The Methodology for Assessing the Radiological Consequences of Routine Releases of Radionuclides to the Environment Used in PC-CREAM 08

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FOREWORD

This report describes the methodology that has been implemented in the software PC-CREAM 08 to assess the radiological impact of routine discharges of radioactive material into the environment. PC-CREAM 08 has been developed by the Health Protection Agency (HPA), with permission from the European Commission (EC), and is an updated version of the EC code PC-CREAM 98. This methodology is closely based on that developed by a number of EU organisations for the EC and which was published in 1995 (RP 72); this was in turn a further development of an original methodology developed for the EC and published in 1979. Although primarily developed for application in Western Europe, a generalised approach has been adopted so that some of the models and methods are appropriate for wider use. Default values are given for many parameters and these have been used to determine illustrative results. The models adopted in the methodology are those considered appropriate for routine releases, ie, releases that can be considered as continuous and constant.

Radiological impact assessments involve the calculation of radiation exposures to both individuals and population groups. In the absence of measurements this can is achieved through modelling. The models described in this report predict the transfer of radionuclides in the environment, the pathways by which people may be exposed to radiation and the resulting radiation doses received.

The radiological consequences of routine releases of radionuclides are determined using the framework of the system of radiological protection recommended by the International Commission on Radiological Protection (ICRP). The most recent recommendations of ICRP, issued in publication 103, have been taken into account in developing this methodology. However, it should be noted that dose coefficients based on revised radiation and tissue weighting factors have yet to be published and therefore these are taken from publication 60.

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The four sections of the original report are combined and published together

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CONTENTS

1 INTRODUCTION

In 1979 the Commission of the European Communities (CEC) published a report describing a methodology for evaluating the radiological consequences of radioactive effluents released in normal operations (CEC, 1979). This report found wide application within the European Union (EU). In 1993, following subsequent developments in all aspects of radiological assessment, the Commission contracted the National Radiological Protection Board (NRPB) to co-ordinate work, also involving other institutions in the EC, to produce a revised methodology for evaluating the radiological consequences of routine releases. The revised methodology, which was called CREAM (Consequences of Releases to the Environment: Assessment Methodology, was described in a European Commission (EC) report, RP 72 (Simmonds, Lawson and Mayall, 1995). Following the publication of the CREAM methodology, the EC contracted NRPB to produce a computer implementation of the methodology, which was called PC-CREAM (Mayall et al, 1997). About 200 copies of the EC computer system, PC-CREAM have been sold worldwide and it has been used in many applications, for example to estimate radiation exposures from discharges in the EU (Harvey et al, 2008). The Radiation Protection Division of the Health Protection Agency (formerly NRPB) has now developed an updated version of PC-CREAM (called PC-CREAM 08, Smith et al, 2009). This incorporates improvements to the methodology since the issue of the original PC-CREAM, as well as updates so that the system is compatible with modern personal computers and to make the user interface more intuitive. In developing PC-CREAM 08 account was taken of the views expressed by the PC-CREAM user group (Simmonds, 1999; Simmonds et al, 2000) on ways in which PC-CREAM could be improved. This report is an updated version of RP 72 that describes the methodology implemented in PC-CREAM 08. It describes the models that are included in the computer system and gives the parameter values that are included as defaults in the PC-CREAM 08 database. Additional information, for example on other models and data not included in PC-CREAM 08, is given in RP 72 (Simmonds, Lawson and Mayall, 1995) and is not repeated in this report.

The methodology is intended for use in assessing the radiological consequences to the population of the EU due to the discharge of radioactive effluents during normal operations. It was originally developed as part of a research programme on plutonium recycling in light water reactors and the radionuclides considered were those thought to be important in this context. The methodology and PC-CREAM 08 now include a broader range of radionuclides including those that might be discharged routinely from operations using naturally occurring radioactive materials (NORM). As the methodology was developed in the 1990s it only covered sites in the EU at that time. In producing PC-CREAM 08 it has been possible to include some additional information for countries that have joined the EU in 1995 or later but only to a limited extent. In some cases information is also included for non-EU European countries. Within PC-CREAM 08 and this report the following terminology is used to make it clear which countries are included.

TABLE 1.1 The terminology used to denote with European Union countries are included in the methodology and PC-CREAM 08

Terminology	Countries included
EU 12	Those in the EU from 1986: Belgium, Denmark, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, United Kingdom
EU 15	Those in the EU from 1995: As EU 12 plus Austria, Finland and Sweden
EU 25	Those in the EU from 2004: As EU 15 plus Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia.

Assessing the radiological consequences involves estimating radiation exposures to both individuals and to population groups (the collective dose). Individual exposures can be compared with the appropriate dose limits or constraints, while collective dose can be used as an indication of health detriment, in the form of the number of possible health effects. Both individual and collective doses can also form an essential input into the optimisation procedure for effluent treatment systems. When the methodology was first published in 1979 (CEC, 1979) it concentrated on estimating health detriment and therefore on the determination of collective doses. An important change in the revised methodology, CREAM (Simmonds, Lawson and Mayall, 1995), which has been maintained here, was that both individual and collective doses were considered, reflecting an increased emphasis on the former in radiological consequence assessments. This necessitated the inclusion of a wider range of potential exposure pathways than considered previously. A generalised approach has been adopted with the aim that CREAM will have broad application for the assessment of the radiological consequences of routine discharges. It may also be useful in other applications where radioactivity may be released to the environment. CREAM consists of a series of interlinked models which describe the transfer of radionuclides through the various sectors of the environment, the pathways by which people may be exposed to radiation, and the resulting radiation doses. The general steps involved in assessing radiological consequences are shown in Figure 1.1.

Radioactive effluents may be discharged to either the atmospheric or the aquatic environment and models have been developed to describe the transfer of radionuclides through the relevant parts of the biosphere to people. Radionuclides discharged to the atmosphere are dispersed due to normal atmospheric mixing processes. As they travel downwind they irradiate the population externally and internally, the latter due to the inhalation of radionuclides from the atmosphere. During their transport downwind radionuclides may be deposited from the atmosphere by impaction with the underlying surface or due to rainfall. This transfer onto land surfaces may lead to further irradiation of people by three important routes: external irradiation from deposited activity, internal irradiation from inhalation of resuspended activity and ingestion of contaminated food. The relative importance of these pathways depends on the radionuclide and the nature of the surface onto which the deposition occurs. Appropriate dosimetric models and habit data are also required to determine individual and collective doses. To estimate collective doses spatial distributions of population and agricultural production within the EU are required. Such spatial distributions are currently available for the EU 12 as matrices based on grids of varying dimensions.

Liquid radioactive effluents may be discharged to freshwater (principally rivers), estuarine or the marine environments. Radionuclides discharged to rivers are dispersed due to general water movements and sedimentation processes. The principal routes leading to the irradiation of people that have been modelled in PC-CREAM 08 are: external irradiation from sediments; ingestion of foods derived from the river; drinking water taken from the river. The dispersion of radionuclides discharged into the marine environment is determined firstly by the local features of the environment, in particular tidal currents and the degree of sedimentation. Subsequent dispersion is influenced by general water movements and sedimentation processes in the larger sea and ocean masses. There are again a number of pathways leading to irradiation of people including: ingestion of marine foodstuffs; external irradiation from activity on beaches; inhalation of sea-spray.

Some radionuclides, due to their long radioactive half-lives and their behaviour in the environment, may become globally dispersed and act as long-term sources of exposure of large populations. Models are described which evaluate the global circulation and exposure to people of such radionuclides as a result of their discharge to either the atmospheric or aquatic environment.

In this report a range of illustrative results of the environmental models is presented. The results are for selected radionuclides released at unit rate to the atmosphere or the aquatic environment and are in the form of predicted concentrations in various environmental materials, such as air or water. The concentration of radionuclides in terrestrial materials is evaluated for unit deposition per unit area of land. Only a limited range of radionuclides is considered, selected to cover those usually present in effluents from nuclear installations and found to have relative radiological significance. The matrix of results can be used to assess individual and collective doses to the EU population from a discharge of radionuclides at a particular location. In the case of atmospheric discharges, generic results are evaluated for a wide range of meteorological conditions and associated parameters. These results can be used for discharges at a particular location by summation over the various conditions, each weighted according to its frequency of occurrence. A less generalised approach has to be adopted for aquatic discharges where account must be taken, from the outset, of the particular features of the discharge environment. In the case of marine discharges, the Mediterranean and Northern European waters have been modelled and results are presented for unit discharge at two locations. Collective and individual doses can be determined from these matrices of results using appropriate habit data and dosimetric models for external and internal irradiation. For collective doses, the spatial distribution of the population and the production of terrestrial and marine foods is also required. The procedure for the evaluation of the matrix of results and its application to determine individual and collective doses is described in this report. A much wider range of results can be obtained using the computer system PC-CREAM 08.

The radiological consequences of routine releases of radionuclides are determined using the framework of the system of dose limitation recommended by the International Commission on Radiological Protection (ICRP). The recommendations of ICRP, issued in publication 60 (ICRP, 1991) and adopted by the EU (CEC, 1996), have been taken into account in developing this methodology. In particular, effective doses have been evaluated as defined in ICRP Publication 60. ICRP has recently published updated recommendations in Publication 103 (ICRP, 2007) and the impact of these is discussed in this report where appropriate. Previously in RP 72 (Simmonds, Lawson and Mayall, 1995) the relationships given in ICRP 60 (ICRP, 1991) between radiation exposure and health effects in the exposed population were used to estimate health detriment. At the levels of individual dose typically encountered from routine discharge, only the stochastic effects of radiation needed to be considered. These comprise fatal and non-fatal cancers in the exposed population and hereditary effects in its descendants. The relationship between doses and the incidence of these effects is discussed in this report taking into account the latest views of ICRP on the use of collective doses to estimate health detriment (ICRP, 2007).

The underlying assumptions and concepts adopted in the methodology are summarised in Chapter 2, together with the dosimetric quantities adopted. The models used to describe the transfer through the environment to man of radionuclides discharged to atmosphere and the aquatic environment, are outlined in Chapters 3 and 4, respectively. The models used to describe the transfer of those radionuclides which become globally dispersed are given in Chapter 5. Dose coefficients for unit intake by ingestion and inhalation of radionuclides are presented in Chapter 6. The procedures to estimate individual and collective doses in the population of the EU are outlined in Chapter 7, while the estimation of health detriment is discussed in Chapter 8.

It should be emphasised that the methodology, CREAM, and this report are concerned with assessing the radiological consequences of routine releases of radionuclides to the environment. Default values have been given for many parameters and they have been used to provide illustrative results. The choice of such default values is necessarily a compromise, taking into account the range of possible values. These default values and illustrative results should be used with caution and for specific sites and applications it will often be appropriate to choose alternative values. The models adopted in the methodology are those considered appropriate for considering routine releases. In each case the limitations of the models are discussed together with their validation and, where appropriate, more detailed models are referred to and their results compared with those included in CREAM.

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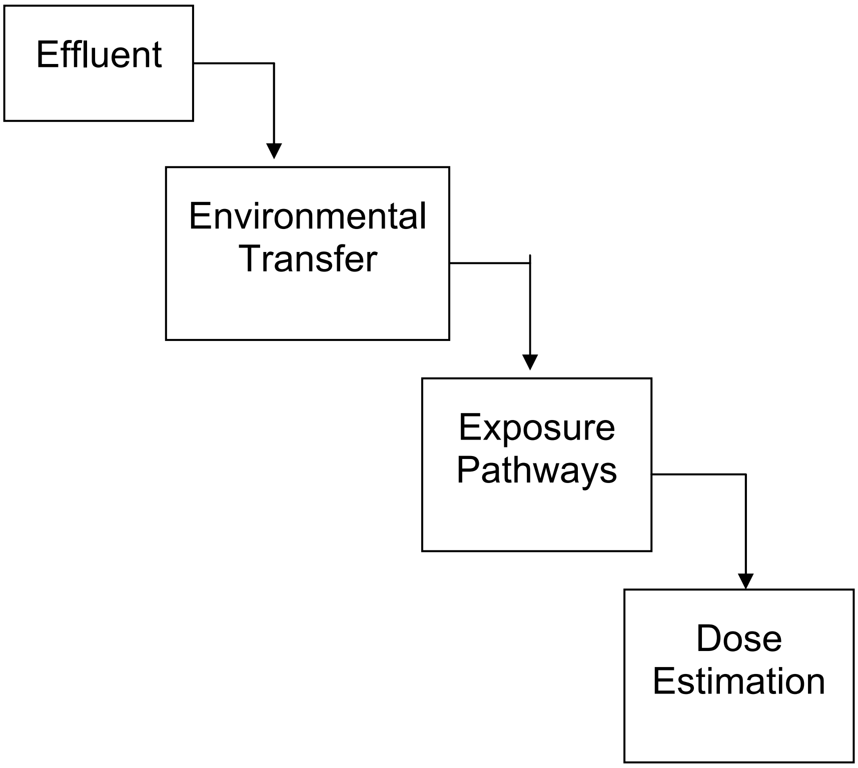

Figure 1.1 General stages in the assessment of the radiological consequences of releases of radionuclides to the environment

2 BASIC CONCEPTS, ASSUMPTIONS AND QUANTITIES

In this chapter some of the basic concepts and assumptions adopted in this methodology and PC-CREAM 08 are outlined and the more important radiological quantities are defined.

