the assumptions employed in the calculations, the limitations of the calculation method and a guide to carrying out an assessment. "Data and references" contains information on the reference dimensions of the organisms used for the calculations, and the principal references used to determine concentration factors.



Please refer to Section A3.1.2.4 for more information about accessing these worksheets.

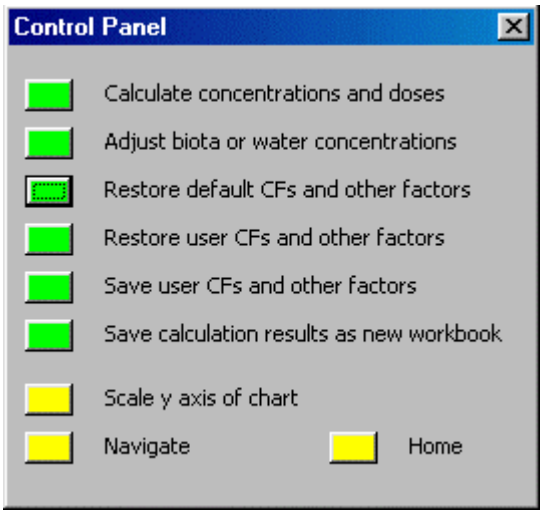
Worksheet and Workbook protection

All worksheets are protected to prevent inappropriate alteration to cell values, and insertion or deletion of rows and columns. The workbook itself is protected to prevent worksheets being inserted, moved or deleted. These things can only be done, when required to execute the calculations, under the control of the spreadsheet programme itself.

Please refer to Section A3.4 for more details of the protection provided.

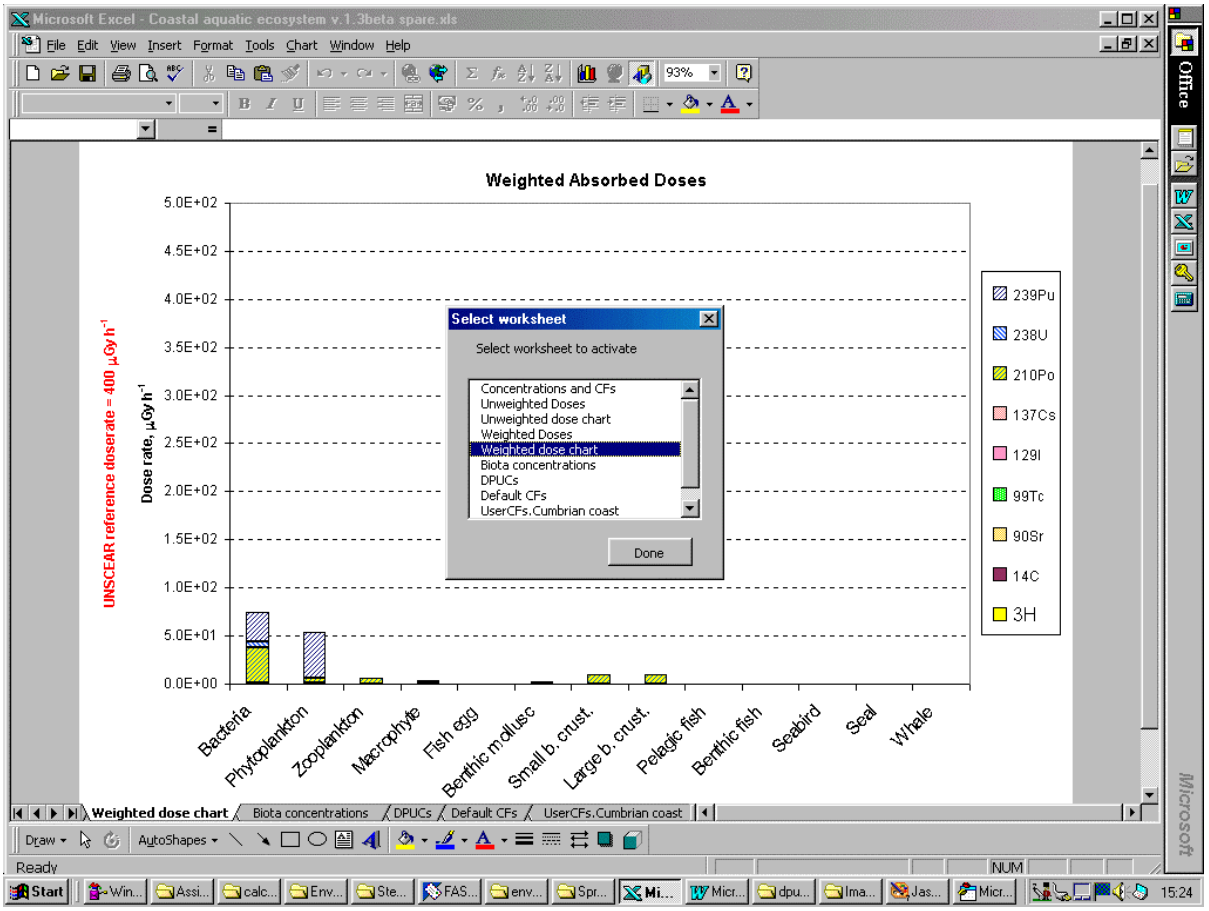
The Control Panel

The Control Panel is not a worksheet, but it is a vital part of the interface with the user. It is a means of controlling the calculations and other actions performed by the spreadsheet application. The Control Panel is called by pressing key 'F1' at any time when the assessment spreadsheet is open:



The functions of the control panel are explained in the step by step instructions for carrying out an assessment, which follow.

By default, worksheets are formatted with split rows and columns whenever they are activated; in this form worksheet tabs are not accessible. A simple utility accessed from the 'Navigate' button on the control panel makes it easy to move from one worksheet to another:



Clicking on a worksheet name in the list activates that worksheet; clicking 'Done' returns to the control panel, whereas clicking 'X' closes the navigation utility without returning to the control panel.

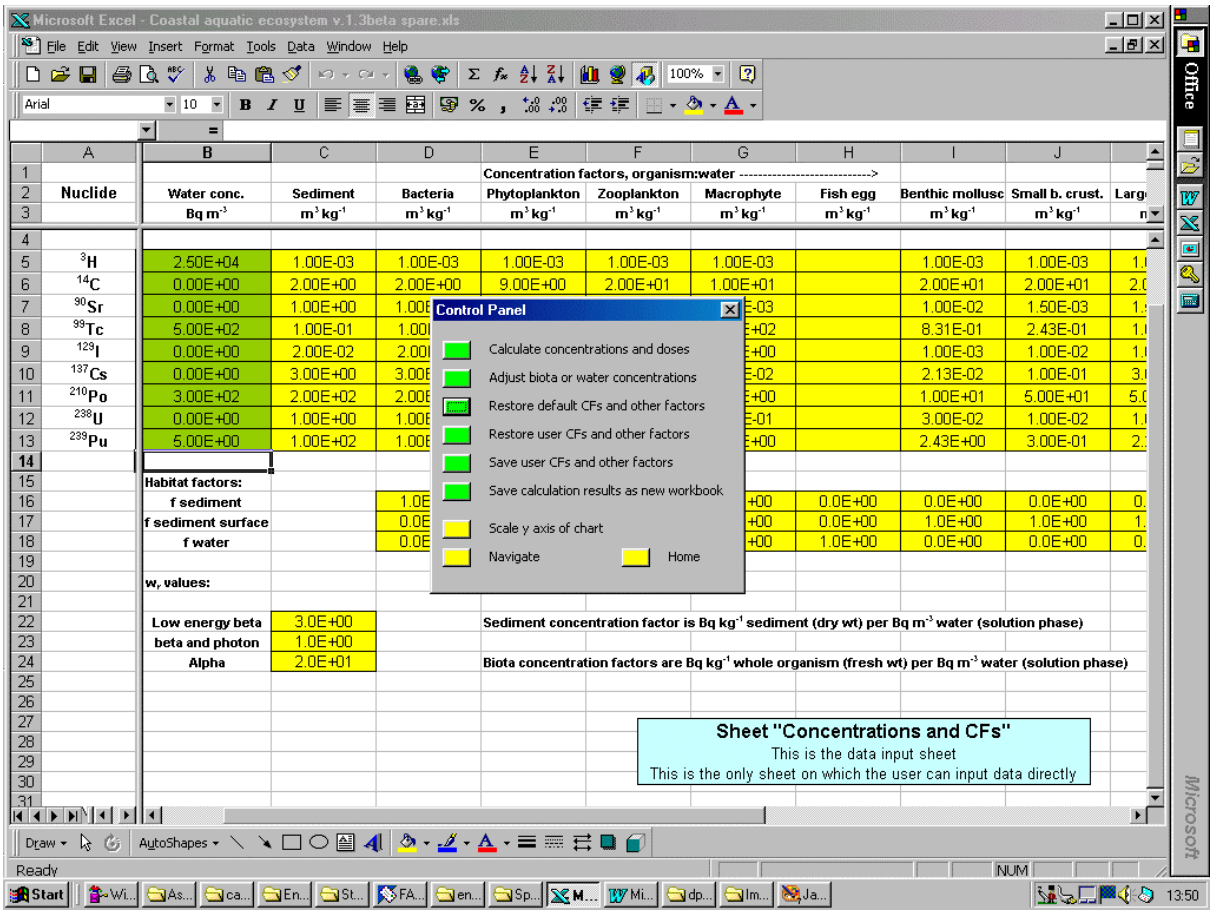
A3.1.2 Performing an assessment

A3.1.2.1 First estimate of concentrations in biota, and doses

On opening, the workbook will always present sheet "Concentrations and CFs". Set all the water concentrations in the green cells to the data you require to drive the assessment.

Note that the required input data are concentrations of each nuclide in the dissolved phase (i.e. filtrate).

Now you need to ensure all the CFs and other factors are set to their default values. Press key "F1" to bring up the control panel:



"Clicking" the button marked "Restore default CFs" will write the default values into all the yellow shaded cells.

If, at this stage, you have site specific CFs - or you wish to use non default values for, say, radiation weighting factors – they can be entered directly onto the data input sheet, after closing the control panel by clicking the "X" button in the top right corner.

Note that:

- **CFs are in the units Bq kg^{-1} fresh weight of organism per Bq m^{-3} solution phase (filtrate) in water.**
- **Habitat factors represent the proportion of time which the organism spends buried in sediment, on the surface of the sediment, and free swimming in the water column**
- **w_r values (radiation weighting factors) represent the relative biological effectiveness of the different radiation types, relative to X- or ã-rays, in producing endpoints of ecological significance**

Once the initial data entry is to your satisfaction recall the control panel (if necessary) and click on the button marked “Calculate concentrations and doses”. The concentrations of all radionuclides in sediment and all the reference organisms, and the consequent radiation dose rates, will then be calculated and written into the appropriate worksheets.

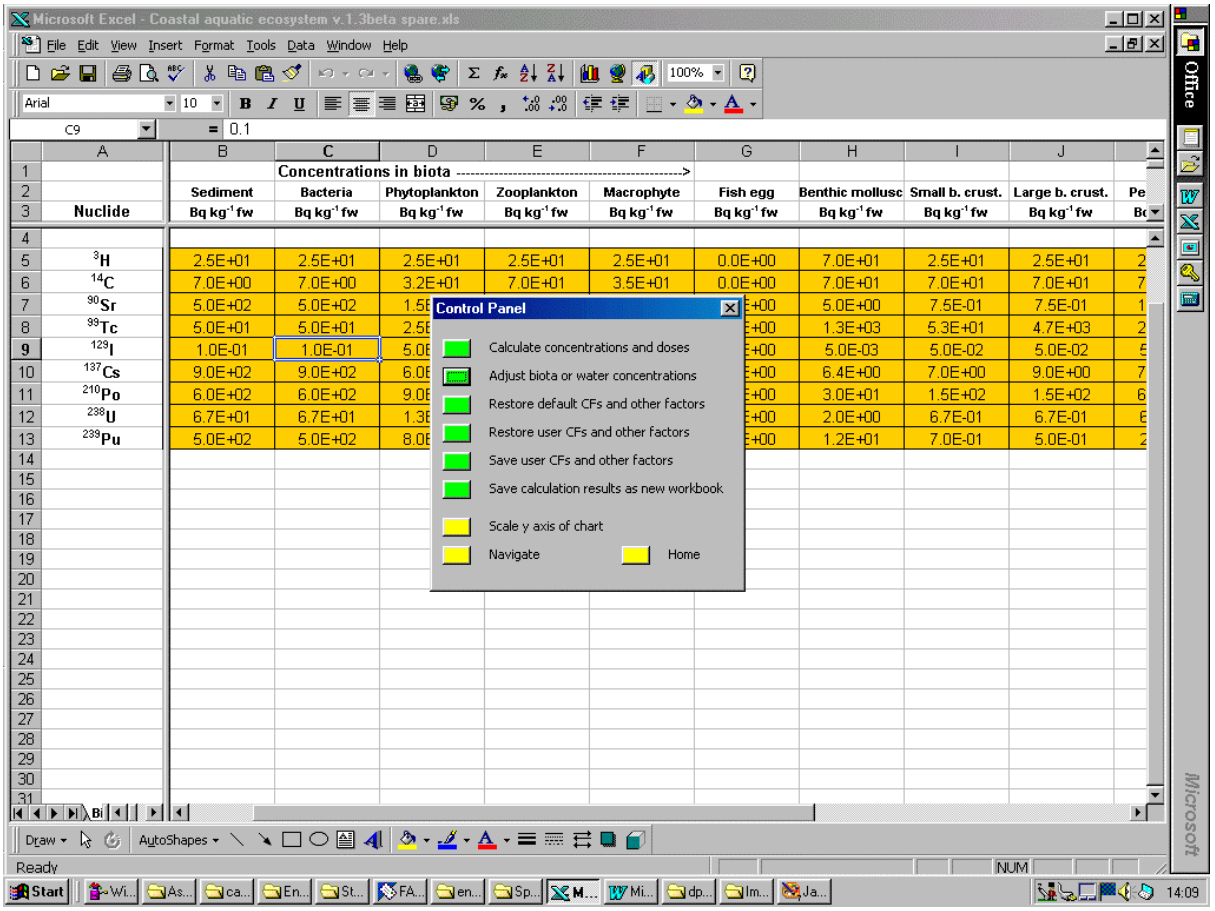
Note that until the ‘Calculate’ button is clicked, the values of cells in the concentration's worksheet and the doses worksheets will not change. Simply changing the values of cells in the “Concentrations and CFs” data entry sheet will not cause any calculations to take place.

A3.1.2.2 Using measured biota concentrations to improve the estimate

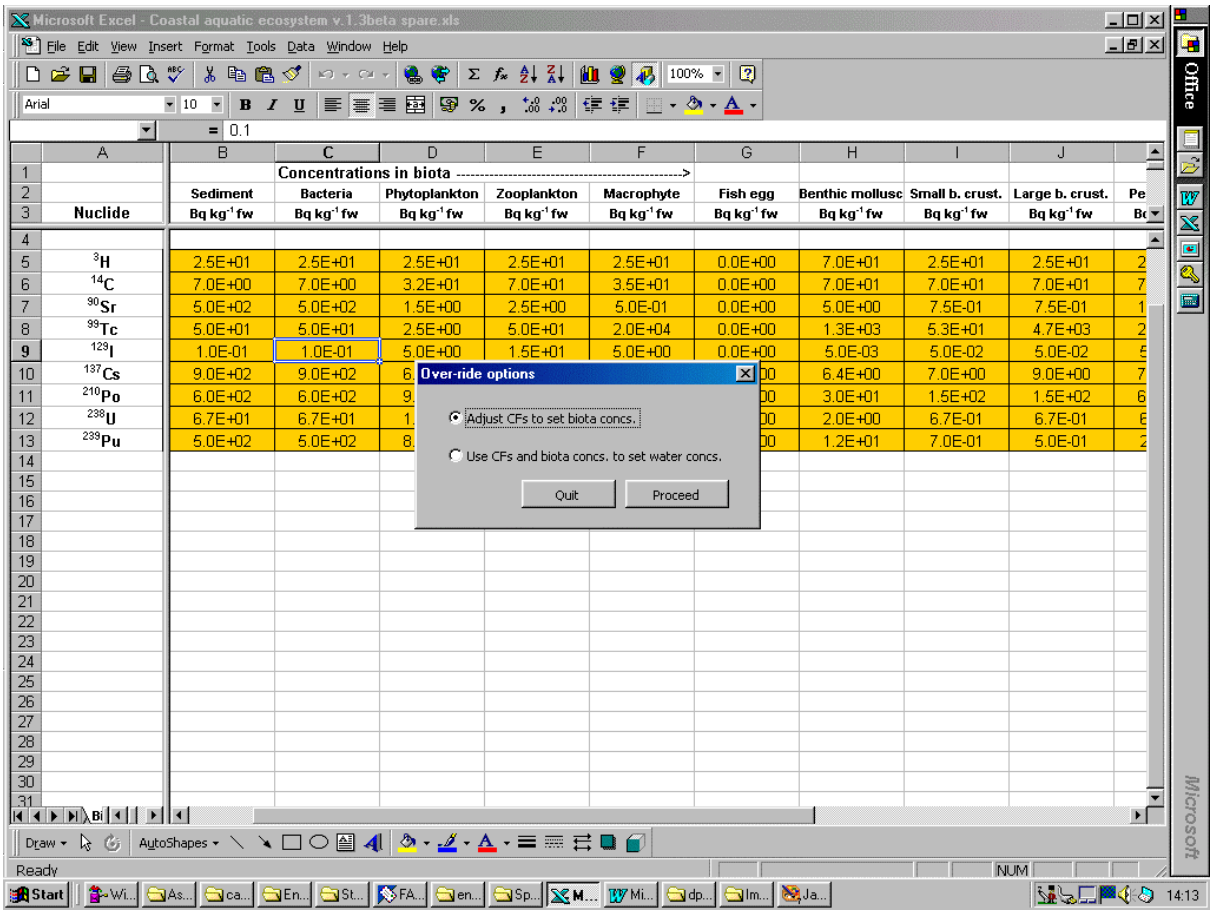
After the calculations are complete, the worksheet “Biota concentrations” will be shown on the screen. You could at this stage use the ‘Navigate’ button on the control panel (see below) to look at the dose result worksheets or charts.

Now is the logical time to bring any relevant environmental measurements into play.

If necessary, recall the control panel and click on the button marked “Adjust biota or water concentrations”:

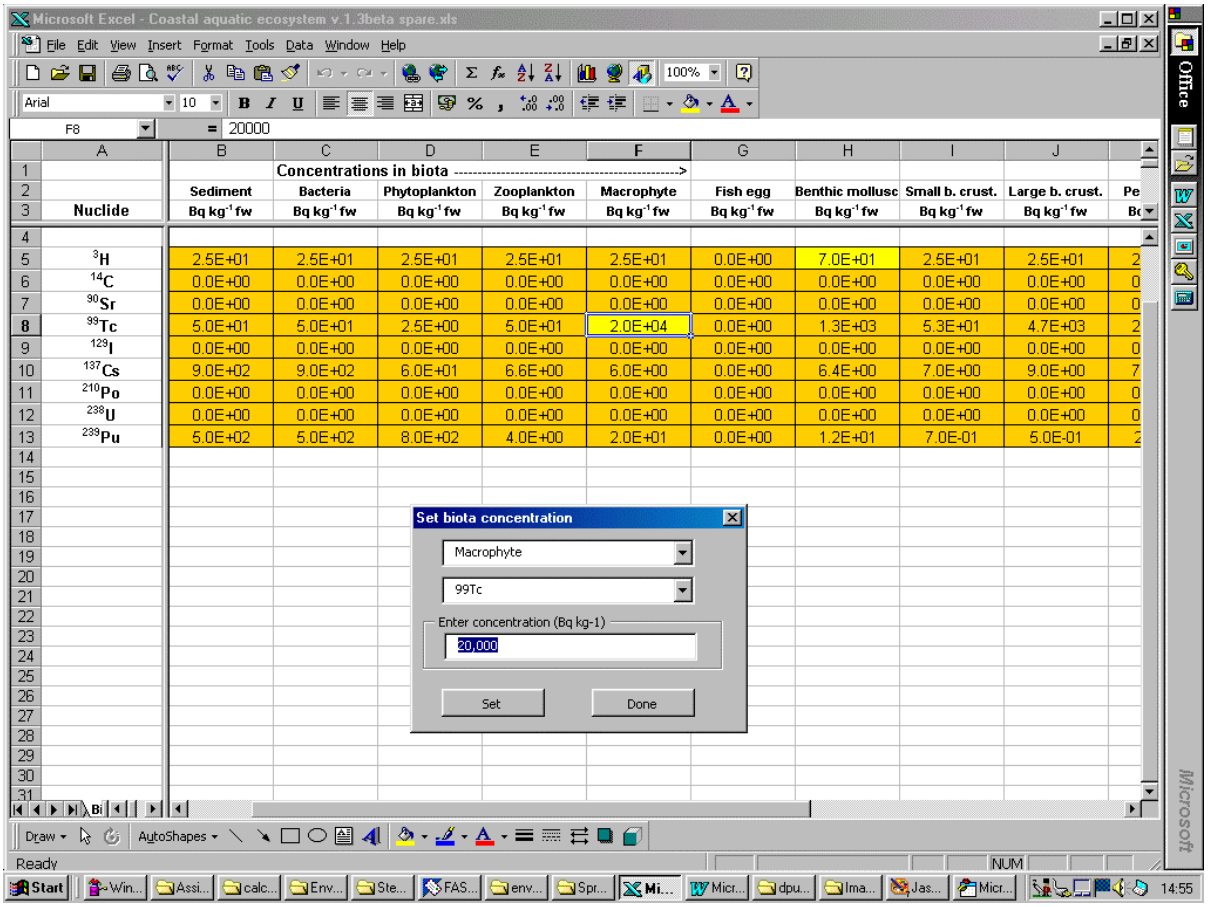


This will bring up a second dialog box with two options:



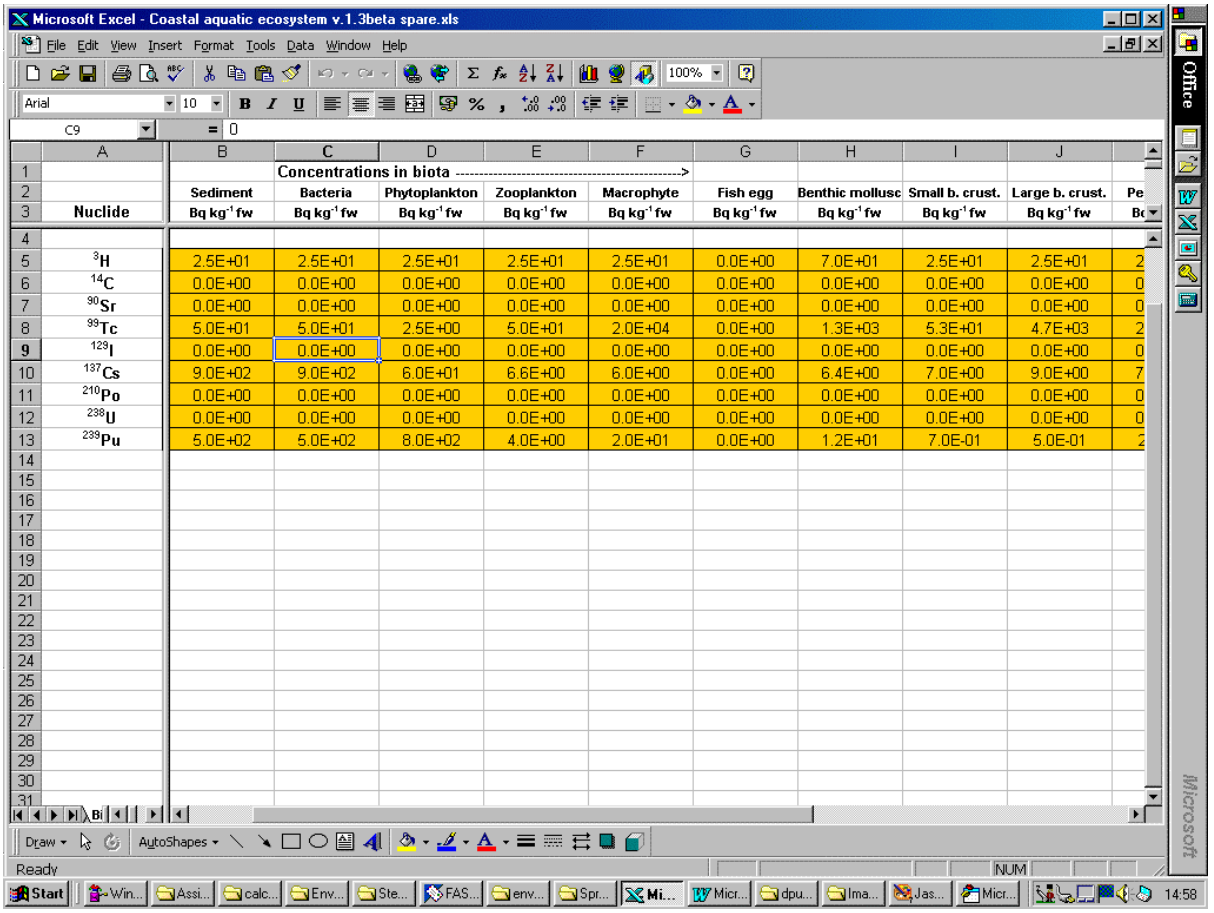
"Adjust CFs to set biota concentrations"

This option allows measured concentrations in biota to be entered, causing the spreadsheet to change the appropriate CF value so that this concentration is produced by the calculation. To do this, select that option in the dialog box and click on 'Proceed':



Select the nuclide and organism in the two upper drop-down boxes, type in the measured value in the bottom text box and click on “Set”. The new value will appear in the appropriate worksheet cell and the cell colour will change to yellow.

Once all necessary concentrations have been entered, click on ‘Done’. This will cause the appropriate CFs on the data entry sheet to be changed, and the concentrations and doses calculations to re-run. Once this has been done the new values will remain on the “Biota concentrations” worksheet but the colour will have reverted to orange, signifying that they have been calculated by the spreadsheet:



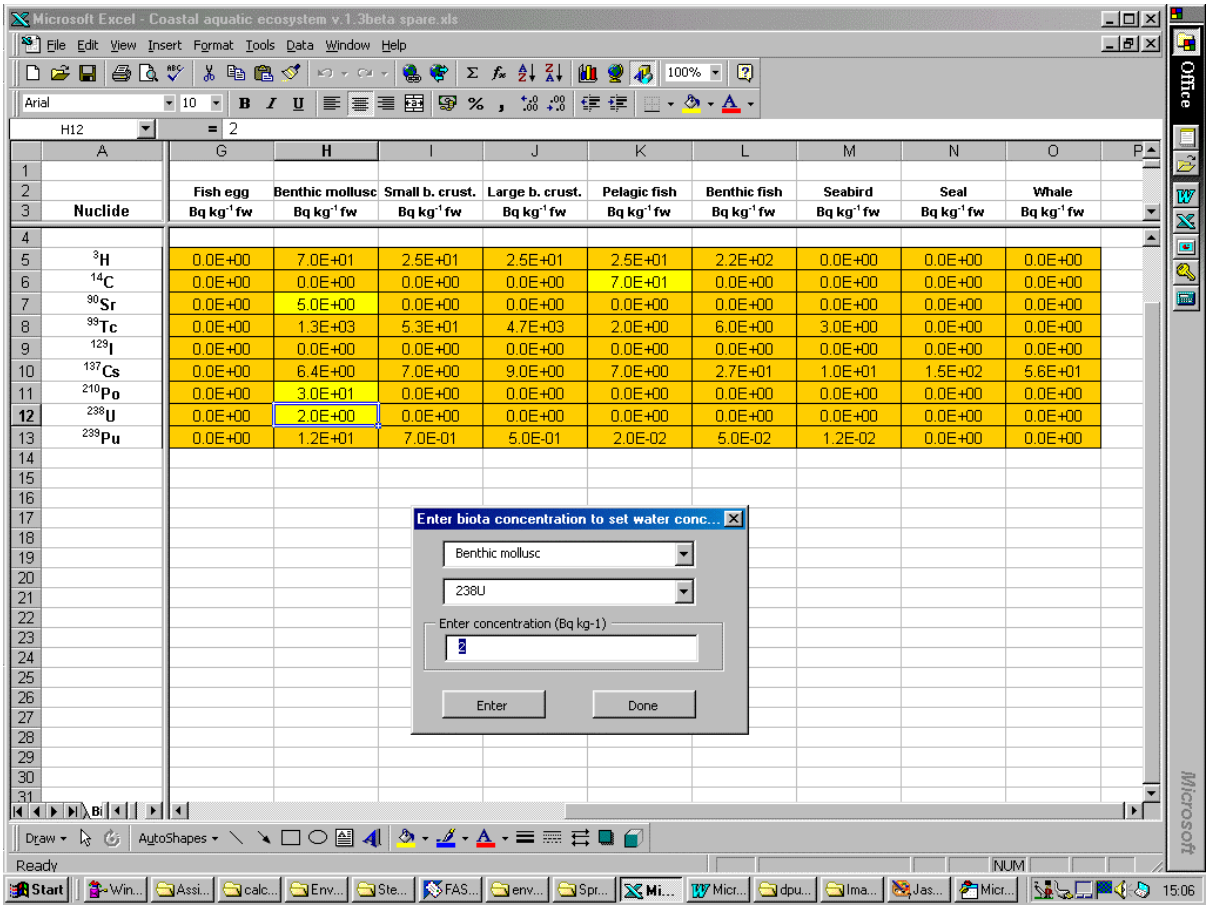
Microsoft Excel - Coastal aquatic ecosystem v.1.3beta spare.xls

	A	B	C	D	E	F	G	H	I	J
1			Concentrations in biota ---------->							
2		Sediment	Bacteria	Phytoplankton	Zooplankton	Macrophyte	Fish egg	Benthic mollusc	Small b. crust.	Large b. crust.
3	Nuclide	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw	Bq kg^{-1} fw
5	^{3}H	2.5E+01	2.5E+01	2.5E+01	2.5E+01	2.5E+01	0.0E+00	7.0E+01	2.5E+01	2.5E+01
6	^{14}C	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
7	^{90}Sr	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
8	^{99}Tc	5.0E+01	5.0E+01	2.5E+00	5.0E+01	2.0E+04	0.0E+00	1.3E+03	5.3E+01	4.7E+03
9	^{129}I	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
10	^{137}Cs	9.0E+02	9.0E+02	6.0E+01	6.6E+00	6.0E+00	0.0E+00	6.4E+00	7.0E+00	9.0E+00
11	^{210}Po	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
12	^{238}U	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
13	^{239}Pu	5.0E+02	5.0E+02	8.0E+02	4.0E+00	2.0E+01	0.0E+00	1.2E+01	7.0E-01	5.0E-01

In this example, we have a number of nuclides for which the water concentrations, and hence the concentrations in biota, are zero. For these nuclides we can use measured concentrations in biota to estimate a water concentration, and hence concentrations in other biota for that nuclide.

Recall the control panel, click on “Adjust biota or water concentrations” and then choose the second option.

"Use CFs and biota concs. to set water concs."