2.1 Radiation effects

In radiological protection a number of dose and risk related quantities associated with the effects of radiation are used. In Publication 103, ICRP reiterates that radiological protection deals with two types of harmful effect (ICRP, 2007). High doses will cause deterministic effects characterised by harmful tissue reactions and these are often as a result of acute exposure above a threshold value. Both high and low exposures may cause stochastic effects, which are characterised by cancers or heritable effects. Stochastic effects may be observed as a statistically detectable increase in the incidence of these effects occurring long after exposure.

At levels of dose and dose rate typically encountered from controlled sources consideration can be limited to stochastic effects. A general assumption is made that over a limited range of doses there is a simple proportional relationship between dose and probability of effect. This implies that stochastic effects can never be eliminated, and that their occurrence can only be minimised.

2.2 Individual quantities

The fundamental dosimetric quantity in radiological protection is the **absorbed dose, D,** the energy absorbed per unit mass. This quantity has the unit joules per kilogram, which is given the name gray (Gy). Absorbed dose was traditionally defined at a point in tissue but ICRP, in Publication 60 (ICRP, 1991) and subsequently in publication 103 (ICRP, 2007), has stated that it can also be averaged over a tissue or organ for the purpose of risk estimation. Based on the assumption that the dose response is linear, the average absorbed dose in a single tissue or organ is used as an indicator for the incidence of stochastic effects in that tissue or organ.

Absorbed dose is a physical quantity which does not fully reflect the possible biological effects of ionising radiation. These effects depend not only on the energy deposited per unit mass of tissue, but also on the type and energy of radiation and the tissue affected. ICRP (ICRP, 1991), has therefore, established radiation and tissue weighting factors and two additional dose quantities. The term **equivalent dose H_T**, has been applied to the tissue - or organ - averaged absorbed dose modified by radiation weighting factors, w_R. The unit of equivalent dose is the joule per kilogram and has the name sievert (Sv). H_T is defined (ICRP, 1991) by:

$$H_T = \sum_R w_R D_{T,R}$$

Where w_r is the radiation weighting factor for radiation R and $D_{T,R}$ is the mean absorbed dose in the tissue or organ T due to radiation R.

ICRP has selected the value of the radiation weighting factor for a specified type and energy of radiation to be representative of values of the relative biological effectiveness of the radiation in inducing stochastic effects at low doses. The values recommended by the Commission (ICRP, 1991) for w_R are given in Table 2.1. It is noted that in their most recent recommendations (ICRP, 2007) ICRP has revised these weighting factors but as dose coefficients calculated using these new factors have yet to be published by the Commission, the values given in Table 2.1 and included in PC-CREAM 08 are those from ICRP 1991.

The second additional dose quantity established by ICRP is the **effective dose, E.** This refers to the equivalent dose further modified by tissue weighting factors, w_T. The effective dose has a unit of joule per kilogram and is given the name sievert. It is given by:

$$E = \sum_T w_T \sum_R w_R D_{T,R}$$

Where $D_{T,R}$ is the mean absorbed dose in the tissue or organ T due to radiation R and w_T and w_R are the tissue weighting factors and the radiation weighting factors, respectively.

The probability of occurrence of a stochastic effect in a particular tissue can be taken as proportional to the equivalent dose in the tissue. However the constant of proportionality and the associated health detriment differ for the various tissues of the body, with detriment being defined by ICRP (ICRP, 1991; ICRP, 2007) as the total harm to health experienced by an exposed group and its descendents as a result of the group’s exposure to a radiation source. Detriment is a complex concept and for the derivation of effective dose ICRP (ICRP, 1991) has used an aggregated representation of detriment which includes four components:

- the probability of attributable fatal cancer
- the weighted probability of attributable non-fatal cancer
- the weighted probability of severe hereditary effects
- the relative length of life lost.

The tissue weighting factors, w_T, are defined by ICRP to assess health detriment arising from the irradiation of different organs and tissues. The recommended values are given in Table 2.2. Again, although ICRP has published revised tissue weighting factors in its most recent recommendations (ICRP, 2007) these are not included here or in PC-CREAM 08 as new dose coefficients have yet to be published by the Commission. The values in Table 2.2 have been developed for a reference population of equal numbers of both sexes and a wide range of ages. In the definition of effective dose the tissue weighting factors apply to workers, to the general public, and to either sex.

The absorbed dose from external irradiation is delivered at the same time as the tissue is exposed to the radiation field. However, for internal irradiation from incorporated radionuclides the total absorbed dose will be delivered over time as the radionuclides remain in the body and decay. The time distribution of the absorbed dose rate will vary with the radionuclides, their form, mode of intake, and biological behaviour. ICRP (ICRP, 1991; ICRP, 2007) has defined the **committed equivalent dose, H_T (τ),** to be the time integral of the equivalent dose rate. τ is the integration time in years following the intake and if not specified is to be taken to be equal to 50 years for adults and from intake to age 70 years for children. If the committed equivalent doses to the individual tissues resulting from an intake are multiplied by the appropriate weighting factors, $w_{t,}$ and then summed, the result is the **committed effective dose E(τ).** This quantity is taken to represent the total health detriment to an individual and their progeny from an intake of radioactive material, including the risk from irradiation in subsequent years resulting from the intake.

2.3 Collective quantities

ICRP has stated that collective quantities can be thought of as representing the total consequences of the exposure of a population or group (ICRP 1991; ICRP 2007). However, their use in this way should be limited to situations in which the consequences are truly proportional to both the dosimetric quantity and number of people exposed. In its most recent recommendations ICRP (ICRP, 2007) only recommends the use of one collective dose quantity, that of the **collective effective dose (man Sv)**, which is defined as the sum of all individual effective doses over the time period or during the operation being considered. The unit of the collective effective dose is the joule per kilogram and this quantity is given the name man Sievert.

The collective effective dose due to individual effective dose values between E_1 and E_2 from a specified source within a specified time period ΔT is defined as:

$$S(E_1, E_2, \Delta\mathrm{T}) = \int_{E_1}^{E_2} E \left[\frac{dN}{dE} \right]_{\Delta T} dE$$

Which can be approximated to:

$$S = \sum_i E_i N_i$$

Where E_i is the average effective dose for a subgroup I, and N_i is the number of individuals in this subgroup. The time period and number of individuals over which the effective doses are summed should always be specified.

If the integration is not over infinite time, the quantity is described as being truncated at a defined time. These truncated quantities provide an indication of the temporal distribution of the health detriment in the exposed population and its descendants. This time distribution enables the distinction to be made between a collective effective dose delivered over, say, a million years and the same dose delivered over a very short period. The levels of individual risk would be very different in these two cases.

It is sometimes useful to express results in terms of doses to a hypothetical average individual in a population. This is the per caput dose and can be defined for either equivalent dose or effective dose. It is obtained by dividing the collective dose in a population by the number of individuals in the population, or more directly by calculating the average exposure to, or intake of, the radioactivity released from the source and hence the average dose. It only rarely represents the dose to an actual individual.

2.4 Assumptions adopted in the methodology and PC-CREAM 08

In estimating collective doses, assumptions have to be made on the magnitude and habits of the exposed population and its descendants and their variation with time. Predictions of population trends are available but they are very uncertain, with the uncertainty increasing over time. Bearing in mind the timescales over which integrations are to be performed in this study, introducing a time varying population does not seem justified. The collective doses evaluated here are based on the assumption that the magnitude of the population of the EU and the world remains constant over all time. It is further assumed that the habits of the population (for example, dietary intake) also remain the same. In CREAM, one further major assumption is made in the estimation of collective doses; the whole population is assumed to be adults. For example, in estimating collective doses from the inhalation and ingestion of radionuclides the whole population is assumed to have the same, adult, intakes and metabolism. Incorporating age related factors is likely to have little effect on the resulting collective dose so this simplifying assumption seems justified. However, data for intakes and dose per unit intake are provided for different age groups to enable individual doses to be determined as a function of age.

Information is given in this report to enable individual and collective doses to be integrated over various times, including a set of pre-defined default times although in PC-CREAM 08 the user is also able to set their own times over which to evaluate the dose. These default times (l y, 50 y, 100 y, 500 y and infinity) are somewhat arbitrary but were selected to give a reasonable representation of the temporal distribution of doses and detriment. Also, the truncated time integrals of dose to time, t, for the annual release of a radionuclide can be equated to the annual dose in the year, t, for a release that has been continuous over that time. Thus, for discharges that are continuous for 50 years (an estimate of the lifetime of nuclear installations) or 500 years (a speculative estimate of the continuing use of nuclear energy), the annual collective or individual doses in these years can be readily estimated from the quantities evaluated in this study.

The models used within PC-CREAM 08 will calculate the activity of the released radionuclide explicitly, and for most models will then make assumptions with regards to the activity of radioactive progeny. For example, for the models DORIS, FARMLAND and RESUS any progeny reaching secular equilibrium within 1 year are not modelled explicitly but are considered to be present with the same activity as the parent radionuclide when the dose is estimated. Within PLUME this time is reduced to 3 minutes, which enables important short-lived radionuclides to be modelled explicitly. For FARMLAND, RESUS and PLUME the activity of the first progeny not in secular equilibrium is modelled explicitly; any other members of the radioactive decay chain are not considered to be present. In DORIS the full decay chain can be considered, however, only those progeny not in secular equilibrium with their immediate parent are modelled explicitly while those in secular equilibrium are assumed to have the same activity as their parent, as mentioned above. For GRANIS the activity of all radionuclide decay chain members are calculated explicitly and included within this are the effects of migration within soil. More detail on the treatment of radioactive progeny within the models of CREAM is included in the following sections describing the models.

2.5 References

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TABLE 2.1 Radiation weighting factors

Type and energy range	W_R
Photons, all energies	1
Electrons and muons (all energies)	1
Neutrons	
<10 KeV	5
10 - 100 KeV	10
100 KeV - 2 MeV	20
2 MeV - 20 MeV	10
>20 MeV	5
Protons, other than recoil protons, energy >2 MeV	5
Alpha particles, fission fragments, heavy nuclei	20

Taken from ICRP Publication 60 (ICRP, 1991)

TABLE 2.2 Tissue Weighting Factors

Tissue or organ	Tissue weighting factor[1], w_T
Gonads	0.20
Bone marrow (red)	0.12
Colon	0.12
Lung	0.12
Stomach	0.12
Bladder	0.05
Breast	0.05
Liver	0.05
Oesophagus	0.05
Thyroid	0.05
Skin	0.01
Bone surface	0.01
Remainder	0.05[2,3]

Notes

(1) The values have been developed by ICRP (see ICRP, 1991) from a reference population of equal numbers of both sexes and a wide range of ages. In the definition of effective dose they apply to workers, to the whole population, and to either sex.

(2) For purposes of calculation, the remainder is composed of the following additional tissues and organs: adrenals, brain, upper large intestine, small intestine, kidney, muscle, pancreas, spleen, thymus and uterus. The list includes organs which are likely to be selectively irradiated. Some organs in the list are known to be susceptible to cancer induction. If other tissues and organs subsequently become identified as having a significant risk of induced cancer they will then be included either with a specific w_T or in this additional list constituting the remainder. The latter may also include other tissues or organs selectively irradiated.

(3) In those exceptional cases in which a single one of the remainder tissues or organs receives an equivalent dose in excess of the highest dose in any of the twelve organs for which a weighting factor is specified, a weighting factor of 0.025 should be applied to that tissue or organ and a weighting factor of 0.025 to the average dose in the rest of the remainder as defined above.

3 RELEASES TO THE ATMOSPHERE

3.1 Introduction

The discharge of radioactive materials into the atmosphere may lead to the exposure of humans via a number of pathways. Airborne radionuclides can give rise to exposure by two principal routes: external irradiation by photons and electrons emitted as a result of the radioactive decay process and internal irradiation following their inhalation. The radionuclides in the plume will gradually be depleted by the processes of deposition onto underlying surfaces and radioactive decay. The wet and dry deposition of radionuclides leads to their transfer in the terrestrial environment where they can continue to expose human populations. Deposited radionuclides may again be available for inhalation as a result of resuspension, caused by wind-driven or man-made disturbance. The radioactive decay of deposited radionuclides will lead to external exposure from photons and electrons. The deposition onto vegetation and soils leads to the transfer of radionuclides into human foodstuffs, the consumption of which will lead to internal exposure. Another possible internal exposure pathway is the inadvertent ingestion of contaminated soils although this pathway is not currently included in PC-CREAM 08. Therefore, an assessment of either collective or individual dose must not only consider the dispersion of radionuclides as they are blown downwind and their subsequent deposition but also their fate once they are in the terrestrial environment.

The methodology described here includes models and parameter values for all the transfer processes and exposure pathways included in PC-CREAM 08. These models are generally similar to those described in the previous report (Simmonds, Lawson and Mayall 1995), although the resuspension and foodchain models have been updated and revised. As noted earlier, the previous report, RP 72 (Simmonds, Lawson and Mayall, 1995) contains additional information on models and data that are not currently included in PC-CREAM 08).