Nuclide	Fish egg Bq kg^{-1} fw	Benthic mollusc Bq kg^{-1} fw	Small b. crust. Bq kg^{-1} fw	Large b. crust. Bq kg^{-1} fw	Pelagic fish Bq kg^{-1} fw	Benthic fish Bq kg^{-1} fw	Seabird Bq kg^{-1} fw	Seal Bq kg^{-1} fw	Whale Bq kg^{-1} fw
^{3}H	0.0E+00	7.0E+01	2.5E+01	2.5E+01	2.5E+01	2.2E+02	0.0E+00	0.0E+00	0.0E+00
^{14}C	0.0E+00	0.0E+00	0.0E+00	0.0E+00	7.0E+01	0.0E+00	0.0E+00	0.0E+00	0.0E+00
^{90}Sr	0.0E+00	5.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
^{99}Tc	0.0E+00	1.3E+03	5.3E+01	4.7E+03	2.0E+00	6.0E+00	3.0E+00	0.0E+00	0.0E+00
^{129}I	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
^{137}Cs	0.0E+00	6.4E+00	7.0E+00	9.0E+00	7.0E+00	2.7E+01	1.0E+01	1.5E+02	5.6E+01
^{210}Po	0.0E+00	3.0E+01	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
^{238}U	0.0E+00	2.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
^{239}Pu	0.0E+00	1.2E+01	7.0E-01	5.0E-01	2.0E-02	5.0E-02	1.2E-02	0.0E+00	0.0E+00

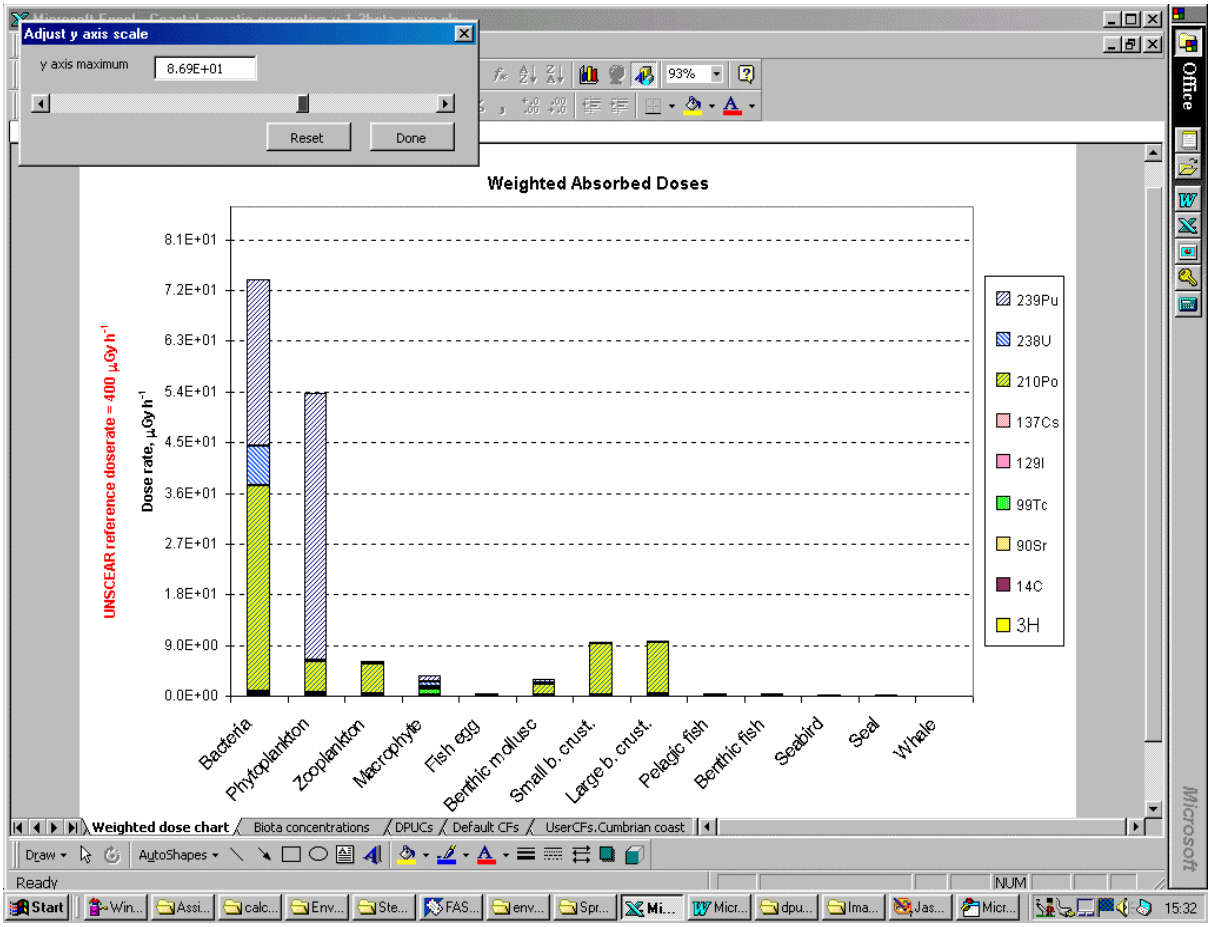
As before, nuclide and organism combinations can be selected and a measured biota concentration can be entered (click "Enter" button). When all are entered clicking on 'Done' runs the concentration and dose calculations yet again, setting water concentrations for the selected radionuclides as required by the set organism concentration and the current CF value on the data entry sheet.

Note that a concentration value for only one organism is required to determine the water concentration for a particular radionuclide. If you do enter concentrations for more than one organism for a single nuclide, it is the last entered value that will determine the water concentration.

A3.1.2.3 Reviewing dose results

Dose results may be reviewed either on the appropriate worksheet tables or, more visually, on one of the two charts provided. Press F1 and click the "Navigate" button to select either the weighted or unweighted dose chart to review the results of the assessment.

The scale of the chart can easily be adjusted with a utility accessed from the “Scale y axis of chart” button on the control panel (F1):



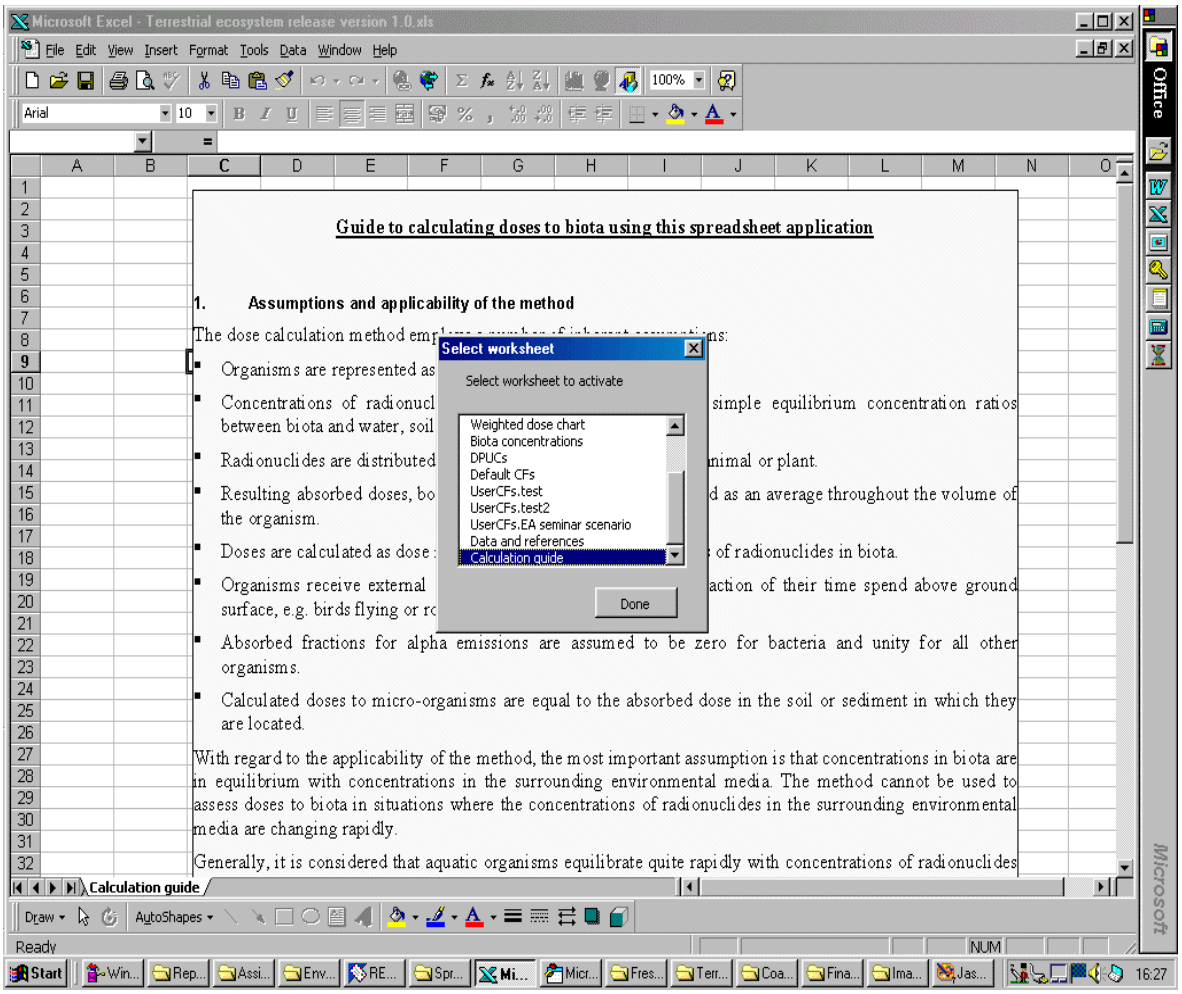
Moving the slider, clicking to one side or another of the slider, or clicking on the slider scroll buttons will adjust the graph scale.

The ‘Done’ button closes the scale adjustment utility. The ‘Reset’ button returns to the default scale maximum, 500 µGy h^{-1} for the aquatic ecosystems and 50 µGy h^{-1} for the terrestrial ecosystem.

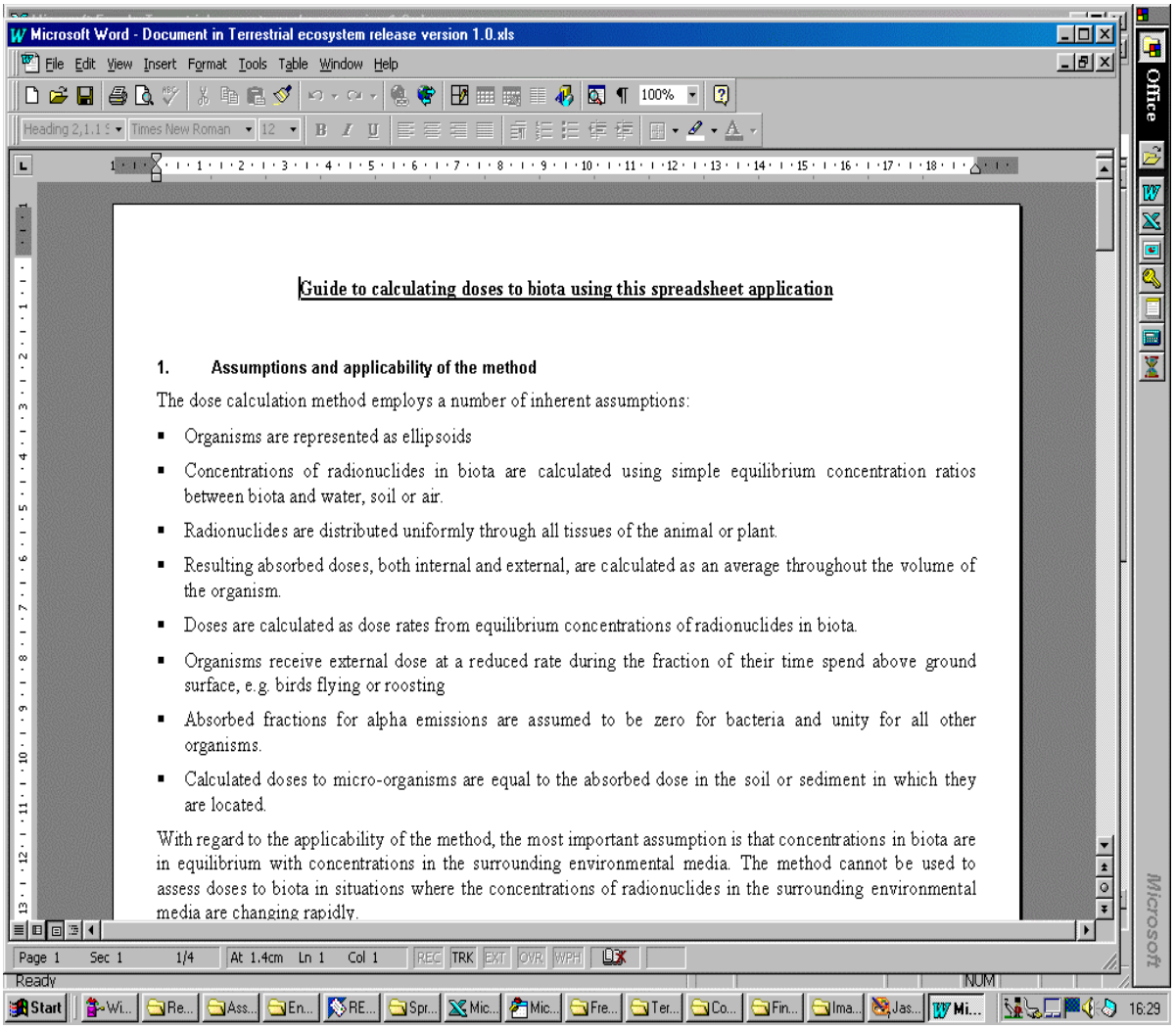
The graphs can be printed from Excel.

A3.1.2.4 Accessing the calculation guide and the worksheet containing data and references

These worksheets are accessed in the normal way through the 'navigate' button on the control panel:



After a short delay MS Word automatically opens the embedded document:

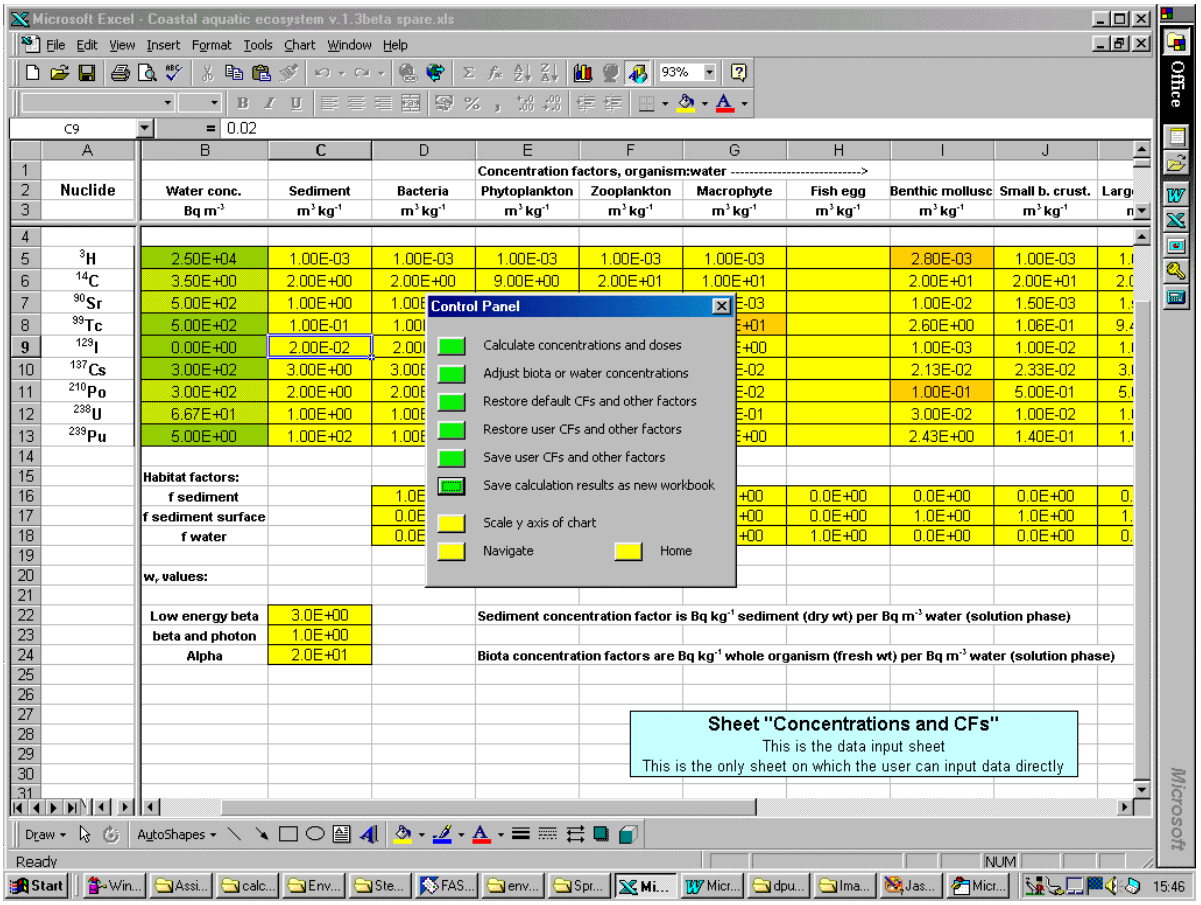


The document can be scrolled exactly as for any Word document. It is locked to prevent any changes, but it can be printed in the normal way. To de-activate the embedded document close it in the normal way and you will be returned to the worksheet. The document can also be re-opened by clicking anywhere on the document outline on the worksheet.

A3.1.3 Saving calculation results

Once the calculation is complete, the results can be saved to a separate workbook, leaving the assessment workbook unchanged and available for further calculations.

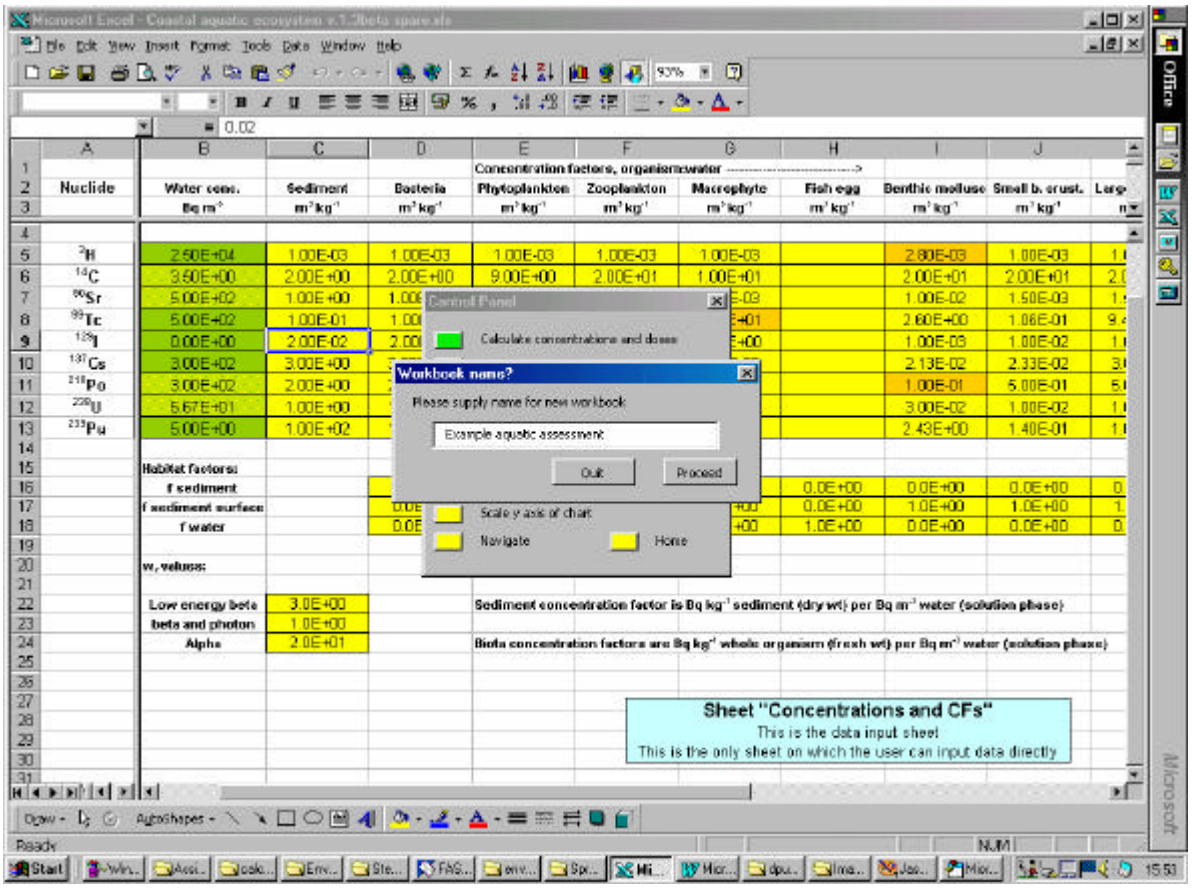
Return to the data input sheet "Concentrations and CFs" using the navigation utility or the "Home" button on the control panel:



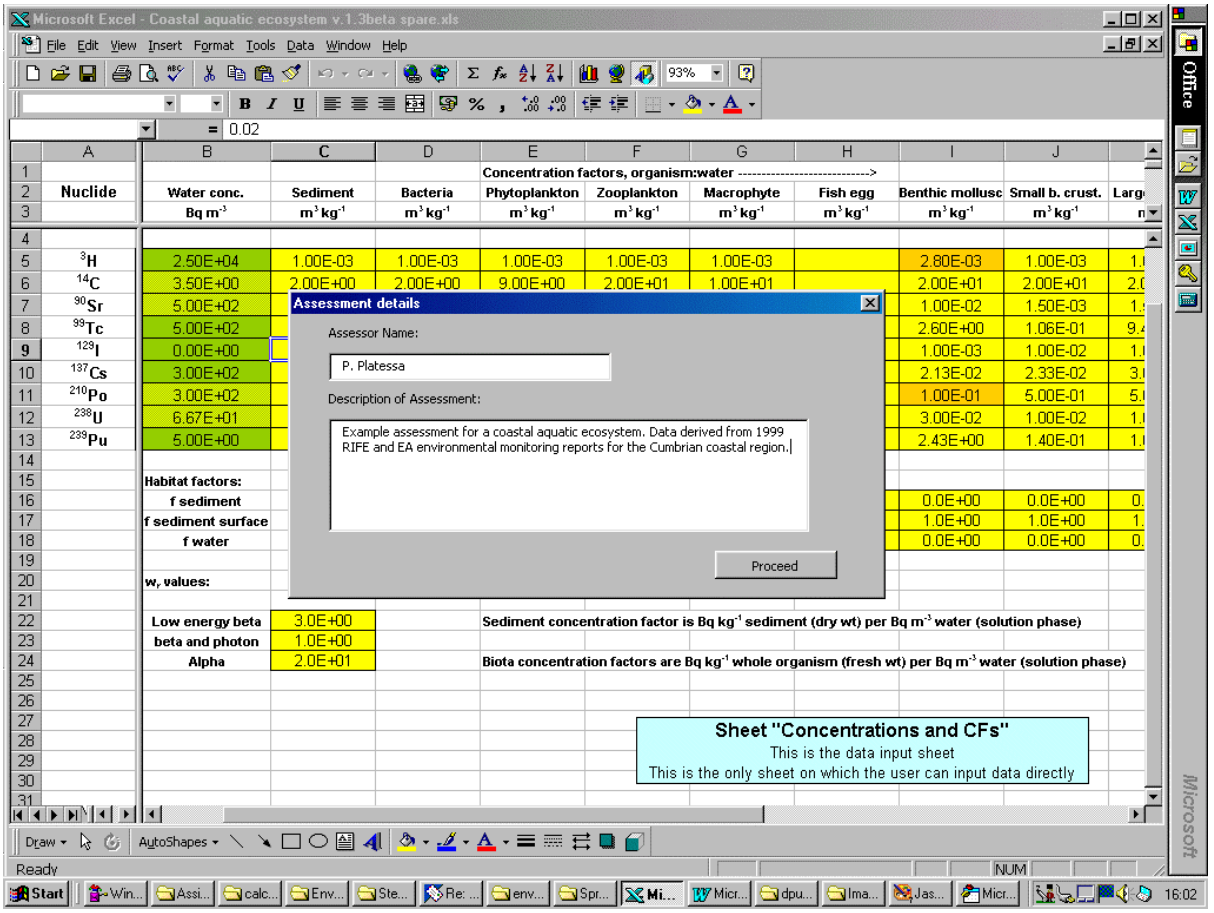
You will notice that concentration factors altered from the default are shaded orange, whilst water concentrations calculated from biota measurements are shaded light green. These codings will be preserved in the saved workbook.

"Save calculation results as new workbook"

By clicking this button you will be prompted for a filename (maximum of 31 characters):

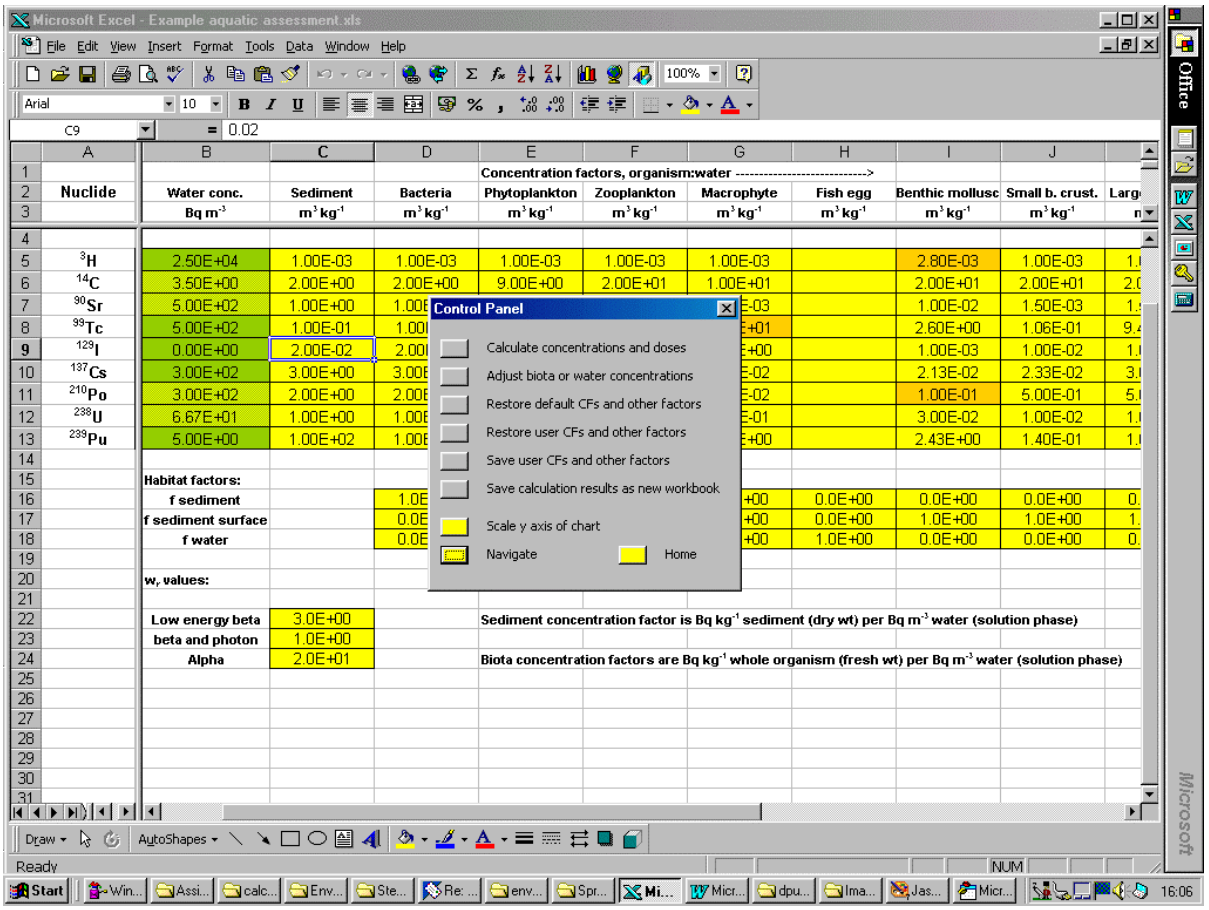


"Quit" cancels the save, "Proceed" activates the save. Next you are prompted for some details about the assessment:



"Proceed" completes the save.

The new results workbook is saved into the same directory as that in which you have placed the assessment workbook; the results workbook becomes the active workbook.



So long as the assessment workbook is open in the background, the control panel can be accessed by pressing "F1"; however only the "Navigate", "Scale y axis of chart" and "Home" buttons are active. "Home" takes you back to the data input sheet of the assessment workbook.

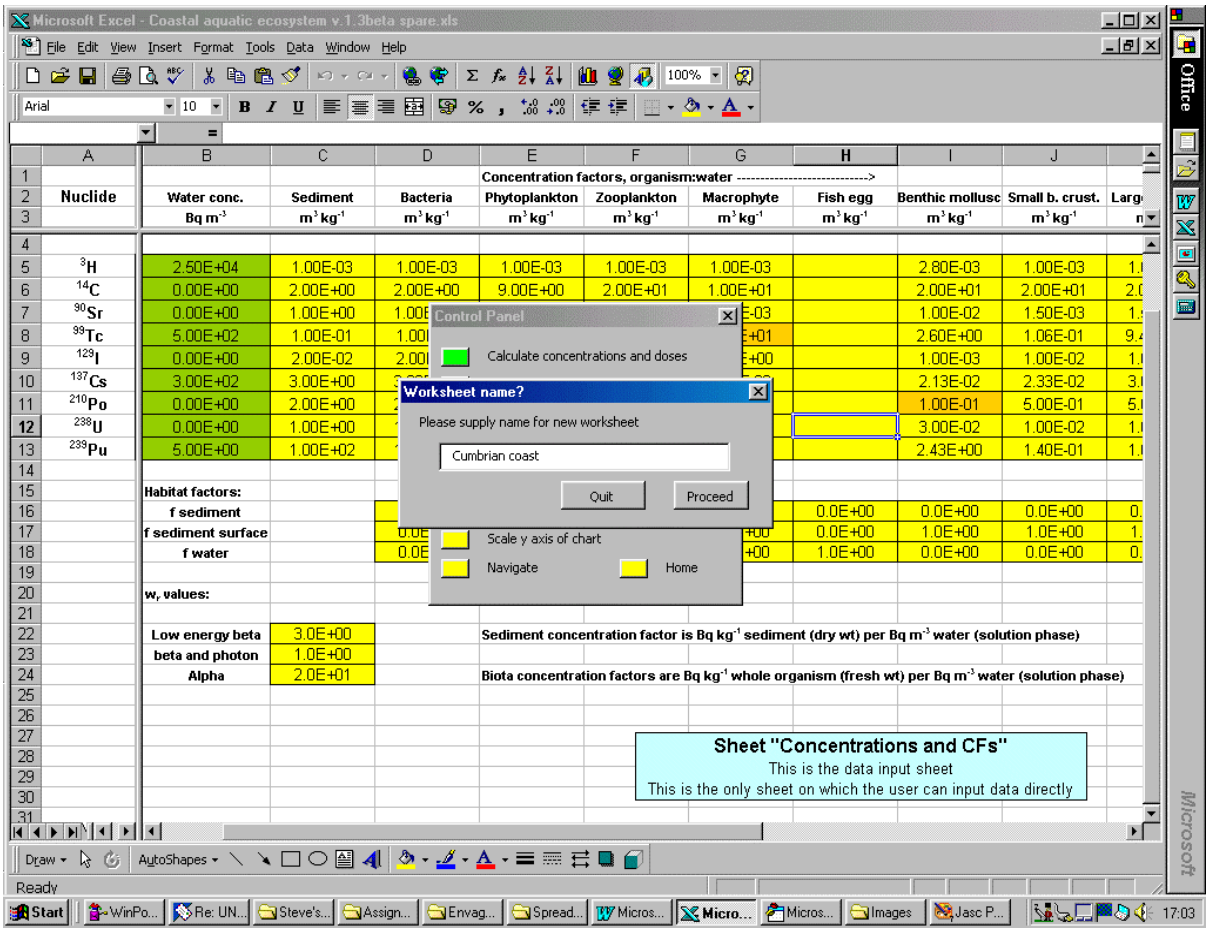
The saved results workbook does not contain any of the macros required to perform calculations; it is purely intended as a record of calculation results.