Each model has been developed and applied generically in order to derive a matrix of results which can then be adapted for a release of radionuclides from any location within the EC. This involves the application of site-specific parameters such as meteorological conditions and radionuclides discharged. In this methodology, concentrations of activity in air, deposition rates and external dose rates from the airborne radionuclides are calculated as a function of distance for a unit release rate of selected radionuclides, for selected atmospheric conditions. The concentrations of activity in air and external dose rates may be combined with the spatial distribution of the EU population with respect to the site in question to estimate collective doses via inhalation and external exposure from the plume. In a similar manner the transfer through the terrestrial environment is evaluated for unit deposit of selected radionuclides; the time-dependent variations of the external dose rates above the surface, the resuspended concentrations of activity in air, and the time-integrated concentration in various foodstuffs are determined. Again, the combination of the matrices of these values with the site-specific distributions of population and agricultural production around a site enables the calculation of collective dose via these routes to the population of the EU.

For the calculation of individual doses within about 30 km of the site more site-specific factors may need to be considered. Information is needed on the locations of individuals and the points at which any local foods, eaten by those individuals, are grown. Activity concentrations in the relevant environmental media and external dose rates are then predicted using the same models as used for collective doses. Habit data for the individuals, such as consumption rates and occupancies, are also required to estimate the intakes of radionuclides and the external dose respectively (see Chapter 7 of this report).

Figure 3.1 illustrates schematically the main stages in an assessment of collective or individual dose following discharges to atmosphere.

The radionuclides which have been included in this methodology are listed in Table 3.1. They were selected on the basis of their potential significance. This list is not exhaustive, other radionuclides can be treated just as readily by this methodology. Not all radionuclides released to atmosphere need to be considered when evaluating transfer through the terrestrial environment. This is either because they are too short-lived or because of their chemical properties, for example, the inert or noble gases do not deposit.

The distance over which radionuclides may be transported in the atmosphere depends on many factors such as radioactive half-life, physico-chemical form, weather conditions and deposition processes. In this study the dispersion of the released material is modelled over a distance of 3000 km which is in excess of the distance of the extreme points in the EC. In the majority of cases most radionuclides are removed from the atmosphere well within this distance. However, some radionuclides, due to their inert nature and long half life or rapid exchange between the atmosphere and other sections of the environment, are transported over much greater distances and subsequently become globally distributed. Separate models have been developed to account for this situation (see Chapter 5 the Global circulation chapter).

3.2 The atmospheric dispersion of radionuclides

3.2.1 Atmospheric dispersion models

Following the release into the atmosphere of radioactive materials their subsequent dispersal downwind will depend on their physical properties and the weather conditions. The estimation of dispersion in the atmosphere is commonly approached by solving the diffusion-transport equation. Many models have been developed for this purpose.

Lagrangian Puff models and *Eulerian grid* models (Simmonds, Lawson and Mayall, 1995) provide a detailed representation of the physical processes of turbulent diffusion. However, they are considered inappropriate and too demanding of computer resources for inclusion in a general methodology of this kind. Moreover this methodology is concerned with long duration releases where simplifying assumptions can be made. For example, diffusion in the direction of the wind can be ignored in this case. *Plume models* are particularly useful for predicting dispersion under these conditions the best known of which is the semi-empirical *Gaussian plume model*. This model is still widely adopted because it is relatively simple to apply, and the parameter values are related to readily measurable quantities such as wind speed and cloud cover. It is also considered suitable where the end-points of the calculations are long-term average or time-integrated concentrations in air, as is normally the case in the assessment of dose from continuous releases. The basic model has changed little since its development although it has been modified and adapted to cope with a greater variety of release scenarios than originally intended. It is based on ideas introduced by Sutton (1932) and developed by Pasquill (1961) and Gifford (1976) and is used where the atmospheric boundary layer is characterised in a relatively simple way according to surface meteorological measurements.

The basic model was developed for dispersion over land of neutrally buoyant releases from isolated stacks in flat terrain. The inclusion of individual dose assessments in this methodology and the need to estimate dispersion over relatively short distances may require the consideration of the possible influence of site-specific factors. Such factors include the influence of turbulent building wakes, plume rise and coastal conditions. The simple model implemented in CREAM does not include these factors but they may be accounted for by applying the recommendations given in Jones (1983) or as discussed in RP 72 (EC, 1995).

3.2.2 The Gaussian plume dispersion model

The concentration in air of a radionuclide of long radioactive half-life, X(x,y,z) is given in the simplest form of the Gaussian plume model as:

$$X(x,y,z) = \frac{Q_o}{2\pi\sigma_y\sigma_z u} \exp\left[-\left(\frac{y^2}{2\sigma_y^2} + \frac{(z-h_e)^2}{2\sigma_z^2} \right) \right] \qquad (3.1)$$

where X(x,y,z) is the activity in air concentration at the point (x,y,z) in Bq m^{-3}. The origin of the co-ordinate system is at ground level beneath the discharge point.

x	is the downwind distance (m)
y	is the cross wind distance from the centre line of the plume (m)
z	is the height above ground for which the concentration is calculated (m)
σ_y	is the standard deviation of the horizontal Gaussian distribution (m) (or horizontal dispersion coefficient)
σ_z	is the standard deviation of the vertical Gaussian distribution (m) (or vertical dispersion coefficient)
Q_o	is the release rate (Bq s^{-1})
u	is the mean windspeed (ms^{-1}) and
h_e	is the effective release height (m)

This equation, which was derived for a gas, is assumed to be equally applicable when considering aerosols. In the derivation of equation (3.1) diffusion in the downwind direction is ignored as it is negligible compared to transport by the wind for releases lasting a finite time.

The concentrations obtained from equation (3.1) are applicable to releases which are short compared with the time taken for the direction of the wind to change. For releases of longer duration the horizontal spread of material is governed by fluctuations in the wind direction. For a continuous release in which the meteorological conditions are assumed to remain constant and the windrose assumed to be uniform, equation (3.1) can be rewritten as:

$$\overline{X}(x,z) = \frac{Q_o}{2\pi x\sqrt{2\pi}\,\sigma_z u_s}\exp\left[-\left(\frac{(z-h_e)^2}{2\sigma_z^2}\right)\right] \qquad (3.2)$$

where $\overline{X}$ is the mean activity concentration in air at the point (x,z) in Bq m^{-3} and u_s is the wind speed at the height of the plume or the effective release height.

If the effective release height, h_e, is less than 10 m then the windspeed at 10 m, u_{10}, should be used instead of u_s. The dependence in the cross wind direction (y) is removed owing to the assumption of a uniform windrose. In reality the meteorological conditions will not remain constant during a prolonged release of radioactivity. The method of including this in the calculation of annual average concentrations is described in section 3.2.3.

3.2.2.1 Reflection from the ground and from the top of the mixing layer

When material is discharged from an elevated source, the plume will disperse and eventually reach the ground. On reaching the ground the plume is reflected and effectively dispersed back up into the atmosphere. To take account of reflection of the plume from the ground, the mean activity concentration in air is modified from that in equation (3.2), as given by:

$$\overline{X}(x,z) = \frac{Q_o}{2\pi x\sqrt{2\pi}\,\sigma_z u_s}\left[\exp\left(-\frac{(z-h_e)^2}{2\sigma_z{}^2}\right)+\exp\left(-\frac{(z+h_e)^2}{2\sigma_z{}^2}\right)\right] \qquad (3.3)$$

The limit to the layer in which mixing takes place in the atmosphere will occur at varying heights depending on the conditions and arise from changes in temperature gradient. Where a finite mixing or boundary layer exists the dispersed material is trapped between the top of this layer and the ground. Reflections in this case occur both on the ground and at the top of the mixing layer as illustrated in Diagram 3.1.

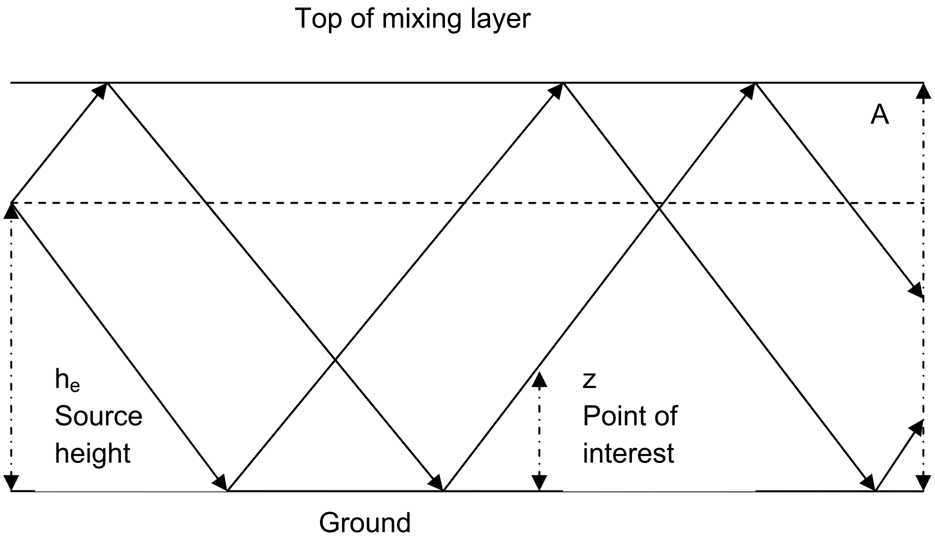

Diagram 3.1 Model for determining the cloud concentration under conditions of limited vertical mixing

Without a mixing layer of finite depth the plume would continue to expand in the vertical direction. The effect of introducing reflections is that the airborne concentration is obtained by summation of contributions from many points over the Gaussian. The effect can be simulated by considering virtual sources at a series of heights. From Diagram 3.1 these are:

$$\text{a)} -h_e \quad \text{b)} 2sA \pm h_e$$

where A is the depth of the mixing layer (m) and s = 0, 1, 2, 3 etc.

In addition the primary dispersion due to a source at effective height, h_e, must be included; this, together with the reflected term from a virtual source at a height, $-h_e$, corresponds to the terms in equation (3.3) for dispersion including ground reflection. Thus for a finite mixing layer, the mean concentration is given by

$$\overline{X}(x,z) = \frac{Q_o}{2\pi x \sqrt{2\pi}\,\sigma_z u_s} \sum_{s=0}^{\infty} \exp\left[-\left(\frac{(2sA \pm h_e \pm z)^2}{2\sigma_z{}^2}\right)\right] \tag{3.4}$$

When s = 0 only the positive z term is required. In practice sufficient accuracy is obtained if the series is truncated after the s = 1 term.

In general this series converges rapidly and can be summed to any prescribed accuracy. At large distances downwind, after multiple reflections or when the value of the vertical dispersion coefficient becomes greater than the depth of the mixing layer, the vertical concentration profile of activity essentially becomes uniform between the ground and top of the mixing layer. Equation 3.4 can then be simplified to the following:

$$\overline{X}(x,z) = \frac{Q_o}{x\,2\pi\,u_s\,A} \tag{3.5}$$

3.2.2.2 *The choice of input parameters for the model*

In order to apply the Gaussian plume model it is necessary to categorise the meteorological conditions into groups, so that the dispersion rate is similar for all conditions within each and then to ascribe appropriate values of the dispersion coefficients (σ_z and σ_y) to those conditions. These groups are generally referred to as stability categories. Atmospheric turbulence is generated by the heating or cooling of the underlying surface and friction forces as air moves over the surface. The grouping schemes need to consider these effects. Because the σ_y term is absent from the form of the equation which is used for calculation of annual average concentrations from continuous releases the following focuses on the vertical dispersion coefficient. Within the EC the atmospheric stability is generally derived in terms of either the Pasquill (1961) or Doury (1976) system.

3.2.2.2.1 *Pasquill/Smith/Hosker scheme*

The primary scheme recommended in this methodology is that of Smith (1973) which is based on the original scheme devised by Pasquill (1961). Pasquill based his scheme on a range of experimental observations and suggested values for dispersion parameters to be used in the Gaussian plume model for six stability categories, which he designated A to F in order of increasing atmospheric stability. The procedure for classifying a given set of conditions suggested by Pasquill was qualitative. However, the Smith scheme is quantitative and takes into account a number of factors including the effect of ground roughness, wind speed over a wide range and the sensible heat flux in the lower levels of the atmosphere (ie, the amount of heat per unit horizontal area passing between the air and the underlying surface as a result of air-surface temperature differences, often caused by net solar heating of the surface during the day or radiative cooling at night). Smith defined the stability in terms of a continuous variable P rather than the discrete categories used in the Pasquill scheme. Values of the stability index of 0.5, 1.5, 2.5, 3.6, 4.5 and 5.5 may be taken as equivalent to the Pasquill categories A to F respectively. Smith extended the scheme to include very stable conditions with a P value of 6.5 otherwise known as category G (Smith, 1973).