A3.1.4 Saving user defined concentration factors and other data

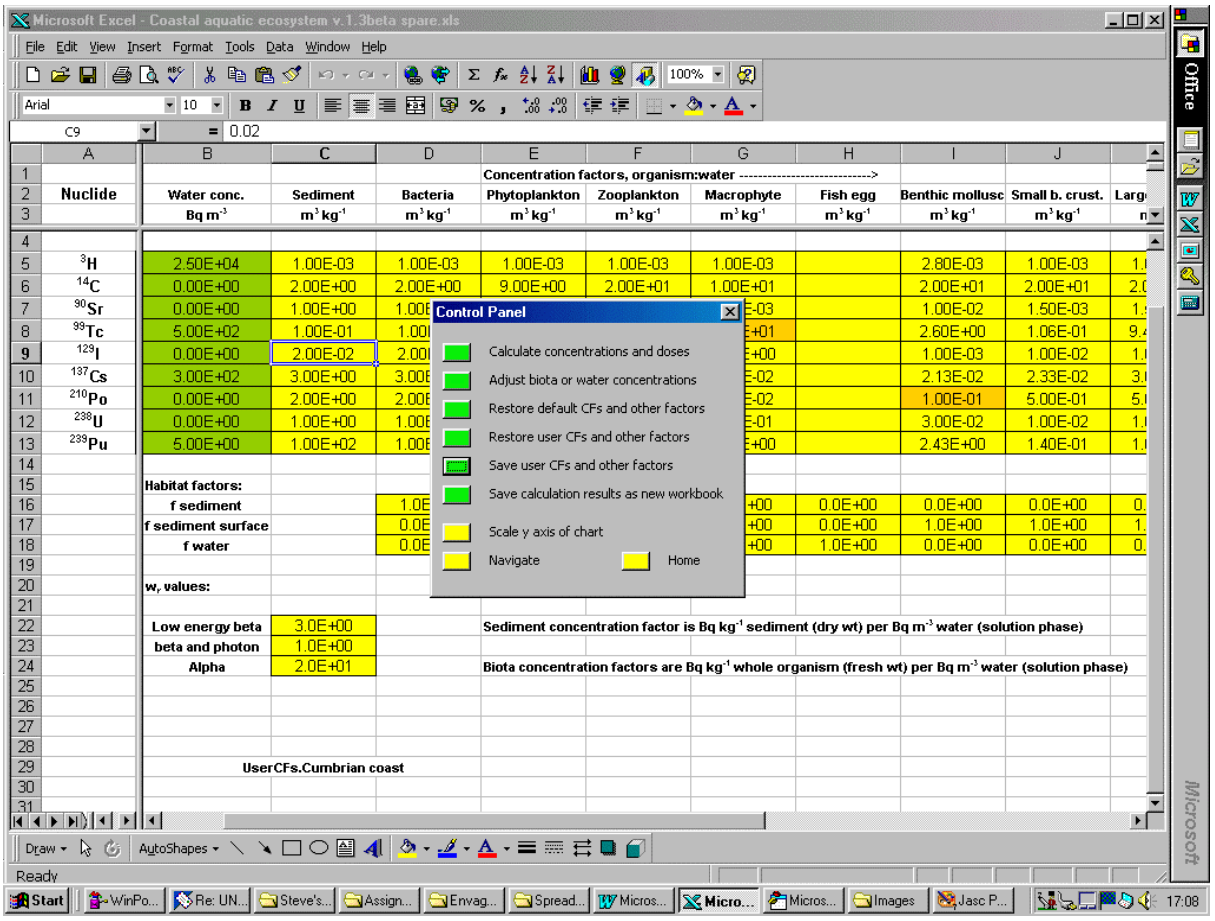
Once a calculation has been completed, you may want to save the concentration factors and other data so that they can be used again in future calculations.

"Save user CFs and other factors"

Clicking this button on the control panel will prompt you for a worksheet name (maximum 23 characters).



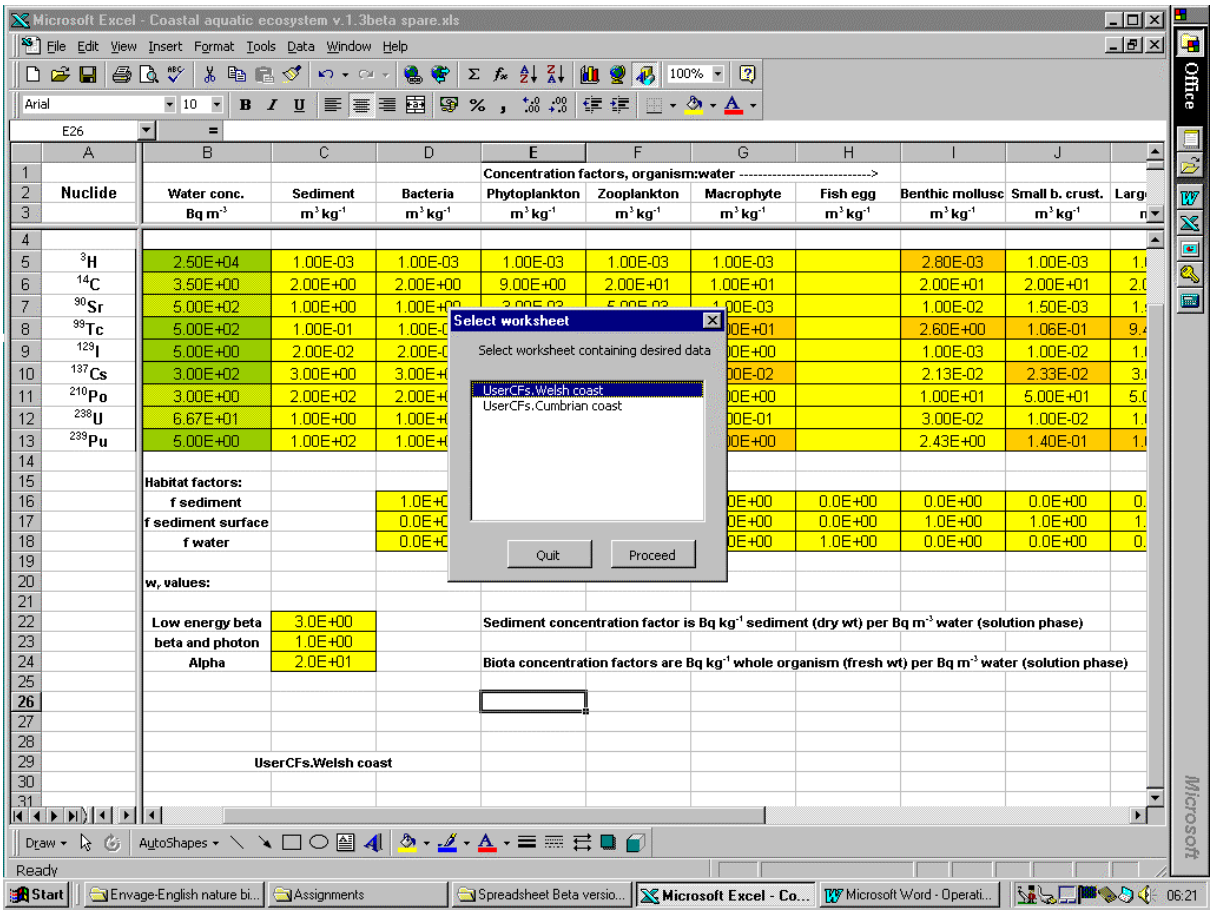
"Proceed" activates the save, whereas "Quit" aborts it. The workbook programme adds the prefix "UserCFs." to your chosen name, to enable it to be identified as user defined data for restoring (see below), and the new worksheet becomes the active sheet:


Microsoft Excel - Coastal aquatic ecosystem v.1.3beta spare.xls
File Edit View Insert Format Tools Data Window Help
Concentration factors, organism:water ---------->
Nuclide
Water conc. Bq m-3
Sediment m3 kg-1
Bacteria m3 kg-1
Phytoplankton m3 kg-1
Zooplankton m3 kg-1
Macrophyte m3 kg-1
Fish egg m3 kg-1
Benthic mollusc m3 kg-1
Small b. crust. m3 kg-1
3H 2.50E+04 1.00E-03 1.00E-03 1.00E-03 1.00E-03 1.00E-03 2.80E-03 1.00E-03
14C 0.00E+00 2.00E+00 2.00E+00 9.00E+00 2.00E+01 1.00E+01 2.00E+01 2.00E+01
90Sr 0.00E+00 1.00E+00
99Tc 5.00E+02 1.00E-01
129I 0.00E+00 2.00E-02
137Cs 3.00E+02 3.00E+00
210Po 0.00E+00 2.00E+00 1.00E-01 5.00E-01
238U 0.00E+00 1.00E+00 3.00E-02 1.00E-02
239Pu 5.00E+00 1.00E+02 2.43E+00 1.40E-01
Habitat factors:
f sediment
f sediment surface
f water
w, values:
Low energy beta 3.0E+00
beta and photon 1.0E+00
Alpha 2.0E+01
Sediment concentration factor is Bq kg-1 sediment (dry wt) per Bq m-3 water (solution phase)
Biota concentration factors are Bq kg-1 whole organism (fresh wt) per Bq m-3 water (solution phase)
UserCFs.Cumbrian coast
Control Panel
Calculate concentrations and doses
Adjust biota or water concentrations
Restore default CFs and other factors
Restore user CFs and other factors
Save user CFs and other factors
Save calculation results as new workbook
Scale y axis of chart
Navigate
Home
Ready
Start
17:08

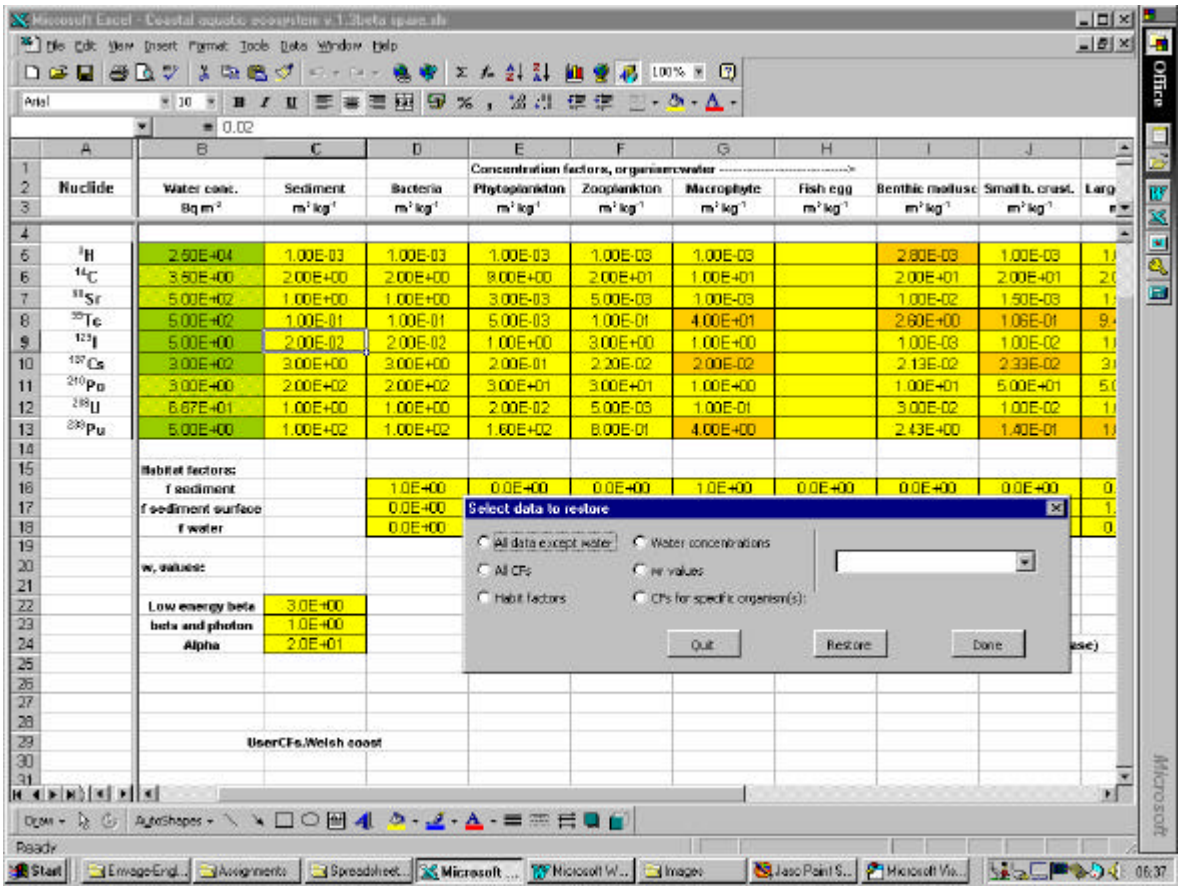
A3.1.5 Restoring saved concentration factors and other data

"Restore user CFs and other factors"

Clicking this button brings up a list box inviting you to select the saved dataset you wish to restore:



Clicking on a name in the list selects that saved dataset and makes its worksheet active. Clicking "Proceed" presents choices as to which data components you wish to restore to the data input sheet:



The options available are:

All data except water : Restores concentration factors, habit factors and w_r values (radiation weighting factors)

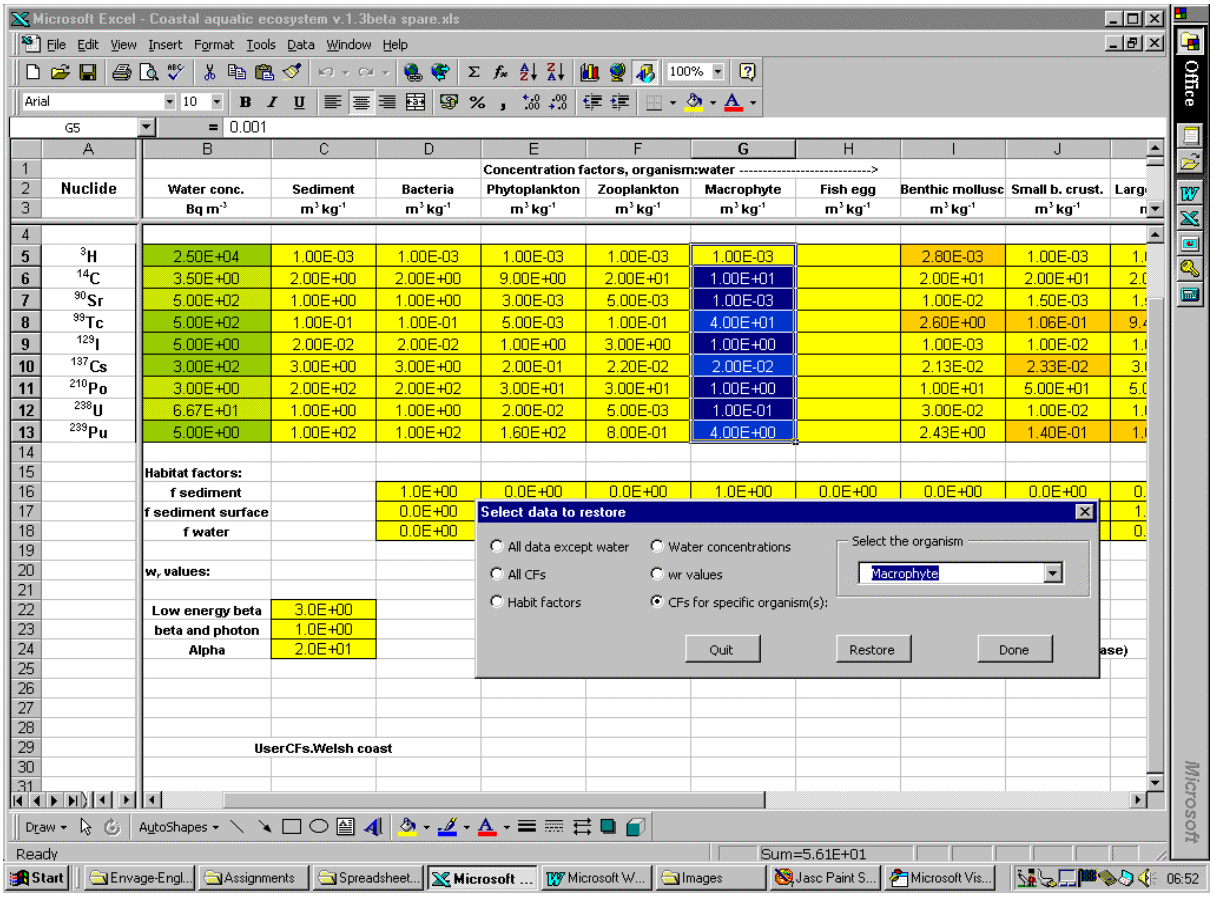
All CFs : restores all CFs only

Habit factors: restores habit factors such as the time spent in different parts of the ecosystem e.g underground or on the surface, only

Water concentrations : restores water concentrations only

wr values: restores w_r values (radiation weighting factors) only

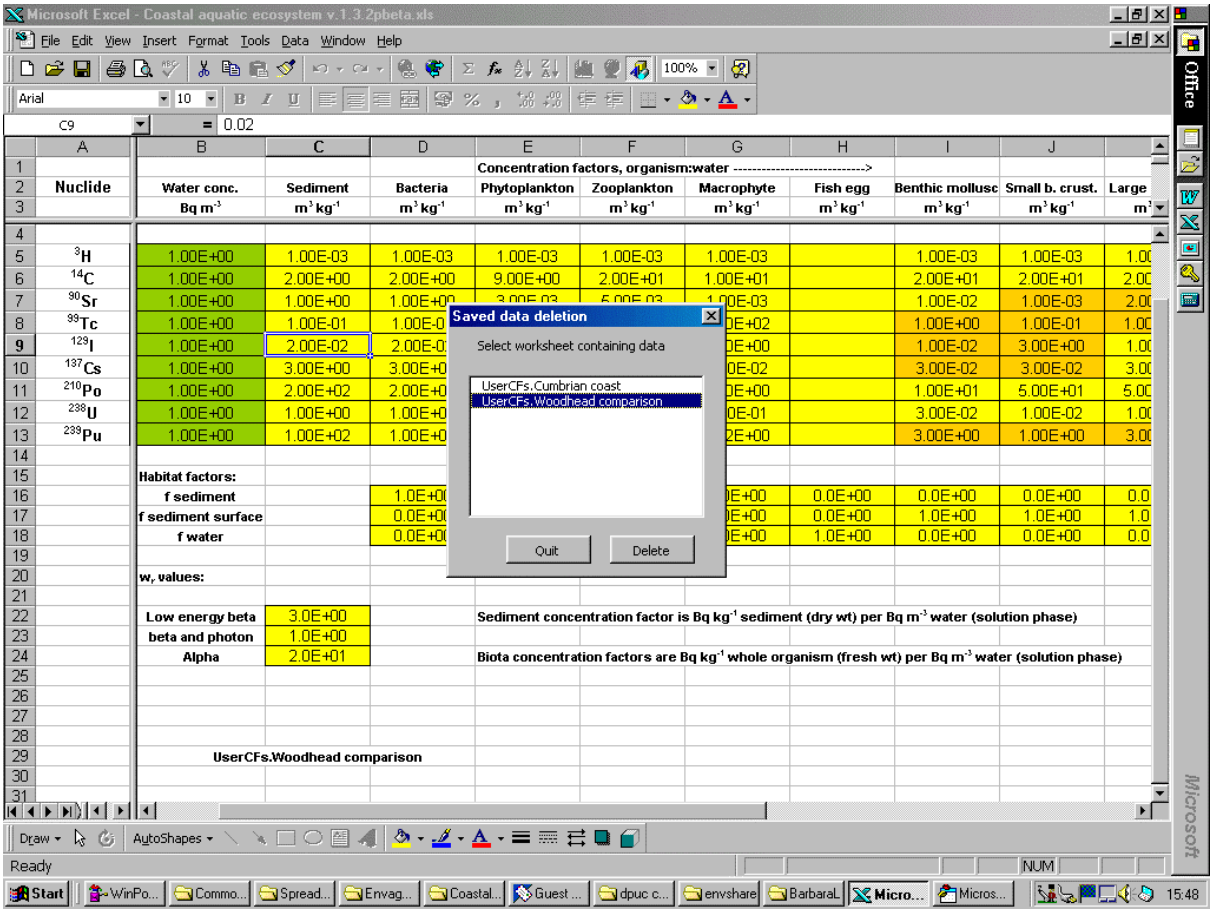
CFs for specific organisms: Activates the drop down box to permit selection of a set of CF values for one organism only:



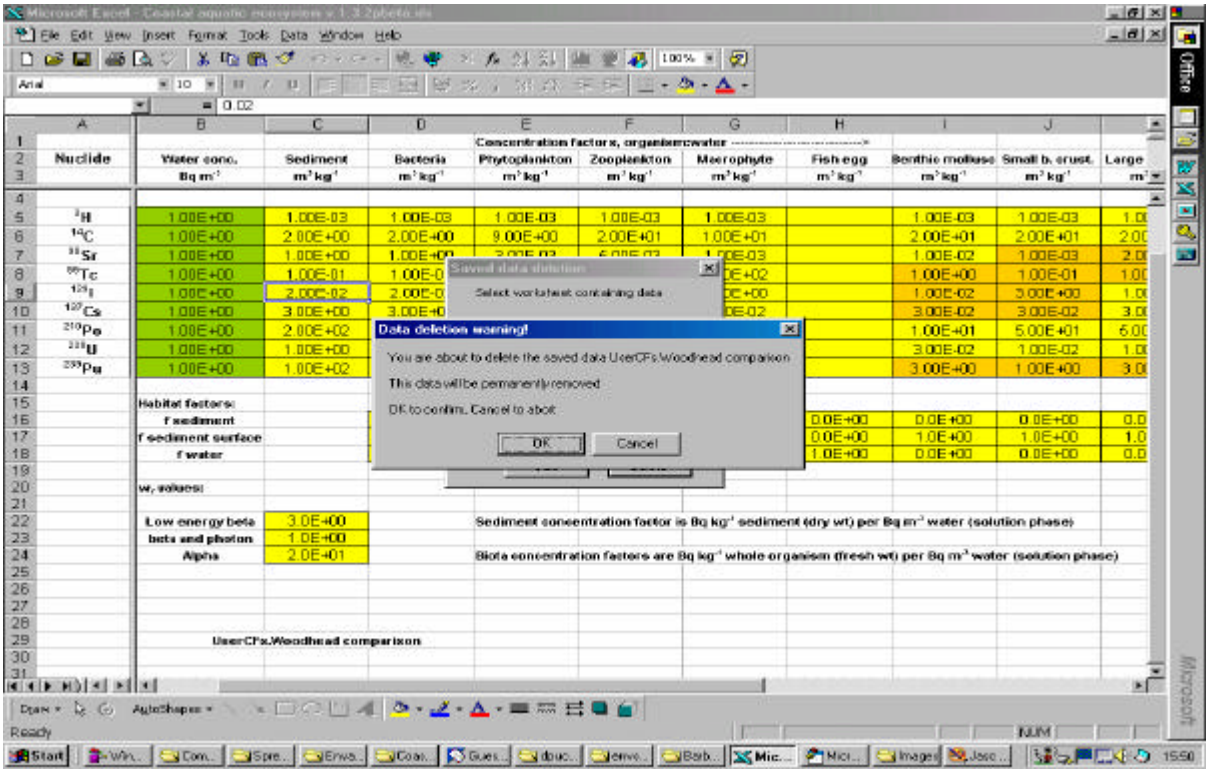
The “Restore” button initiates restoration of the selected data to the data input sheet. The dialog box remains active to permit further selections to be made. When all selections have been made, “Done” re-runs the concentration and dose calculations with the restored input data.

A3.1.6 Deleting saved concentration factors and other data

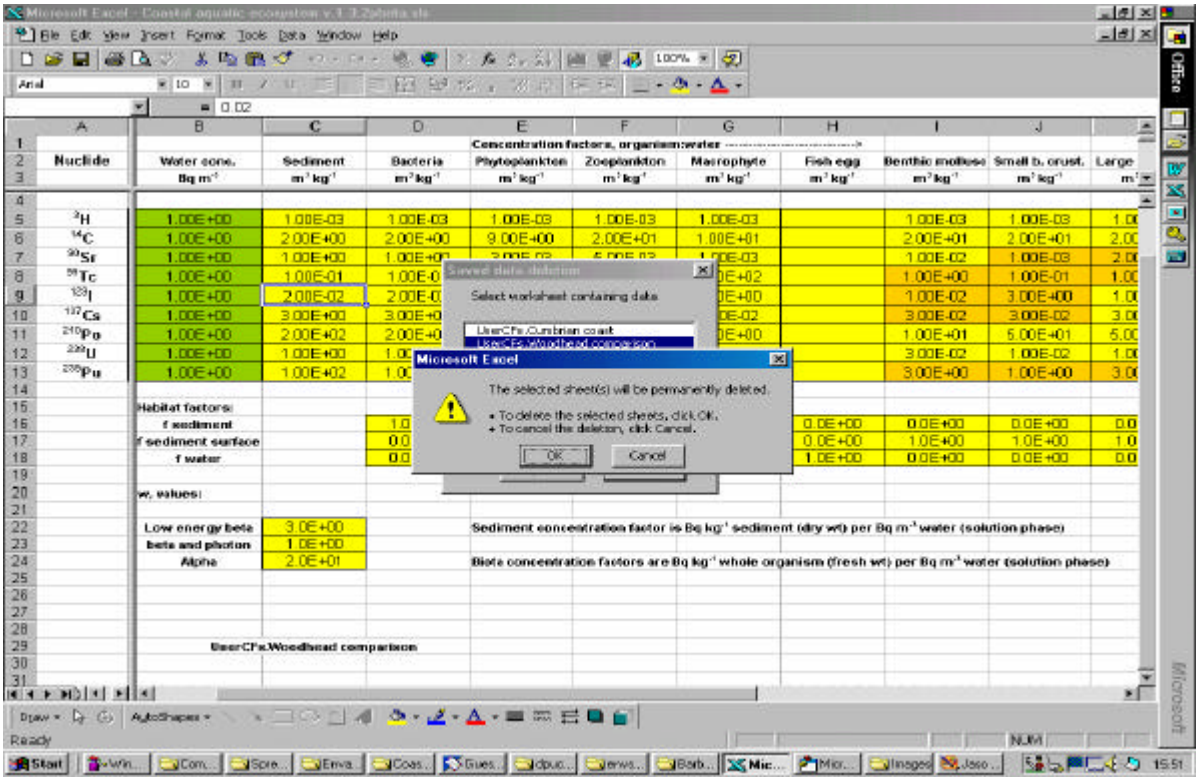
You cannot delete saved input sheets from the "Excel Edit – Delete sheet" menu because all worksheets are protected. Pressing F2 will bring up a dialogue box inviting you to select a sheet to delete:



Note that you are only offered the option of deleting sheets with a “UserCFs.” prefix. Highlight the sheet you want to delete and press “Delete”; this will bring up a warning:



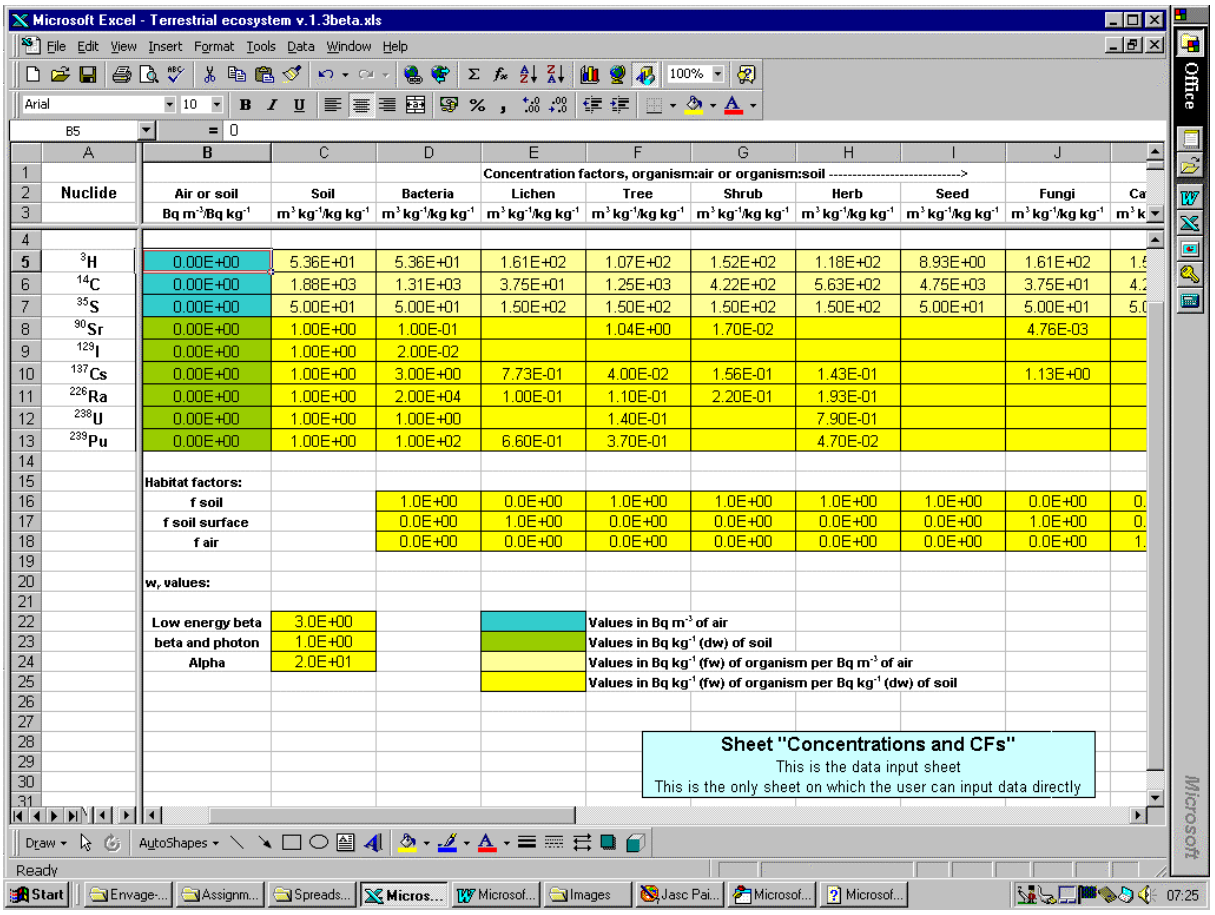
"Cancel" will abort the delete, whereas "OK" will elicit a final warning from Excel itself:



Pressing "OK" will cause the selected sheet to be deleted.