Smith (1973) solved the diffusion equation numerically over a range of atmospheric conditions and wind speeds, and calculated values for σ_z, so that a Gaussian plume model gave the same prediction of concentration as a more complex model. This allowed a further parameter to be introduced into the calculations, the ground roughness length. The ground roughness length (z_o) is a measure of the mechanical turbulence introduced into the atmosphere by the roughness of the underlying surface and is approximately equal to 10% of the average height of the surface protrusions. Table 3.2 gives values recommended for different terrain types as well as representative values of n for use in equation 3.10. To facilitate numerical analysis Hosker (1974) fitted equations to Smith's results which were presented graphically out to distances of 100 km. The equation has the form

$$\sigma_z = \frac{a\,x^b}{1+c\,x^d} F(z_o, x) \tag{3.6}$$

where $F(z_o,x)$ is a roughness correction factor given by

$$F(z_o, x) = \ln\left(f\,x^g \left[1 + \frac{1}{hx^j} \right] \right)\ ,\ z_0 > 0.1\,\text{m} \tag{3.7}$$

$$F(z_o, x) = \ln\left(f\,x^g \left[\frac{1}{1+hx^j} \right] \right)\ ,\ z_0 \leq 0.1\,\text{m} \tag{3.8}$$

The values for the coefficients a, b, c and d in equation 3.6 and for the coefficients f, g, h and j in equations 3.7 and 3.8 are given in Table 3.3. This representation for σ_z is used in this study and assumed to apply over all distances considered.

3.2.2.2.2 Doury scheme

From a review of experimental data Doury (1976) derived a relationship between the vertical dispersion coefficient and the travel time, t. The relationship has the form

$$\sigma_z = (Mt)^k \tag{3.9}$$

Two variations of σ_z with time are considered and categorised as normal and poor diffusion; combining these with different wind speeds a range of dispersion conditions may be accommodated. The values of M and k for a variety of travel times in each diffusion category are given in Table 3.4. For each diffusion category three wind speeds are considered (see Table 3.4).

3.2.2.2.3 Wind speed and boundary layer depth

The wind velocity in the mixing layer increases with height above the ground because of the reduced effect of the earth's frictional forces. The increase, which is initially rapid, gradually slows until the wind attains the geostrophic wind velocity at a height at which the earth's surface no longer exerts an influence.

Different wind speed quantities are included in the variations of the model presented. Equation 3.1 requires the input of the mean wind speed which may be that measured at a height of 10 m, a height commonly used for meteorological measurements (Clarke, 1979). This is because the values of σ_y derived for use with equation 3.1 correspond with the use of the wind speed at that height, and because the product of wind speed and σ_y is almost independent of height. The disappearance of σ_y from equation 3.3 means that the wind speed adopted in this equation does not need to be that measured at 10 m. Studies have suggested that the best fit between predicted and measured concentrations is obtained using the wind speed at the height of the release. The formula is generally used with the wind speed at 10 m height for releases at heights lower than this.

For the purpose of relating the wind speed at the 10 m reference height, u_{10}, and the wind speed at any other height u(z) the following power law may be used (Clarke, 1979):

$$u(z) = u_{10}(z/10)^{n} \tag{3.10}$$

where u(z) is the wind speed at height z. The shape of the profile depends on n (see Table 3.2) which is a function of ground roughness (ie, the physical features on the earth’s surface) and to some extent atmospheric stability.

In most cases the boundary layer depth is unlikely to be available. Table 3.5 gives default values for this parameter, together with wind speeds for each stability category for the Pasquill and Doury schemes for occasions when meteorological data are not available.

Measurements of wind speed over extended periods are likely to include periods of calms ($u_{10} < 2\ \mathrm{ms}^{-1}$). The Gaussian plume model was not designed to be applied to these conditions therefore calms should be grouped into the lowest wind velocity class and assigned to the wind direction sectors with a distribution corresponding to that of the lowest velocity class.

3.2.2.2.4 Long range dispersion

The model assumes that the meteorological conditions present at the release point remain constant during the entire travel time of the plume. When modelling long range dispersion in the calculation of collective doses this assumption becomes less valid due to the increased likelihood of changes in both wind direction and stability category en-route. The mean duration of a particular stability category is only a few hours. As material travels along it disperses vertically and will eventually fill the boundary layer. Changes in the vertical distribution subsequently reflect changes in the depth of the boundary layer caused by stability changes.

Changes in stability before the material has spread uniformly throughout the boundary layer affect the subsequent dispersion in two ways. Firstly, the depth of the boundary layer changes allowing material to disperse throughout the new boundary layer. Secondly, the change of stability alters the rate at which the plume increases in size. In addition, the increase in wind speed with height will affect the plume as it grows to fill the boundary layer.

Changes in wind direction could have only a small effect on the annual average concentration at great distances in a given sector for two reasons. Firstly, directional changes resulting in the transfer of material between sectors will tend to balance each other out. Secondly, concentrations at great distances are largely determined by neutral conditions in which the trajectories are likely to be approximately straight. The average trajectory length to points within about 1000 km of a release has been estimated to be only 15% greater than the straight line distance to the points (Smith, 1979).

The differences in results of a range of models which used a variety of schemes for incorporating stability changes during plume travel were considered by a UK Working Group on Atmospheric Dispersion (Jones, 1981a). The Group subsequently recommended that long range dispersion should be evaluated assuming that all releases occur in neutral stability, but allowing for a range of wind speeds. It is the wind speed classes which define the set of conditions over which a weighted sum is required, as described in section 3.2.3. The Group also recommend that a boundary layer depth of 800 m be assumed for all conditions. This represents a compromise between the complexity and cost of using models which incorporate stability changes through the tracking of plume segments coupled with extensive meteorological databases, the accuracy required from a model, and the need to assume that the original conditions will not persist for extended travel times.

3.2.2.3 Removal processes

The concentrations derived from equation 3.4 apply to the dispersion of inert, long lived material (eg, krypton-85) which is not removed significantly from the plume as it travels downwind. A number of processes may act to reduce further the concentrations of discharged activity, in particular radioactive decay and dry and wet deposition. These processes are most readily taken into account by modifying the initial source strength, Q_o, in equation (3.4) to allow for depletion.

(a) Radioactive decay

Radioactive decay will reduce the concentrations of a radionuclide as it disperses downwind; the modified concentration can be obtained by substituting a modified source strength, Q_oR_p, into equation (3.4) where

$$R_p = \exp\left[-\lambda_p \frac{x}{u_s}\right] \qquad (3.11)$$

where λ_p is the radioactive decay constant of the radionuclide (s^{-1}).

Progeny will grow into the plume with the decay of the parent radionuclide and the concentration of the progeny can be obtained by substituting Q_oR_d for Q_o in equation (3.4) where

$$R_d = \frac{\lambda_d}{\lambda_p - \lambda_d}\left[\exp\left(-\lambda_d \frac{x}{u_s}\right) - \exp\left(\lambda_p \frac{x}{u_s}\right)\right] \qquad (3.12)$$

where λ_d is the radioactive decay constant of the daughter (s^{-1}) and λ_p that of the parent.

In PC-CREAM 08 only the first progeny (or first two if branching occurs immediately) that grows into the plume is modelled. If the progeny are very short lived they are ignored as they are assumed to be in secular equilibrium with their parent radionuclide. The first progeny modelled is the progeny which has not reached secular equilibrium 3 minutes after the release. This short time period ensures that potentially important progeny such as Rh-106 and Ba-137m are modelled explicitly.

(b) Wet deposition

Radioactive particulates and gases in the plume may be removed by various types of precipitation mechanisms. There are two distinct processes by which this removal occurs: rain falling through the plume, which is termed washout: removal of activity incorporated in the rain cloud, known as rainout. The model considered in this methodology includes both of these processes. Activity can also be removed by the action of snow; however, for routine releases this can be adequately described by the model for the removal by rain.

Rain falling through a plume removes material throughout the whole of the plume volume. The wet deposition rate is therefore a function of the total activity throughout the depth of the plume being rained on rather than the activity concentration in air at ground level. The wet deposition rate during rainfall can be calculated using the washout coefficient, (or more accurately the wet removal coefficient), defined as the fraction of material within the plume removed by rain in unit time. Washout is affected by the size distribution of the rain drops as well as the properties of the diffusing material. Rainout is influenced by condensation processes within the cloud and by the rate at which diffusing material is swept into a rain-cloud. Due to the difficulty in separating the contributions from rainout and washout the values of washout coefficient incorporate the effects of both processes. Its use at short distances will tend to overestimate the deposition rate, as rainout does not occur while the plume is contained in the lower levels of the boundary layer. A review of washout coefficients has been carried out by Underwood on behalf of the UK Atmospheric Dispersion Modelling Liaison Committee (ADMLC, 2001).

Calculations suggest that, at a typical rainfall rate of 1 mm h^{-1}, the washout coefficient of 1 μm particles is about 4 10^{-5} s^{-1} while that for 20 μm particles it is about 3 10^{-4} s^{-1} (Chamberlain, 1953; Engelman, 1968). Experiments to measure washout coefficients are reviewed by Slinn (1978). Suggested values for use in this methodology are given in Table 3.6 for four types of material for a rainfall rate of 1 mm h^{-1}. It is recommended that the default value for the washout coefficient be 10^{-4} s^{-1} for 1 μm particles but the user may wish to choose a value from the appropriate range after carrying out a sensitivity study. Noble gases are assumed not to undergo washout; tritium and carbon-14 are assumed to be returned to the atmosphere in a timescale that is short compared to their half-lives, therefore no allowance is made for washout.

It is considered adequate to ignore any variation of the washout coefficient with rainfall rate in routine release assessments as the variation is not large. The washout coefficient has been found to be proportional to rainfall rate to a power between 0.5 and 1.0 (Engelman, 1968; Brenk and Vogt, 1981).

The total amount of material Q′ remaining in a plume, subject to continuous rainfall at a constant rate, is given by

$$\frac{\mathrm{d}Q'}{\mathrm{d}t} = -\Lambda Q' \qquad (3.13)$$

which gives $Q' = Q_o \exp[-\Lambda t]$ where Λ is the washout coefficient and t is the time from the start of the rain in seconds.

The interaction between the plume and rain is in reality very complex, with the plume subject to continuous rain being only one of a range of possible sequences. This is discussed in more detail in RP 72 (Simmonds, Lawson and Mayall, 1995)

Activity concentrations in air are then obtained using equations 3.2 – 3.4 as appropriate, but replacing Q_o by $Q'(t)$ in category C and D or Doury normal conditions.

The wet deposition rate per unit area D_w is then found by:

$$D_w = \frac{\Lambda Q'(t)}{x\,\alpha\,u_s} \tag{3.14}$$

where x is the distance from the release point, α is the angular width of the sector (radians) and u_s is the mean wind speed.

(c) Dry deposition

Dry deposition is a complex process by which material is removed from the plume by impaction with the underlying surface or obstacles on it, such as vegetation. The rate at which material is deposited from the plume will depend on the nature of the airborne material, the underlying surface and the state of the atmosphere and can be estimated using the concept of a deposition velocity, V_g. The deposition velocity was defined by Chamberlain (1953) as the ratio of the amount of material deposited on the surface per unit area per unit time, to the air concentration per unit volume at the surface.

The dry deposition rate or its time integral, D_D, is given by

$$D_D = V_g C \tag{3.15}$$

where C is the activity concentration in air at ground level or its time integral, respectively.

Table 3.6 gives suggested values for the deposition velocity applicable for deposition onto grass in neutral conditions. The values quoted for reactive gases are intended primarily for elemental iodine and are for use at short distances from the release. A number of reviews of deposition velocity have been undertaken, the most comprehensive being those of Slinn (1978) and Sehmel (1980a). Garland (1980) has described the variation of deposition velocity with surface type and atmospheric conditions. For reactive gases the deposition velocity varies little with vegetation height and it is likely that deposition in the urban environment will be no higher than that on pasture (Jones, 1983). Information on the deposition of particles to tall vegetation (other than forest canopies) and in urban areas is sparse and the limited evidence suggests that deposition is not much greater than on grassland. The evidence for smooth surfaces indicates that deposition may be somewhat lower than to grass. The variation of deposition velocity with atmospheric conditions is largely determined by its variation with wind speed. Therefore the deposition velocity in low wind speed, stable conditions could be lower than the values given in Table 3.6. A review of dry deposition velocities has been carried out by Underwood on behalf of the UK Atmospheric Dispersion Modelling Liaison Committee (ADMLC, 2001).

A single value of 10^{-3} m s^{-1}, which is representative of 1 µm particles, is used in this methodology for all radionuclides, except for noble gases (which are assumed not to deposit) and iodine. The deposition velocity for organic iodine is much lower, for which a representative value of 10^{-5} m s^{-1} is adopted whilst the value for inorganic iodine is higher, for which a value of 10^{-2} m s^{-1} is used. No allowance is made for the deposition of tritium and carbon-14 for the reasons described in the section on wet deposition.