A3.2 Terrestrial ecosystem

Operation of the spreadsheet programme for the terrestrial ecosystem is exactly the same as for the aquatic ecosystem. The only difference is in the data input sheet "Concentrations and CFs":



Here, input concentrations for ^{3}H, ^{14}C and ^{35}S are as calculated or measured **Bq m^{-3} in air**. Concentration factors for these nuclides are in the units **Bq kg^{-1} fresh weight of organism per Bq m^{-3} in air** or **Bq kg^{-1} dry weight of soil per Bq m^{-3} in air**.

For all other nuclides input data are in terms of **Bq kg^{-1} dry weight of soil** and **Bq kg^{-1} fresh weight of organism per Bq kg^{-1} dry weight of soil.**

The concentration factors specified for these other radionuclides are in **Bq kg^{-1} dry weight of soil per Bq kg^{-1} wet weight of soil** and allow the spreadsheet to take account of the moisture content of the soil when calculating external doses.

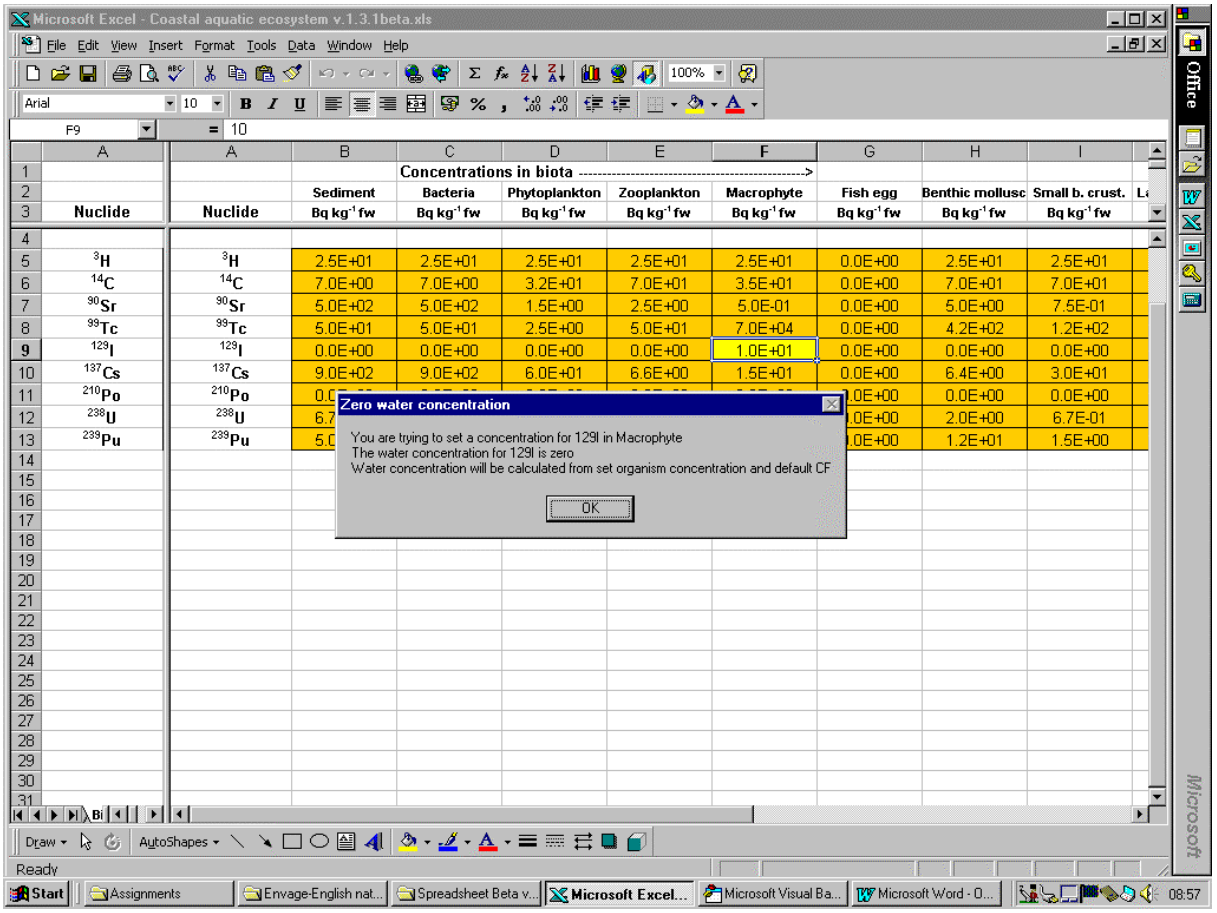
The habitat factors in this case represent the proportion of time which the organism spends buried in soil, on the soil surface, or above the soil surface when flying or roosting.

Operation of the spreadsheet is exactly the same as explained above for the aquatic ecosystem spreadsheets.

A3.3 Error messages

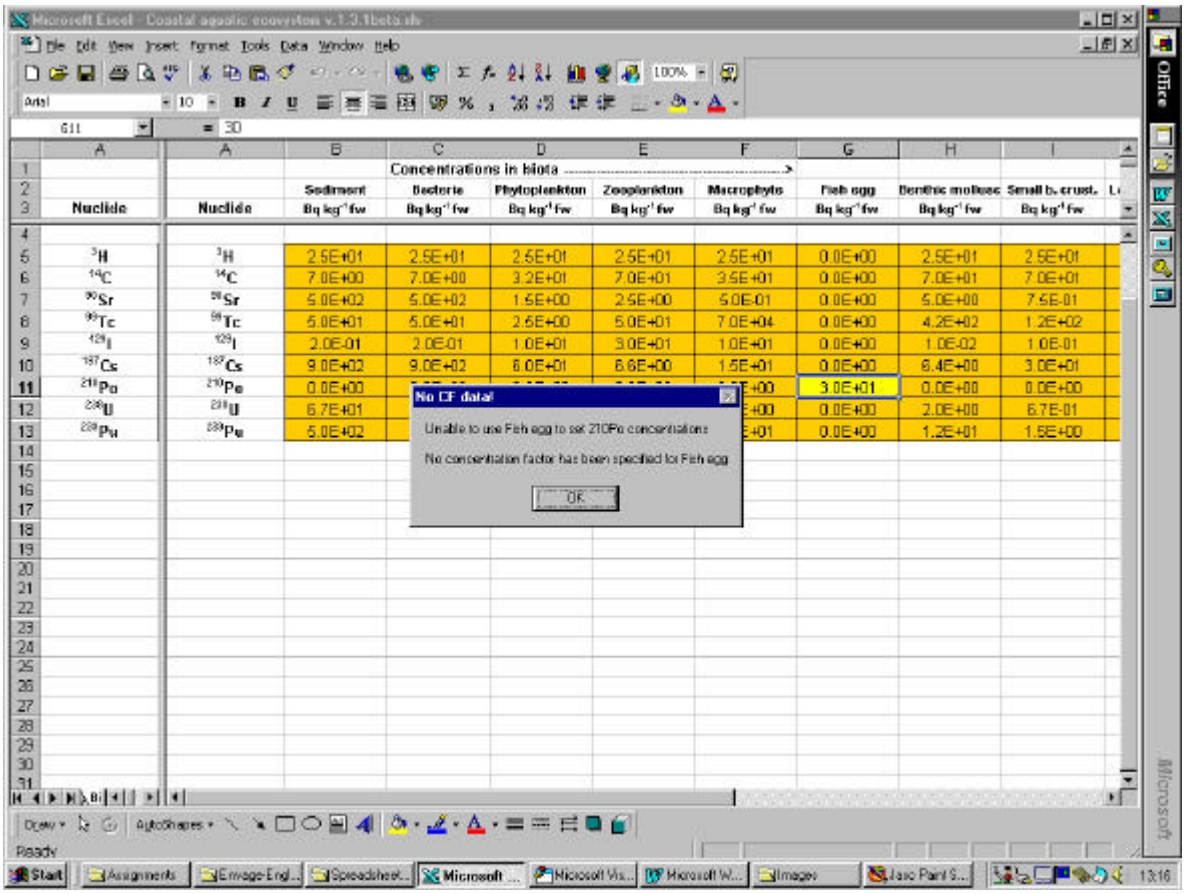
The programme code which executes the calculations and controls the user interface has a number of 'traps' to catch errors which would otherwise cause the programme to fail. These generate messages to alert the user as follows:

A3.3.1 Adjusting biota concentrations



Example 1:

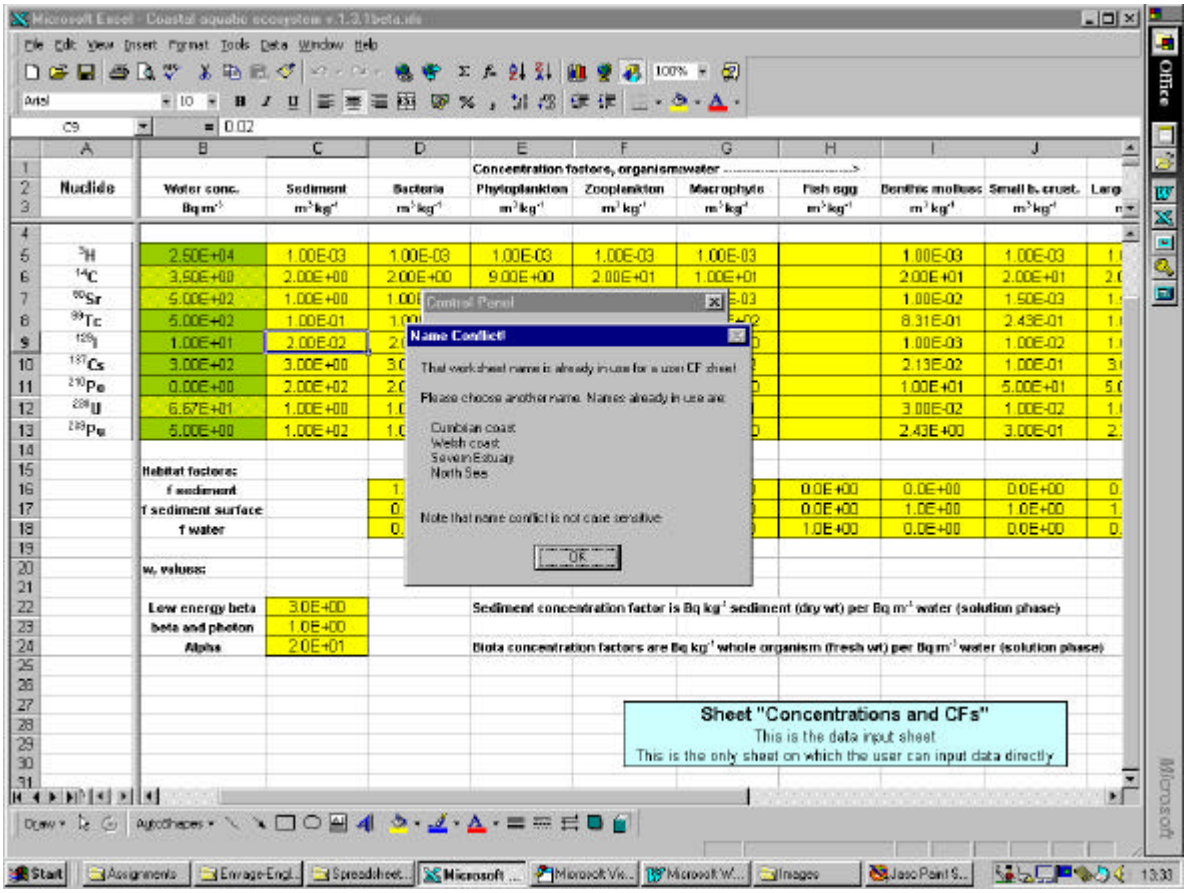
You are trying to fix a concentration of ^{129}I in macrophytes by adjusting the CF value for macrophytes. But the water concentration of ^{129}I is zero, so the programme can't do this. Instead it will use the default CF value and the concentration you have entered to estimate the water concentration.



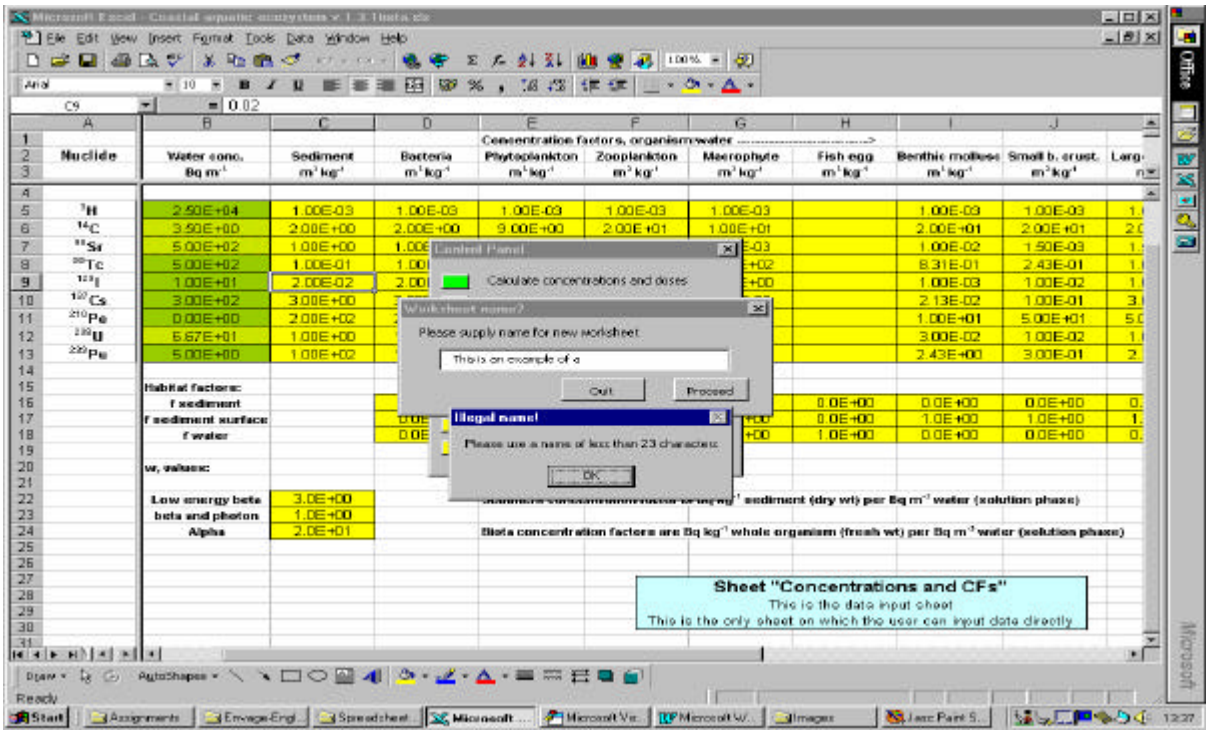
Example 2:

You have tried to use a measured concentration of ^{210}Po in fish eggs to adjust the CF value. But the current concentration of ^{210}Po in water is zero. After getting the previous error message and clicking OK, you are now advised that no default CF has been provided for ^{210}Po in fish eggs, so the programme can do nothing. On clicking OK both the water concentration for ^{210}Po will remain at zero and the CF will still be undefined.