Where the plume is being depleted by dry deposition the activity concentration in air is given by equations 3.2 to 3.4 but the source strength, Q_o, is replaced by a modified source strength, Q^* (x), given by

$$Q^*(x) = Q_o \left\{ \exp\left[F_D(x)\right] \right\}^{Vg/u_s} \qquad (3.16)$$

where

$$F_D(x) = -\sqrt{\frac{2}{\pi}} \int_o^x \frac{1}{\sigma_z} \left\{ \exp[-\frac{h_e^2}{2\sigma_z^2}] + \exp[-\frac{(h_e + 2A)^2}{2\sigma_z^2}] + \exp[-\frac{(h_e - 2A)^2}{2\sigma_z^2}] \right\} dx \qquad (3.17)$$

while $\sigma_z(x) < A$

where

A is the height of the boundary layer (m) and h_e is the effective release height (m).

and $$F_D(x) = F_D(x_c) - \frac{x - x_c}{A} \qquad (3.18)$$

while $\sigma_z(x) \geq A$

and where x_c is such that σ_z (x_c) = A where A is the height of the boundary layer.

This model is known as the *source depletion model* and applies to a release of any duration (Jones, 1981b). The integrals specified in equation 3.17 cannot in general be evaluated analytically and must therefore be evaluated numerically.

When calculating deposition rate from equation 3.15 the activity concentration in air at ground level (z = o) should be used. Although deposition velocities are normally determined at a reference height of a few metres, this approximation is considered to be adequate.

The fraction of material remaining in the plume, Q^* (x)/Q_0, for two release heights in all stability conditions considered is given in Table 3.7 for a deposition velocity of 10^{-2} ms^{-1} representative of inorganic iodine. The table contains values calculated using the wind speed at the stack height. Values for other deposition velocities or wind speeds can be derived by scaling using the following equation

$$\frac{Q^*(x, V_g, u_s)}{Q_o} = \left\{ \frac{Q^*(x, V_{gT}, u_{sT})}{Q_o} \right\}^{V_g u_{sT}/V_{gT} u_s} \qquad (3.19)$$

where $Q^*(x,V_g,u)/Q_o$ is the fraction of activity remaining in the plume at distance x, for a deposition velocity V_g and wind speed u. V_{gT} and u_T are the values of deposition velocity and wind speed for which the tabulated values are calculated.

The consideration of radioactive decay for parent/daughter combinations where the deposition velocities are equal requires the substitution of Q_o in equation 3.16 with either Q_oR_p or Q_oR_d for parent and daughter respectively. If the deposition velocities of parent and daughter are different then a different approach is required.

The user should be aware of various limitations of the source depletion model. The model assumes that atmospheric turbulence is sufficient to maintain the plume's Gaussian vertical profile as material is removed at the ground. It therefore breaks down in conditions of low turbulence, ie in stable conditions categories F and G. The results of this model should be treated with caution if a significant fraction of material is predicted to be removed from the plume in these conditions. A further caveat is that the model makes no allowance for gravitational settling and should not be applied to particles of activity median aerodynamic diameter greater than about 10 μm.

(d) Simultaneous depletion of the plume by various mechanisms

When the plume is being depleted simultaneously by a combination of radioactive decay, wet and dry deposition, the source strength Q_o in equation 3.4 is replaced by the appropriate multiplication of the above factors.

3.2.3 Application of the dispersion and deposition models for continuous releases

The models for atmospheric dispersion and deposition described in the previous sections are applicable when the atmospheric stability can be assumed to remain constant during the release period. This assumption is not applicable when calculating annual average concentrations, and the models must be modified accordingly.

The activity concentration in air or deposition rate is assumed to be given by

$$X_i(r,z) = \sum_j f_{i,j}\, X_{i,j}(r,z) \qquad (3.20)$$

where $f_{i,j}$ is the frequency with which the wind blows into a particular sector in the direction of interest (i) in atmospheric stability category j

r is the distance from the source

and the summation is over stability categories.

The activity concentration in air in each category is calculated by a modified form of equation 3.3 in which the term $2\pi x$ is replaced by $r\theta$, with θ being the angular width of the sectors considered in radians.

3.2.4 Validation of models

Many studies have been published in which the validity and accuracy of the Gaussian plume model have been considered. These studies have been reviewed by, for example, Little and Miller (1979) or by Jones (1989). The likely accuracy of the form of the Gaussian plume model recommended by the UK Atmospheric Dispersion Modelling Working Group, which is used as the basis of the model given here, has been considered in a report by that Group (Jones, 1986). These studies show that the likely accuracy of a predicted concentration improves as the release duration increases. Annual average concentrations within a few kilometres of the site are likely to be predicted within a factor of 2. The uncertainty increases with increasing distance from the site, but concentrations within about 100 km are likely to be predicted within a factor of 4. A comparison of predictions from a Gaussian plume model and the UK Met Office's Lagrangian particle model, NAME, has been carried out by Lutman et al (2004) for a long term release. Differences between the two models are generally small compared to the expected precision of the models.

3.2.5 Limitations of the Gaussian plume model

The Gaussian plume model is semi-empirical in nature and was initially developed from experimental work carried out in the idealised conditions of flat terrain, uniform surface roughness and heat flux. The adaptation of the model for use in estimating dispersion in a wide range of conditions has inevitably led to various caveats being placed on its use. The following represents a summary of those factors affecting the use of the model of which the user should be aware.

The Gaussian plume model is not applicable in calm or near-calm conditions. The formula contains the reciprocal of the wind speed and therefore a zero wind speed cannot be used. Also, its use in conditions of a very low wind speed is not advised, because the wind speed and direction are very variable in these conditions so that a well-defined plume is unlikely to exist. Section 3.2.2.2.3 describes the procedure for including calms obtained from measured wind roses in the calculation of long-term average values.

The model recommended assumes a straight line trajectory and that the meteorological conditions prevalent at the discharge point at the time of release remain constant throughout the travel time of the plume. Implicit in this approach are the assumptions that wind speed and direction are uniform throughout the region of interest, that the surface roughness remains constant and that the terrain remains flat, ie topography does not affect the airflow. The assumption of flat terrain is often adequate when calculating short range dispersion from nuclear sites particularly from those situated by the coast. However, when estimating long range dispersion, and short range dispersion for sites situated in areas of complex terrain, the model may lead to considerable error. It is very difficult to give quantitative guidance on whether a particular site should be regarded as flat or complex terrain. Table 3.8 presents some simple criteria for neglecting orographic effects, ie, for assuming flat terrain. It can be seen that in neutral or unstable conditions the surrounding terrain can be regarded as flat if its gradient is less than about 1 in 10 and in stable conditions less than about 1 in 100. It should be noted that in very stable conditions, eg, category G, terrain effects could be observed with a slope as low as 1 in 1000 (Jones, 1986).

The application of the Gaussian plume model in a region of complex terrain in Spain is considered in (Simmonds, Lawson and Mayall, 1995). This considers the problems presented by the topographical and meteorological conditions prevalent on the Iberian peninsula together with an intercomparison of the Gaussian plume model with a more complex Lagrangian trajectory model for a site in south-west Spain.

In general the Gaussian model overpredicted concentrations at the majority of receptor points, although the concentration field did not follow the topography around the site. The ability of the complex model to describe the dispersion over the local terrain is dependent on the quality and quantity of relevant input data. The intercomparison highlighted the problems in obtaining adequate input data to run a complex model for an area of complex terrain when synoptic scale winds are weak and local flows dominate. In order to measure all the important meteorological variables with a satisfactory time and space resolution it is necessary to have a dense network of meteorological towers; this is clearly impracticable for the majority of sites.

The simple Gaussian model is not applicable for releases from sources near buildings or at coastal sites and does not include the effects of plume rise. However, extensions to the model have been proposed and the user is referred to RP 72 (Simmonds, Lawson and Mayall, 1995) and Jones (1983).

Many different atmospheric dispersion models are available meeting many different needs. These include updated Gaussian plume models, such as ADMS (CERC, 2008), Lagrangian particle models such as NAME (Jones et al, 2007) and computational fluid dynamic models. In future, it is intended that users of PC-CREAM 08 will be able to use the results of their own atmospheric dispersion models as a starting point to an assessment.

3.2.6 External radiation from the plume

The estimation of external radiation from a plume is in general carried out in two stages; the evaluation of the absorbed dose in air followed by the conversion of the absorbed dose in air to the equivalent dose in appropriate tissues and the effective dose. Different approaches are adopted depending on the nature of the radiation.

3.2.6.1 External irradiation by photons

(a) Absorbed dose in air

Two models are commonly used depending on the dimensions of the plume and the distribution of the activity within it; they are categorised as the semi-infinite and finite cloud models respectively. The finite cloud model is implemented in PC-CREAM 08 and is used for all energies above 20 keV, for energies of 20 keV or lower, the semi-infinite cloud model is used as presented below.

Semi-infinite cloud model

The estimation of absorbed dose in air from a plume emitting photons is most simply achieved by use of a semi-infinite cloud model. Implicit in this approach are the assumptions that the activity concentration in air is uniform over the volume of the plume from which photons can reach the point at which the dose is delivered and that the cloud is in radiative equilibrium: the amount of energy absorbed by a given element of cloud is then equal to that released by the same element. The absorbed dose rate in air can be expressed as:

$$D_\gamma = k_1 X \sum_{j=1}^{n} I_j E_j \qquad (3.21)$$

where D_γ is the absorbed dose rate in air (Gy y^{-1})
X is the atmospheric concentration of the nuclide (Bq m^{-3})
E_j is the initial energy of the photon (MeV)
I_j is the fraction of photons of initial energy E_j emitted per disintegration
n is the number of photons of particular energies emitted per disintegration
k_1 = 2.0 10^{-6} (Gy y^{-1} per MeV m^{-3} s^{-1})

Where the concentration distribution in the plume is sufficiently non-uniform to invalidate this approach, a finite cloud model must be used. This latter approach is adopted in PC-CREAM 08 since the use of the semi-infinite cloud model can lead to large errors over a considerable range of distances downwind, particularly for elevated releases. At large distance, when the lateral dimensions of the plume are large compared with the mean free path of the photons considered, the predictions of the finite and semi-infinite models converge.

Finite cloud model

The finite cloud model involves simulating the plume by a series of small volume sources and integrating over these sources. There are two stages in the calculation, the evaluation of the photon flux at the point of interest and the conversion of the photon flux to absorbed dose in air. In general a number of photons of differing energy and intensity are associated with the decay of a particular radionuclide. The procedure for estimating the dose for photons of a discrete decay energy is described; the evaluation of the dose from the decay of any radionuclide is obtained by summation over the photon decay energy spectrum.

The photon flux at a distance from a point source has two components, the unscattered and scattered flux. The scattered flux has undergone one or more collisions with air molecules and has a different energy from the unscattered flux; the latter has an energy equal to that of the decay photons. The effective photon flux, F, at a distance, r, from a point source is obtained by using a multiple scattering build-up factor and is given by Jones (1980):

$$F = \frac{q\,B\,(E_\gamma, \mu r)\,e^{-\mu r}}{4\pi r^2} \qquad (3.22)$$

where F is the effective flux (γ m^{-2} s^{-1})

q is the source strength (γ s^{-1})

r is the distance from the source (m)

μ is the linear attenuation coefficient (m^{-1})

B is the energy deposition build up factor and

E_γ is the initial photon energy (MeV)

The effective flux from a volume element, δV, of a plume with a concentration X per unit volume is obtained by replacing q in equation 3.22 by $X\delta V$. The total effective photon flux, F_c, from the finite cloud is obtained by integrating this modified expression over all space, ie

$$F_c = \int_V \frac{f\, X\, B\,(E_\gamma\, \mu r)\, e^{-\mu r}}{4\,\pi\, r^2}\, dV \qquad (3.23)$$

where f = intensity of the specified gamma energy

This integral is evaluated numerically in PC-CREAM 08 using spherical polar co-ordinates with the origin at the point for which the dose is to be calculated. The energy deposition build-up factors calculated by Berger, and described in Chilton (1968) are adopted, with the parametisation he suggested

$$B = 1 + a\mu r\ e^{b\mu r} \qquad (3.24)$$

where a and b are energy dependent parameters. Because the above integral cannot be evaluated exactly, approximate methods are used. The integral is transformed into polar co-ordinates and simplified. This includes defining an “edge” for the plume (even though the use of the Gaussian plume model means that the integral should really be taken over all space). The simplified integral is evaluated using a Gaussian quadrature method. The dose in air is then obtained by summing over all energy groups.

The fluxes must be converted to absorbed dose in air per unit time, using the factors in Table 3.9, and from absorbed dose to effective dose using factors that allow for the irradiation geometry. The conversion factors are given in Table 3.9 to calculate effective dose for isotropic irradiation. These are taken from ICRP Publication 74 (ICRP, 1996). The factors for isotropic irradiation are appropriate for calculating doses from material in the air; those for rotational irradiation are appropriate for calculating doses from material deposited on the ground. Factors for other irradiation geometries are given in ICRP Publication 74, and should be used if appropriate.

(b) Conversion of absorbed dose in air to dose in body organs

Conversion factors for other organs are given in ICRP Publication 74 (ICRP, 1996). These relationships are applicable to irradiation from a semi-infinite cloud but are assumed equally valid in the case of a finite cloud; this assumption is justifiable at all but the smallest distances from the release point.

The procedure adopted to estimate doses for particular radionuclides is described fully in Jones (1980) together with the nuclear data adopted which were taken from ICRP Publication 38 (ICRP, 1983). Doses are evaluated at the energies specified in Table 3.9 and values for particular radionuclides obtained by interpolation at the energies of interest taking into account the intensity of the emission at each energy.