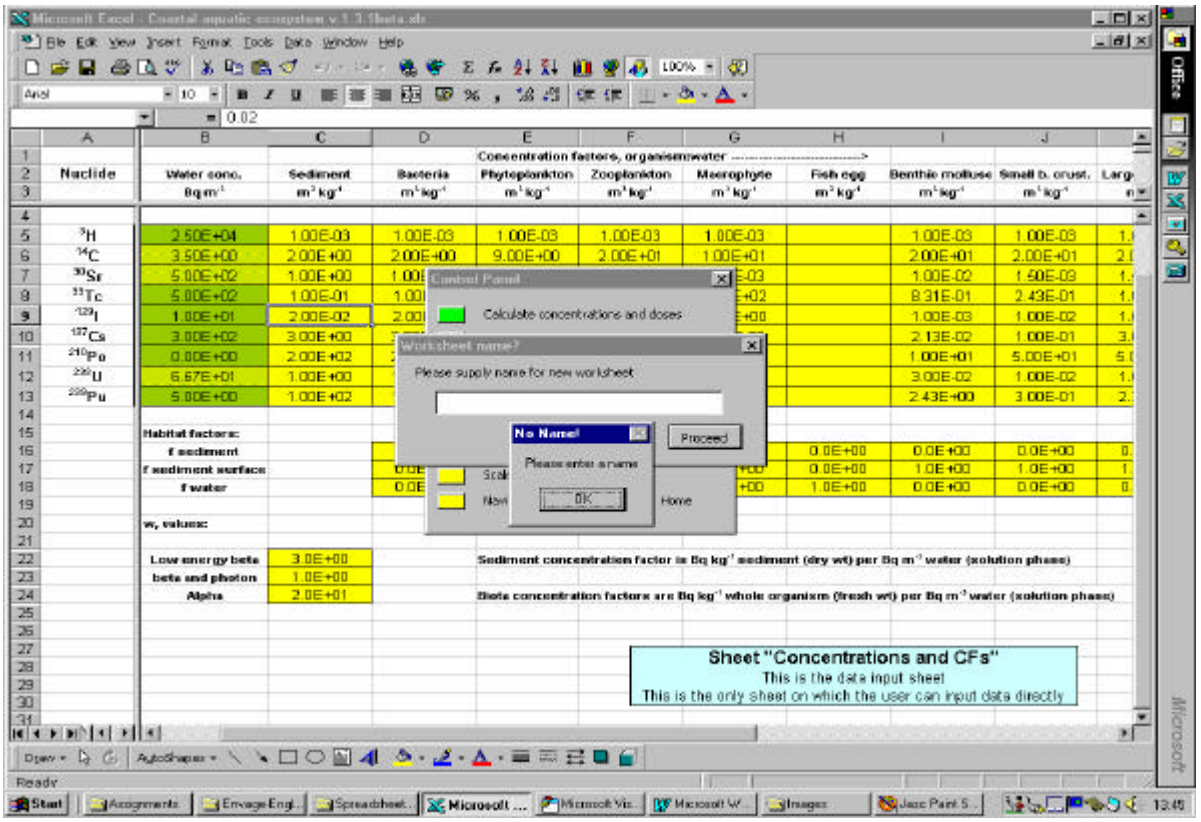
A3.3.2 Saving input data worksheets and results workbooks



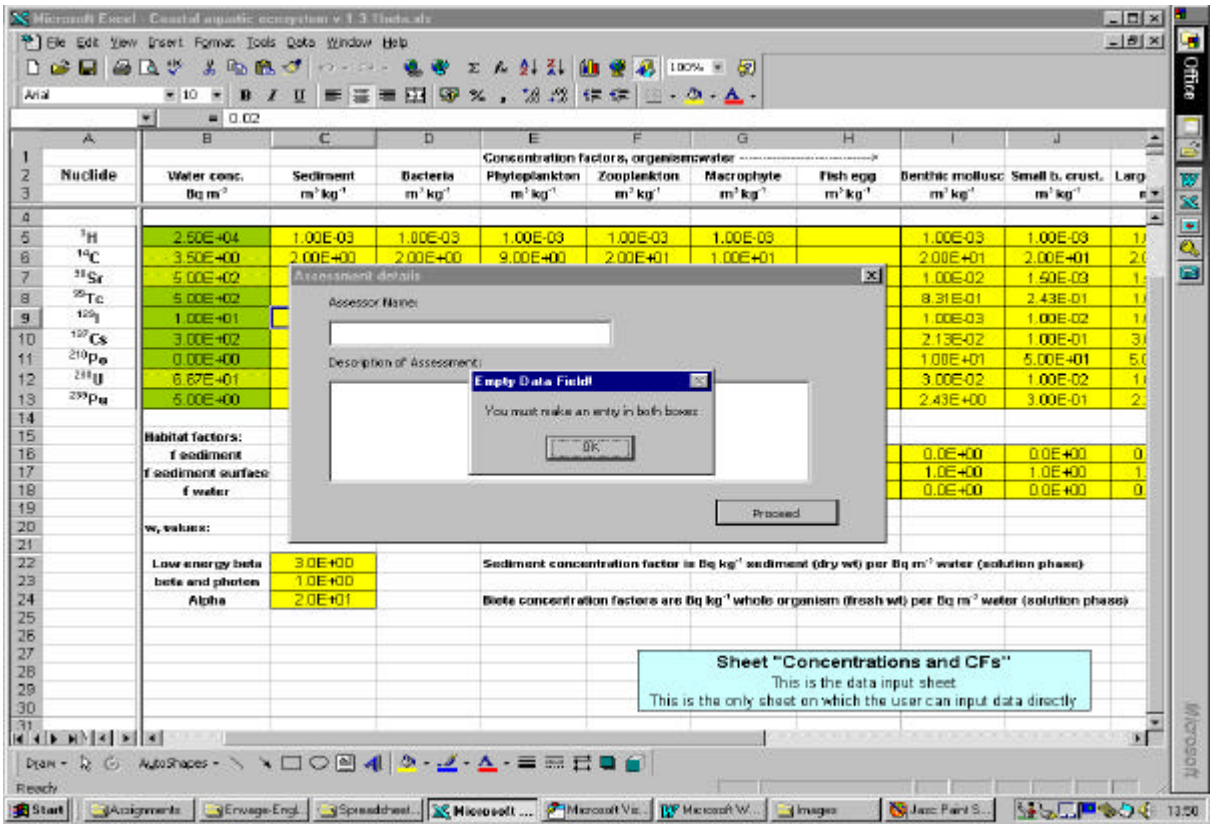
Another worksheet or workbook already exists with the name you have chosen. Click OK and choose another name. Note that name conflicts are not case sensitive: Windows will treat “Cumbrian Coast” and “Cumbrian coast” as the same name.



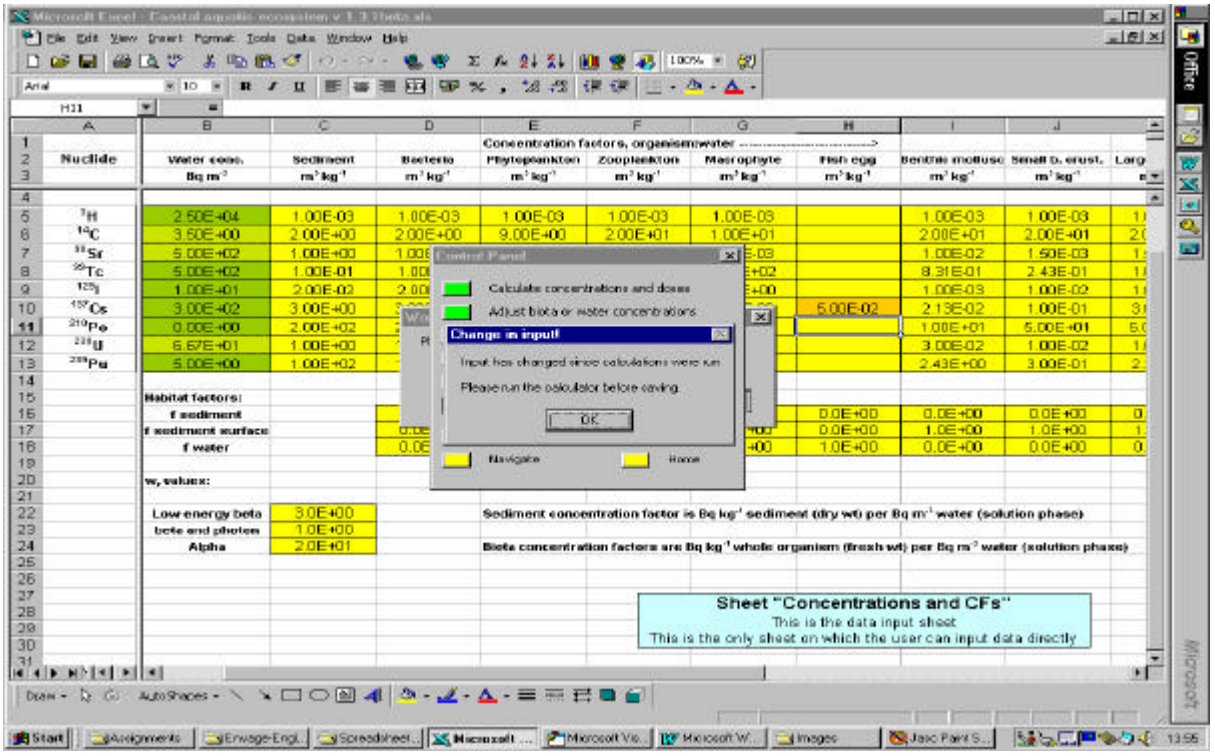
The number of characters you have typed in to the name box exceeds the maximum number of characters you can use in a name. For a workbook, you can have 31 characters in the name; for a worksheet only 23 characters are allowed because the programme must add the prefix "UserCFs." Click "OK" and type a shorter name.



You haven't entered any characters at all for the name. Press OK and enter at least one character.



When saving a results workbook you must provide both an assessor name and at least some identifying details for the assessment.



You have altered the input data, then tried to save the results workbook without running the concentration and dose calculator. The saved input data and results would then be inconsistent, so the programme insists you run the concentration and dose calculation prior to saving.

A3.4 Worksheet and workbook protection

The worksheets within the calculation workbook are protected against change, as is the worksheet layout in terms of rows and columns; the calculation workbook is protected against the insertion, deletion, or re-naming of worksheets. Routines initiated by the control panel remove and re-apply this protection as required for their operation; but the unprotected workbook and worksheets are never directly accessible to the user. As a result,

You can:

- Make direct entries in the yellow and green shaded cells of the worksheet "Concentrations and CFs";
- Save the workbook, or save the workbook with another filename;
- Re-name the workbook using the normal Windows menus;
- Delete the workbook only after removing the read-only password ("Biota") accessed through its 'file properties' tab.
- Copy ranges of cells, or charts, using the usual Excel commands and paste them into another workbook or another application such as a Word document.

You cannot:

- Make direct entries in any cells other than the yellow and green shaded cells of "Concentrations and CFs";
- Add or delete rows and columns in any worksheet;
- View or amend the Visual Basic for Applications code which drives the calculations;
- Add or delete any worksheets, other than by using the save and delete utilities on the control panel.

Saved results workbooks are protected in a similar way, but in this case even the yellow and green shaded cells of "Concentrations and CFs" are protected.

A3.5 Equations

The spreadsheet calculates doses according to the following equations. Calculation results always reflect the cell values on "Concentrations and CFs" and "DPUCs" at the time of execution. **If cell values in "Concentrations and CFs" are changed the calculation results will not change until the button "Calculate concentrations and doses" is clicked.**

A3.5.1 Aquatic ecosystems

Equations for the coastal and freshwater aquatic ecosystems are identical.

$$(\text{Sediment conc})_{nuclide} = (\text{Water conc})_{nuclide} \times CF_{nuclide}^{\text{sediment}} \times (\text{solids fraction})$$

$$(\text{Internal dose})_{nuclide,organism} = (\text{Water conc})_{nuclide} \times CF_{nuclide}^{\text{organism}} \times DPUC_{nuclide,organism}^{\text{internal}}$$

$$(\text{External dose})_{nuclide,organism} = DPUC_{nuclide,organism}^{\text{external}} \times \begin{bmatrix} (\text{Sediment conc})_{nuclide} \times (fsed_{organism} + fsedsur_{organism}/2) \\ + fwater_{organism} \times (\text{Water conc})_{nuclide}/1000 \end{bmatrix}$$

Where:

Sediment concentrations are in Bq kg^{-1} dry weight;

Water concentrations are in Bq m^{-3} in the dissolved phase;

Concentration factors *(CF)* are in m^3 kg^{-1};

Dose per unit concentration factors *(DPUC)* are in μGy h^{-1} per Bq kg^{-1} fresh weight;

$fsed_{organism}$ is the fraction of time the organism spends buried in sediment;

$fsedsur_{organism}$ is the fraction of time the organism spends at the sediment/water interface; and

$fwater_{organism}$ is the fraction of time the organism spends free swimming in the water column.

Doses are calculated as weighted or unweighted by use of unweighted dose per unit concentration factors for the separate radiation types, to which an appropriate radiation weighting factor can be applied.

A3.5.2 Terrestrial ecosystem

The equations used to calculate dose in the terrestrial ecosystem are very similar:

$$(\text{Soil conc})_{nuclide} = (\text{Air conc})_{nuclide} \times CF^{\text{soil}}_{nuclide} \qquad (\text{for } {}^{3}\text{H, } {}^{14}\text{C and } {}^{35}\text{S})$$

$$(\text{Soil conc})_{nuclide} = (\text{Soil conc(dry)})_{nuclide} \times (\text{solids fraction}) \qquad (\text{for other nuclides})$$

$$(\text{Internal dose})_{nuclide,organism} = (\text{Air conc})_{nuclide} \times CF^{\text{organism}}_{nuclide} \times DPUC^{\text{internal}}_{nuclide,organism} \qquad (\text{for } {}^{3}\text{H, } {}^{14}\text{C and } {}^{35}\text{S})$$

$$(\text{Internal dose})_{nuclide,organism} = (\text{Soil conc})_{nuclide} \times CF^{\text{organism}}_{nuclide} \times DPUC^{\text{internal}}_{nuclide,organism} \qquad (\text{for other nuclides})$$

$$(\text{External dose})_{nuclide,organism} = DPUC^{\text{external}}_{nuclide,organism} \times \left[(\text{Soil conc})_{nuclide} \times \left(\begin{array}{l} (fsoil_{organism} + fsoilsur_{organism}/2) \\ + fair_{organism} \times (\text{reduction factor})_{\text{radiation type}} \end{array} \right) \right]$$

where:

Air concentrations for ^{3}H, ^{14}C and ^{3}H are in Bq m^{-3}, and for other nuclides the input values for soil are in Bq kg^{-1} dry weight;

Concentration factors for ^{3}H, ^{14}C and ^{3}H are as Bq kg^{-1} (fresh weight) of soil or organism per Bq m^{-3} in air, and for other nuclides are as Bq kg^{-1} (fresh weight) of organism per Bq kg^{-1} (dry weight) of soil;

(solids fraction) is the fractional dry solids content of fresh soil;

fsoil is the fraction of time the organism spends buried in, or burrowing into, soil;

fsoilsur is the fraction of time the organism spends on the ground surface;

fair is the fraction of time the organism spends above the ground surface, flying or roosting etc.;

(reduction factor) is a factor, dependent on radiation type, by which the radiation dose rate above the ground surface is lower than that within the soil itself. The default vales set for this factor are zero for á and low energy â radiation, and 0.25 for high energy â radiation and ã ray photons.