The doses evaluated are appropriate to individuals out of doors in a rural area during the transit of the plume. Doses to people indoors will be significantly lower owing to shielding provided by building structures, etc. The reduction will depend on the time spent indoors and the nature of the buildings. Theoretical studies of the shielding provided by buildings indicate that the dose rates inside the building would be between about 0.01 and 0.9 of the dose rates in the absence of the building, depending upon the nature of the construction materials and the type of building (Brown, 1988). Values of 0.2 for family houses, 0.07 for multi-storey buildings and 0.2 for general assessments are considered typical for EU countries (Brown and Jones, 1993). Doses to people outdoors in an urban area are also likely to be smaller than those outdoors in a rural area due to shielding by the surrounding buildings. Brown and Jones (1993) suggest that dose rates outdoors in urban areas can be obtained from the calculated dose rates for outdoors in rural areas by multiplying by 0.7.

In PC-CREAM 08 for cloud gamma an indoor location factor of 0.2 and an outdoor location factor of 1 are given as a default; with a default occupancy factor of 90%. Location factors for gamma exposure and indoor occupancies can be changed in PC-CREAM 08 when calculating individual doses.

3.2.6.2 External irradiation by electrons

The range in air of electrons emitted by the radionuclides of interest is in general small (several metres at most) compared to the dimensions of the plume and an infinite cloud model can be used to estimate the dose rate. In PC-CREAM 08 datasets (USDOE, 1988) providing the dose rate to skin per unit air concentration are used to estimate the dose from external irradiation by electrons (Table 3.10). The reference (Simmonds, Lawson and Mayall, 1995) describes a more fundamental approach for calculating the dose from external irradiation by electrons.

The imprecise nature of the estimation of the dose in skin from electrons must be stressed. This imprecision arises from theoretical difficulties associated with the estimation of electron absorption in the epidermis which varies in thickness over the body and from practical considerations such as absorption by clothing and other nearby objects, for example, seats. To account for such factors is complex and beyond the scope of this study. Nevertheless it must be recognised that because they have not been considered the doses given tend to be overestimates. It is not considered appropriate to set location factors for external irradiation from electrons (cloud beta) in PC-CREAM 08 as the exposure is only from activity in the immediate vicinity and the activity concentration in air is considered to be the same indoors and outdoors.

3.2.7 Example results

The activity concentrations in air, deposition rates and doses from external radiation from the plume have been evaluated as a function of distance for a release rate of 1 Bq s^{-1} of each radionuclide for an effective release of 30 m and Pasquill D meteorological conditions (Tables 3.11 to 3.14).

Some of the more important features of the results are illustrated in Figures 3.2 to 3.4. The variation of activity concentration in air assuming a uniform windrose of a long-lived non-depositing radionuclide released at different heights in various meteorological conditions is shown in Figure 3.2. Two features are apparent: the peak activity concentration in air for a release in a particular weather category decreases with increasing height of release and it also occurs further downwind; the activity concentration in air is seen to vary significantly with meteorological conditions and the variation, at greater distances where the height of release no longer has an influence, is typically more than an order of magnitude.

The activity concentrations in air as a function of distance, for unit release of several noble gases of various half-lives, are shown in Figure 3.3. The concentrations correspond to releases from an effective stack height of 100 m in Pasquill category A conditions. The influence of radioactive decay in reducing the concentrations of the shorter-lived nuclides at long distances is clearly illustrated. The different pattern of variation of activity concentration in air with distance of a daughter nuclide is also shown, the less rapid rate of decrease in its concentration with distance, at intermediate distances, is a consequence of its accumulation in the plume owing to the decay of its parent.

The variation with distance of the activity concentration in air and effective dose rate from external γ radiation from the cloud for unit release of krypton-85 at different release heights is shown in Figure 3.4. Although there are very large differences in ground level concentrations in air at short distances for releases at different heights (several orders of magnitude), the difference in the effective dose rates from external γ radiation is small (within a factor of about 2). The marked difference in activity concentration in air is a consequence of the plume not having reached the ground in the case of the more elevated release; the effect on the dose from external γ radiation, however, is much smaller owing to the mean free path of the γ rays being comparable with the elevation of the respective plumes.

3.3 Transfer of radionuclides through the terrestrial environment

3.3.1 Introduction

Radioactive material deposited from the atmosphere onto land surfaces will be transferred through the terrestrial environment and may lead to irradiation of man by three main routes, inhalation of resuspended activity, external irradiation, and ingestion of contaminated foodstuffs. Models have been developed which describe the transfer of activity from the atmosphere to land surfaces, its subsequent transfer through the terrestrial environment and the pathways by which man may be exposed.

Each model is described in the following sections and they are applied to evaluate the time integrals of activity concentrations in air due to resuspension, of external dose and of activity transferred to selected food products as a function of time after the continuous deposition on unit area of land of each radionuclide for a year at unit rate. The matrix of results obtained, when combined with the results from application of the atmospheric dispersion model and appropriate habit data or the spatial distributions of population and agricultural yield, can be used to assess the exposure of individuals or the population via the terrestrial environment following the release of activity to the atmosphere.

3.3.2 Resuspension of deposited activity

Radioactive material deposited onto the ground can be resuspended into the air by wind or by other disturbances. The subsequent inhalation of radioactivity is recognised as a potentially significant exposure pathway. Also resuspension and subsequent re-deposition in agricultural areas may lead to contamination of crops and foods derived from grazing animals. Following the Chernobyl nuclear accident it has also been found that there may be a significant potential for the contamination of clean surfaces and crops by resuspended material (Garland, Pattenden and Playford, 1992). There have been several reviews of the processes leading to particle resuspension (eg, see Garland, Pattenden and Playford, 1992; Nicholson, 1988; Sehmel, 1980b; Anspaugh et al, 1975; Linsley, 1978 and Walsh, 2002) and only a brief summary is given here.

Resuspension varies with particle size and for the purposes of radiological protection, where the requirement is to estimate radiation doses from the inhalation of resuspended material, then it is respirable particles, ie smaller than about 50 μm, that are of concern. Particles less than about 50 μm diameter may be resuspended in response to wind action or disturbance and remain suspended for significant periods of time. Very small particles (eg, 0.1 μm or less) are unlikely to be resuspended alone (Garland and Nicholson, 1992). There is a marked increase in resuspension with wind speed (Sehmel, 1983; Garland, 1983), although resuspension has been observed to occur in all wind speeds. Sehmel (1980b) has summarised data which show the resuspension to have increased with wind speed, u, according to a power relationship in the range $u^{1.1}$ to $u^{6.4}$, depending on the nature of the surfaces and the source material being measured. Over grass and soil surfaces in the UK, Garland (1979 and 1982) has observed the resuspension factor to vary with wind speed to the power 2 or 3 but for weathered deposits the wind speed dependence power is lower, between 0 and 1. It has been suggested (Nicholson and Branson, 1991) that wind generated resuspension is likely to be episodic and that the largest particles are most readily resuspended.

Resuspension mechanisms can be broadly sub-divided into man-made and wind-driven disturbances. Resuspension caused by man-made disturbances such as vehicular traffic, digging and farming activities is usually localised. Its magnitude is variable depending on the nature of the disturbance. Evaluation of the exposure of individuals due to localised resuspension requires habit surveys in which occupational factors must be determined. Localised resuspension is a potentially important exposure pathway for particular individuals, however, for collective doses to populations the wind-driven mechanism is more important. In PC-CREAM 08 only wind driven resuspension is included as resuspension due to man-made disturbances is very situation and site specific and so is not included in this generic model. Further information on the estimation of individual doses due to man-made resuspension is considered and models are given in Simmonds, Lawson and Mayall (1995).

The process of wind-driven resuspension is complex and resuspension models for use in radiological protection tend to describe experimental observations of airborne levels above a contaminated surface, rather than mechanisms. The availability of material for resuspension varies with surface type and to some extent with the climate. Resuspension from undisturbed and ploughed soils as well as from urban surfaces is considered. Compared with undisturbed surfaces, ploughing and cultivation of land on which deposition has occurred is likely to reduce the initial wind-driven concentration in air due to resuspension because of dilution of the contaminated surface layer. At long times, however, ploughing may enhance the concentration in air due to resuspension by returning activity to the soil surface. Wind-driven resuspension from urban surfaces has received relatively little attention but the mechanisms are different from those from agricultural land. Wind-driven resuspension is likely to vary with general meteorological conditions. It is expected that resuspension would be higher in relatively dry regions, such as those found in the southern countries of the EC, than in the damper northern countries. Resuspension in these two regions is therefore considered separately in this report. However, it should be noted that a review of resuspension following Chernobyl (Garland, Pattenden and Playford, 1992; Garland and Nicholson, 1992) did not find the expected effect of climate.

The transfer of radionuclides to plant surfaces through resuspension, including the movement of radionuclides attached to soil particles onto the plant surface, can be an important mechanism. This is particularly the case for relatively long-lived radionuclides, such as the actinides, which are not readily taken up by the plant via the root system. The importance of this transfer has been widely reported (for example, see Linsley, 1983; Pinder and McLeod, 1989; Iranzo, Espinosa and Iranzo, 1988; Romney and Wallace, 1977). It is included in the model for transfer of radionuclides in terrestrial foodchains and is discussed further in section 3.3.3.4 of this report.

3.3.2.1 Review of resuspension data

A large proportion of data on resuspension has come from measurements at sites of nuclear weapons trials in semi-arid regions of the USA and Australia. However, there have also been experimental studies in damper, vegetated conditions; for example, those of Garland (1979, 1982, 1983), and measurements throughout Europe following the Chernobyl accident (Garland, Pattenden and Playford, 1992). In all cases the activity concentration in air due to resuspension is observed to decline with time after the material is first deposited. The decline is due to the surface deposit becoming progressively less available for resuspension as a result of chemical and physical changes. At longer times following deposition radioactive particles tend to become associated with soil and are therefore less available for resuspension. Information on different approaches for modelling resuspension is given in Simmonds, Lawson and Mayall (1995).

In PC-CREAM 08, Resuspension is modelled using the resuspension factor, k:

where $$k(\mathrm{m}^{-1}) = \frac{\text{Concentration in air due to resuspension (Bq m}^{-3})}{\text{Surface deposit (Bq m}^{-2})} \qquad (3.28)$$

The resuspension factor is a convenient way of expressing the observed relationship between surface and air concentration but it has a number of limitations. There is an implicit assumption that measured airborne contamination originates solely from an area local to the surface sampling position. In practice, the airborne contamination at any location will include resuspended material from a range of upwind surfaces which may be contaminated at varying levels. Also, reported resuspension factors tend to be time averaged. They may give erroneous predictions of instantaneous air concentrations, for example, in conditions when there is little or no wind or other disturbances (Linsley, 1978). The use of a time-averaged resuspension factor is generally adequate when assessing radiation exposures from continuous releases of radionuclides to the environment when, for example, annual average concentrations in air due to resuspension are used. However, caution is required when predicting instantaneous concentrations in air due to resuspension, for example, following accidental releases. The value of k also depends on the depth of surface material sampled to determine the surface deposit. Measured resuspension factors also have the drawback that the surface deposit is unlikely to be homogeneous over large distances. Resuspension factors in the range 10^{-2} to 10^{-10} m^{-1} (Linsley, 1983) have been reported and this large range may partly reflect the use of different sampling depths as well as other factors, such as age of the deposit and the surface type.

A decrease in the amount of resuspension occurring from a source has been observed with time. This time dependence has been considered by both modellers (Anspaugh et al, 1975; Simmonds, Lawson and Mayall, 1995; Linsley, 1978) and experimentalists (Garland, Pattenden and Playford, 1992; Zombori, 2000). The time dependence is not fully explained by loss of material from the surface (eg, migration of the material downwards in the soil). The material becoming less readily resuspended (Anspaugh et al, 1975) also contributes to the time dependence. Garland (1982) observed a decrease in resuspension with time and commented that some of the deposited material became positioned toward the bottom of the grass canopy, which would be relatively sheltered from the wind, and much more difficult to resuspend than the material at the top of the grass. Many authors have described the time dependence by exponential factors (Anspaugh et al, 1975; Linsley, 1978; Zombori, 2000) but Garland (1979) found that an inverse power law fitted his wind tunnel data for times varying from several minutes to a number of months.

Most measurements of resuspension factor indicate an initial value in the range 10^{-4} to 10^{-9} m^{-1} falling to about 10^{-9} m^{-1} after a few years (Sehmel, 1983; Linsley, 1978). Information on the resuspension factor at long times following deposition can be obtained from measurements of nuclear weapon test debris. Garland et al have reported (Garland, Pattenden and Playford, 1992) that such data indicate that k was less than $2\ 10^{-9}$ m^{-1} after 13 years and less than $5\ 10^{-10}$ m^{-1} 22 years after peak deposition from fallout. Following the Chernobyl accident, Garland et al (Garland, Pattenden and Playford, 1992) have reported resuspension factors in the range $3\ 10^{-10}$ to $1\ 10^{-6}$ m^{-1}. The highest value reported is thought to be due to the effect of vehicular traffic. In all cases following Chernobyl the resuspension factors decreased with time. They were also inversely correlated with initial deposit; this may be due to the movement of radioactivity from more highly contaminated regions to less contaminated regions. Some observers consider that this effect may be due to the nature of the initial deposit, with lower resuspension being observed following wet deposition than following dry deposition, although it is not clear why this effect should persist over many years (Garland and Nicholson, 1992).

An experimental radiological surveillance programme has been carried out in the area of Palomares, in the south-east of Spain, since the time of the 1966 aircraft accident. This is a typical agricultural Mediterranean area with low annual precipitation, and could be considered as representative of southern conditions. The resuspension factor due to the residual contamination in this area has been measured (Iranzo, 1994) based on yearly average concentrations of plutonium in air obtained over many years and the concentration of plutonium in surface soil (Iranzo et al, 1994; Iranzo and Salvador, 1970; Iranzo, Salvador and Iranzo, 1987). The data obtained indicate that the resuspension factor decreased from an initial average value of the order of 10^{-7} m^{-1} to values of the order of 10^{-9} m^{-1} some months later and of the order of 10^{-10} m^{-1} years later. The variability between years is about 40% and the spatial variability even greater on a distance scale of hundreds to thousands of meters. These measurements are in good agreement with the model of a time dependent resuspension factor developed by Garland (Garland, Pattenden and Playford, 1992) based on northern European conditions.

Many of the original data on resuspension were for plutonium but these have now been supplemented by wind tunnel experiments, using a variety of materials, and by data for caesium following the Chernobyl accident. Given the general variability in resuspension data there is no indication that resuspension is element dependent, although in principle the physio-chemical form of the deposit should influence its resuspension behaviour. However, given the uncertainties in resuspension data no distinction can be made between elements.

3.3.2.2 Resuspension models

(a) Wind-driven resuspension

For wind-driven resuspension in northern European conditions the model suggested by Garland (Garland, Pattenden and Playford, 1992) is used in PC-CREAM 08. This model is based on a time-dependent resuspension factor with the resuspension decreasing according to 1/T, modified to account for long term suspension (Walsh, 2002).

The formula used for estimating the resuspension factor is:

$$k(t) = 1.2\,10^{-6}\,t^{-1} + 10^{-9} \tag{3.29}$$

During the first day a resuspension factor of 1.2 10^{-6} m^{-1} is assumed (Walsh, 2002) and to allow for radioactive decay the modified Garland formula becomes

$$\mathrm{k(t)} = (1.2\ 10^{-6}\,\mathrm{t}^{-1} + 10^{-9})e^{-\lambda \mathrm{t}} \tag{3.30}$$

where k is the resuspension factor (m^{-1}), λ is the radioactive decay constant (day^{-1}) and t is the time after deposition in days.

The integrated activity concentration in air due to resuspension I_R, to time t_2 for times beyond one day from deposition can be expressed as

$$\mathrm{I_R} = \int_{t_1}^{t_2} \mathrm{k(t)dt} = 1.2\ 10^{-6} \int_{t_1}^{t_2} \frac{e^{-\lambda \mathrm{t}}}{\mathrm{t}} \mathrm{dt} + 1\,10^{-9} \int_{t_1}^{t_2} e^{-\lambda \mathrm{t}} \mathrm{dt} \tag{3.31}$$

The concentrations in air due to resuspension of radioactive progeny, where they are important, can also be determined in a similar manner. In PC-CREAM 08 the first progeny that does not reach secular equilibrium after one year is modelled explicitly.

For continuous deposition, the integrated activity concentration in air due to resuspension I_R, to time t_2 resulting from the deposition of a radionuclide continuously for 1 year at a rate of 1 Bq m^{-2} s^{-1} can be expressed as

$$I_R \equiv 8.64\ 10^4 \sum_{\mathrm{t_1=1}}^{365} \int_{1}^{\mathrm{t_2-t_1}} \mathrm{k(t)\,dt} \tag{3.32}$$

where t is days and k(t) is the resuspension factor (m^{-1}). For large t_2, the integral can be written explicitly as

$$\mathrm{I_R} = 3.15\,10^7 \int_{1}^{\mathrm{t_2}} \mathrm{k(t)\,dt} \tag{3.33}$$

These relationships are appropriate for undisturbed surfaces in a rural environment in northern Europe. For an urban environment the situation will be different. A series of experiments has been conducted by Nicholson and Branson (1990) to determine atmosphere-surface exchange in towns and cities. Wind-tunnel experiments on resuspension from a concrete surface and field measurements of traffic resuspension from a road have both indicated rapid resuspension rates. However, the results from this work are not in a form that they can easily be incorporated into the models included in this methodology as only resuspension rates are given. It is interesting that the reduction in resuspension rate with time over a concrete surface was found to follow a similar 1/T relationship to that found by Garland for a grass surface. In the absence of further information it is suggested that for estimating average or collective radiation exposures the relationship given in equation 3.29 is assumed to apply to both rural and urban environments.

The values obtained in a semi-arid region in southern Europe (Palomares) for the resuspension factor were discussed in section 3.3.2.1. The observed time dependence is roughly in agreement with the Garland model and values are compatible with those based on fallout measurements. Therefore, the Garland model seems also to be applicable to the dryer conditions which are frequent in southern Europe. However, when using the Garland model in these conditions the predicted values should be considered approximate, due to the observed variability in time and space.

(b) Man-made resuspension

As discussed earlier man-made disturbances such as digging, agricultural activities and vehicular traffic can lead to localised enhanced levels of concentrations in air due to resuspension. A limited number of studies have been carried out in this area (for example, see Nicholson,1988; Nicholson and Branson, 1991; USEPA, 1977; Batel, 1975; Nicholson and Branson, 1990). However, there are insufficient data to develop detailed models of man-made resuspension, for example distinguishing between different activities, including time dependence. Currently, PC-CREAM 08 does not include this exposure pathway but it would be possible to use data obtained from the code and a dust loading approach (Simmonds, Lawson and Mayall, 1995) to estimate radiation doses.

(c) Model limitations

The models suggested for resuspension are very simple, empirical models to represent the complex processes involved. They are considered to be sufficiently robust for use in assessing individual and collective doses due to routine releases of radionuclides to the environment. For such assessments, annual or long-term integrated exposures are required and so the averaging implicit in the models is appropriate. However, care should be used in applying such models in considering the accidental releases of radionuclides. The models do not include short-term time dependence which could be important in some circumstances given the episodic nature of wind-driven resuspension. Also, no account is taken of movement of activity between areas contaminated at different levels.

Further factors affecting resuspension include soil moisture content and the extent of the vegetation canopy (Sehmel, 1984). Both are particularly small in semi-arid environments, such as the ones found in southern Europe. In addition, coastal southern regions are influenced by wind breeze cycles. All of these characteristics suggest that short term time dependence of resuspension is particularly important in southern Mediterranean sites near to the coast. Resuspension indoors has also been considered in some studies and the extent of resuspension has been found to be lower than for outdoor environments (Walsh, 2002).

More complex models have been developed to take into account short-term effects and soil inhomogeneities (eg, Garcia-Olivares, 1992). Other models take account of saltation (Travis, 1975) or wind speed (Berofsky, 1987). The main limitation of these more complex models is the large amount of site-dependent data required.

3.3.2.3 Results of the resuspension model

A matrix of results has been evaluated using the time dependent resuspension factor model, modified for radioactivity decay, given by equation 3.30. The results calculated are the instantaneous and time integrated (to various times) resuspended air concentration of each radionuclide above a surface on which the radionuclide has been deposited at a rate of 1 Bq m^{-2} s^{-1} for one year. As equation 3.30 does not have an exact integral the integration has been carried out numerically. The resuspension factor is independent of the deposited element and variation in the resuspended air concentration is solely a function of radioactive half life.

Table 3.15 gives the resuspended activity concentration in air integrated to various times and to infinity from a deposit of 1 Bq m^{-2} s^{-1} for one year for a selection of radionuclides. Figure 3.5 shows the variation in the integrated concentrations in air with time following this deposition for four radionuclides. The effect of radioactive decay is clearly seen from these results with the longer-lived radionuclides having higher integrated concentrations in air than the short-lived ruthenium-103.

It is possible to compare the predicted activity concentrations in air due to resuspension with those that would have given rise to the original deposition. The relation between concentration in air and ground deposition depends on the dry deposition velocity assumed, neglecting wet deposition (see section 3.2.2.3). For a dry deposition velocity of 10^{-3} ms^{-1} the concentration in air giving rise to a dry deposition of 1 Bq m^{-2} s^{-1} would be 10^3 Bq m^{-3}. Integrated over a year this is 3.15 10^{10} Bq s m^{-3}, considerably higher than the integrated concentrations in air due to resuspension given in Figure 3.5 and in Table 3.15.

3.3.2.4 Validation

Some additional validation of the Garland formula for resuspension, a modified version of which is used in PC-CREAM 08, has taken place using additional data obtained following the accident at the Chernobyl nuclear power plant. See Walsh (2002) for more details.

3.3.3 The transfer of radionuclides through terrestrial foodchains

3.3.3.1 Introduction

The transfer of radionuclides through the terrestrial environment into foods is complex; many processes are involved and much depends on the characteristics of the radionuclide and the particular environment. The more important transfer processes are illustrated schematically below.

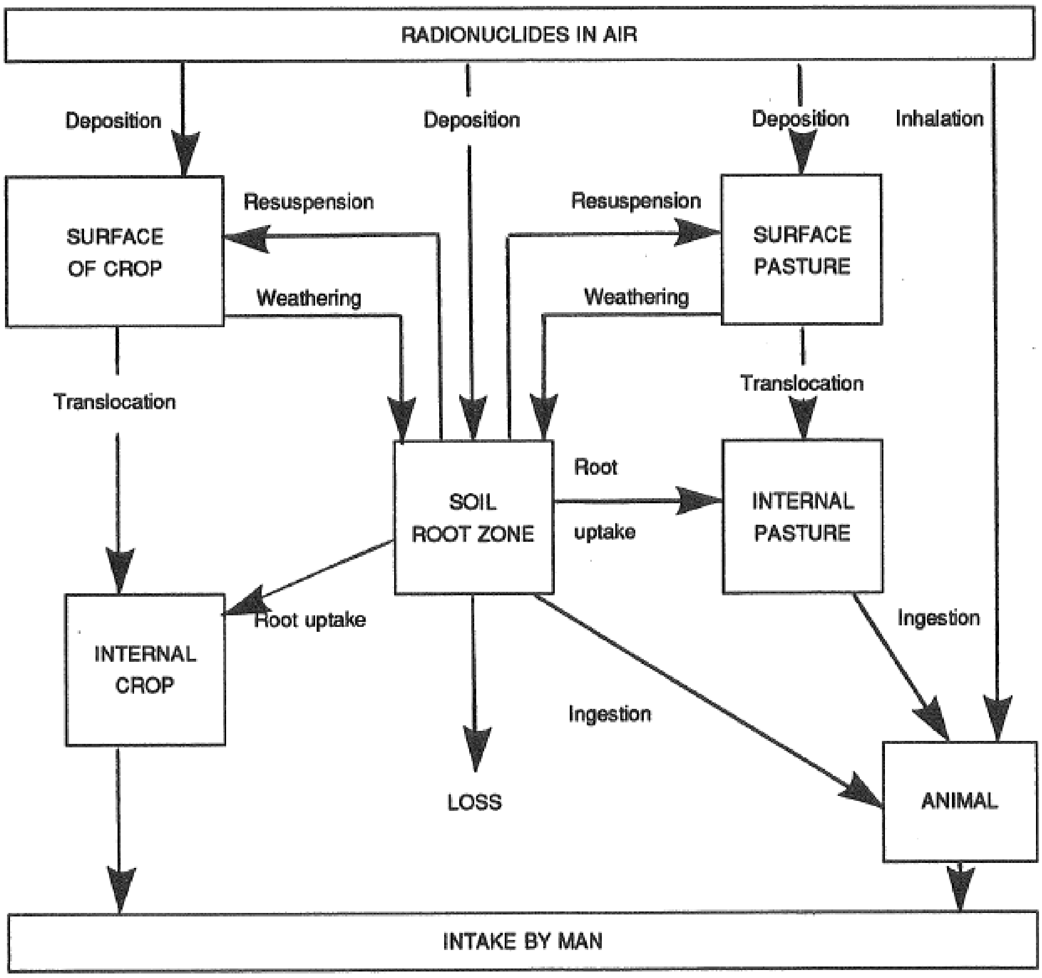

Diagram 3.2 The important processes for transfer of radionuclides through terrestrial foodchains

These transfer processes are discussed in the sections below.

Various models have been developed to predict the transfer of radionuclides through foodchains following both routine and accidental releases of radionuclides to atmosphere. The most appropriate foodchain model to use depends on the particular application and on the desired endpoints. In this report parameter values are provided as defaults for use with a variety of foodchain models but particular examples are based on HPA's dynamic foodchain model FARMLAND (Brown and Simmonds, 1995) which is used in PC-CREAM 08. There are many foodchain models available of differing degrees of complexity and an important distinction is whether they are designed to predict the time dependence of the transfer, or just the total amount of activity transferred. Multiplicative models use a series of factors to relate levels of radioactive material in the various components of the foodchain to man. Models of this type are widely used, particularly in the United States (see USNRC, 1977; Till and Meyer, 1983) and were suggested by the International Atomic Energy Agency (IAEA, 2001). Such models are relatively simple to apply and have a large database available but they give only limited information on the time dependence of the transfer. These results represent an averaging over time which is generally acceptable for assessing the transfer following continuous releases of radionuclides to the environment.

The physical processes by which radionuclides transfer through terrestrial foodchains are very complex and it is often convenient to model these processes by a series of interconnected compartments, each representing different parts of the foodchain. Many dynamic models are of the compartment type and have the advantage of being very flexible and able to accommodate large differences in the amount of detail included in different parts of the system. The HPA model FARMLAND is of this type and is outlined in this report. Other dynamic models include: ECOSYS (Müller and Pröhl, 1993), which was developed at Gesellschaft für Strahlen-und-Umweltforschung, FRG and the Food Standards Agency's (FSA) model PRISM in the UK (Thorne, Maul and Robinson, 2005; Thorne, Maul and Robinson, 2004), which was developed by Associated Nuclear Services in the UK for the Ministry of Agriculture, Fisheries and Food (MAFF). These models have many similar features but also differences mainly due to the different purposes for which they were developed. Various comparisons have been made between the models and some of these are summarised in Simmonds, Steinhauer and Haywood (1987).

The models and parameter values given in this report are specifically for evaluating the transfer of radionuclides through foodchains following deposition from atmosphere as a result of routine, continuous releases to the environment. For other applications these models may not be adequate and so should be used with care. In particular, no consideration is given to variations in agricultural practice or in season of the year as these are not important for continuous releases. However, they can be significant for accidental releases and extensions to the models to take these into account may be appropriate. The use of foodchain models in accident consequence assessment is discussed elsewhere (see, for example, Simmonds, Steinhauer and Haywood, 1987; Sinnaeve and Gerber, 1991). FARMLAND can be used for both routine and accidental releases as discussed in Brown and Simmonds (1995).

Under the EC post-Chernobyl research programme the two dynamic foodchain models ECOSYS (Müller and Pröhl, 1993) and FARMLAND (Brown and Simmonds, 1995) were used as a basis for recommending a general model for use in the EC (Sinnaeve and Gerber, 1991). This general model was for use in calculating Derived Intervention Levels. Parameter values were provided for use with ECOSYS, FARMLAND or any similar model. This work formed an input to the version of FARMLAND used in PC-CREAM 08, with modifications to take account of the source being continuous rather than accidental.

Many laboratory and field studies have been carried out that are relevant to the transfer of radionuclides through terrestrial foodchains. These have been related to fallout from the atmospheric testing of nuclear weapons, routine discharges from nuclear installations and accidental releases of radionuclides to the environment. Of particular note is the 1986 accident at the Chernobyl nuclear power station in the Ukraine which has provided a wide body of data for use in model development and validation. However, in all cases attention has generally been focused on a fairly limited range of elements, notably caesium, strontium and iodine, whose transfer through foodchains are of particular importance, together with plutonium and americium which are of general radiological concern. The movement of other elements through foodchains has not been studied so widely and therefore is relatively less well understood. This lack of data can lead to significant uncertainties in the transfer of such elements, which should be recognised. Further aspects of the limitations of the models are discussed later.

A wide range of radionuclides has been considered and are included in PC-CREAM 08. However, a different approach has been adopted for tritium and carbon-14. Due to their special behaviour in the environment a specific activity approach has been adopted (see section 3.3.3.6). In PC-CREAM 08 the first radioactive progeny, of each discharged radionuclide, that does not reach secular equilibrium after one year is modelled explicitly. However, there are some cases where differences in model configurations mean that no progeny can be modelled.

The number of foods consumed by people is considerable and their relative contribution to the total diet varies considerably between individuals with distinct regional variations (Sinnaeve and Gerber, 1991). Data on the transfer processes of radionuclides in a wide range of food types are limited and for carrying out radiological assessments it is not necessary to distinguish between many similar foods. Therefore the various foods have been grouped into a number of categories found to be important components of the EU diet. These categories are: green vegetables; grain products; root crops including potatoes; fruit; meat; liver; milk and milk products. The animals considered in PC-CREAM 08 are cattle and sheep, for which full dynamic models incorporating both transfer through pasture and animal metabolism are provided. The intake of activity by pigs and chickens is variable depending on feeding practices, particularly as in many instances they are permanently reared indoors. The behaviour of goats is not as well studied as that of cattle and sheep and so it is not possible to provide full models for the transfer of radionuclides to goat products. Currently the transfer of radionuclides to pigs, goats and chickens is not included in PC-CREAM 08 but relevant transfer data are available in Simmonds, Lawson and Mayall (1995). In modelling the movement of activity through the terrestrial environment a modular approach is adopted due to the large number of pathways and processes that must be considered. Each module is concerned with a particular part of the environment but there may be more than one module for each part. The development of more than one module may reflect the need for models of differing complexity or for models to be tailored to make use of the environmental data that are available. The modules can be combined in various ways to describe the movement of activity through the terrestrial environment. Although the modules were developed in the context of land contamination resulting from atmospheric deposition they can, with appropriate choice of parameters, be applied to other types of deposition such as irrigation although this is not done in PC-CREAM 08.

The modules developed to describe the three main processes that influence the transfer of radionuclides through foodchains to man are outlined in the following sections. The three processes are: migration of radionuclides in soil, transfer to plants; transfer to animals. The technique of compartmental analysis is adopted to model the transfer of activity through each system. The technique in its most general form is summarised in Appendix A. Activity can be transferred between compartments representing different parts of the system according to prescribed transfer rates and within each compartment uniform mixing is assumed.

The models and parameter values suggested in this study have been chosen to be suitable for the EC as a whole. However, the variations in agricultural practices and consumption, together with behaviour of radionuclides in the foodchain across the EC, should be recognised. A previous study (Sinnaeve and Gerber, 1991) identified differences between northern and southern (particularly Mediterranean) conditions as being potentially important. These differences have been considered (Métivier and Bénédetti, 1994) and the findings are summarised in a later section.

3.3.3.2 Migration of radionuclides in soil

Migration into soil and radioactive decay are the principal mechanisms which determine the long term time dependence of the uptake of radionuclides into plants and animal products following the deposition of activity onto land. For some elements, changes in chemical form can also affect their availability for uptake by roots. Many parameters influence the rate of migration, particularly the nature of the element and its chemical form, soil composition, climate, and rainfall. Agricultural land can be categorised into one of two types for the purposes of modelling migration: undisturbed land (for example, permanent pasture) or land where the soil is kept well-mixed by frequent ploughing or cultivation. Two models have been developed to represent migration in the respective conditions and their main features are outlined.

(a) Model for well-mixed soil

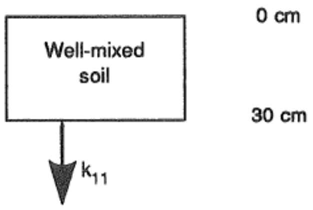

Diagram 3.3 Schematic diagram of the model for well-mixed soil

The model, shown schematically above, is intended to represent land which is ploughed or cultivated annually or more frequently. The radionuclides are assumed to be uniformly mixed and equally available through the top 30 cm of soil; the choice of a depth of 30 cm encompasses the variation in depth of the root zones of most plants. Implicit in this approach is the assumption that the uniform profile is not significantly altered by migration in the intervening period between ploughing and cultivation; this assumption is in general valid for the radionuclides considered in this study. Loss from the root zone occurs by downwards penetration processes of which diffusion and transport along with general water movement are the most important; the rate of loss is represented by the transfer coefficient, k_{11}.

(b) Model for undisturbed land

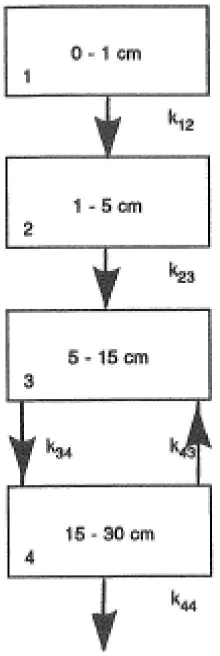

Diagram 3.4 Schematic diagram of the model for undisturbed land

The model, shown schematically above, is intended to represent migration through undisturbed agricultural land of which permanent pasture is an example. The movement of the radionuclides through the soil column is represented by a series of transfers between compartments of varying depth; within each compartment the radionuclide is assumed to be uniformly mixed. The rates of transfer are represented by the transfer coefficients k_{12} to k_{44}. The selection of compartments is a compromise between ensuring an adequate representation of the migration processes and including those that have physical significance for other parts of the terrestrial model. For example, resuspension of the contaminant onto plant surfaces is assumed to be derived solely from the top 1 cm of soil, that is, the surface compartment. Soil which is consumed inadvertently by animals is also assumed to be derived from the top 1 cm layer. The root zone of pasture grass is considered to extend to 15 cm in depth and the contaminant present in each of the upper three compartments is available for root uptake.

Migration into soils has only been studied for a limited number of elements, mainly those long-lived radionuclides which are present in nuclear weapons fallout, in particular isotopes of caesium, strontium and plutonium. In addition there are some migration data available, particularly for plutonium, from single deposits on the ground following tests and accidents with nuclear devices (Aarkrog, 1977l; Nyhan et al, 1976) and leakage of radioactive material at a nuclear facility (Krey et al, 1977). The rate of movement into the soil for caesium, strontium and plutonium is slow, although there is significant variation between the results of the various observations due to differences in soil composition and annual rainfall.

In view of the limited data on soil migration the transfer coefficients used in the model are based on the soil migration data for plutonium and are assumed to be applicable to all other elements. This approach is considered realistic for elements such as caesium and strontium which appear to migrate at a similar rate to plutonium, but for more mobile elements such as iodine the approach is conservative. The transfer coefficients have been derived from experimental measurements of the migration of single deposits of plutonium in various soils (Aarkrog, 1977; Nyhan et al, 1976; Krey et al, 1977) and are summarised in Table 3.16. The observed rates of migration show considerable variation with soil type and the transfer coefficients adopted are those which give the best overall fit. A recent review of the soil migration model and comparison with more recent measurement data by Busby (1999) supports the continuing use of these values for all elements, with a slightly modified approach for strontium (Haywood et al, 1980).

Observations of the migration of plutonium in soils have only been made for periods extending a few decades after the initial deposition. Any estimate of the rate of migration at greater times must therefore be speculative. From a review and extrapolation of the migration data for plutonium, strontium and caesium a judgement has been made that a half-life of 100 years can be taken as representative of the rate of removal of activity from the top 30 cm of soil (or half-life of 50 years for removal from the 15–30 cm zone in the undisturbed model). The speculative nature of this estimate must however be recognised. The transfer coefficients k_{11} and k_{44} in the well-mixed and undisturbed soil models, respectively, are consistent with the above half-lives. The transfer coefficients given in Table 3.16 relate to the migration of a stable radionuclide; radioactive decay, or in-growth of daughter products, are incorporated separately into the model with additional transfer coefficients from each of the compartments.

3.3.3.3 *Transfer of radionuclides to plants*

The main features of models to describe the transfer of radionuclides to plants are illustrated schematically below.

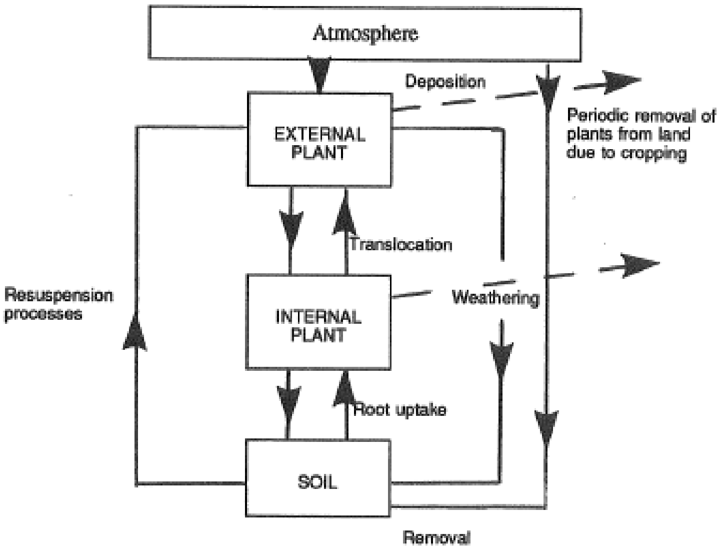

Diagram 3.5 Schematic representation of model for transfer of radionuclides to plants

The compartment marked `soil’ represents the model for migration in soil appropriate to the particular plant and agricultural conditions considered. All plants consumed directly by people are assumed in this study to be derived from land that is frequently cultivated and the migration model for well-mixed soil is most appropriate in these circumstances. Grass, however, is assumed to be produced only on undisturbed pasture in which case the migration model for undisturbed soil is applicable. Both internal and external compartments are considered for plants: transfer to the external plant surfaces may occur by interception of depositing activity or by resuspension of activity from soil; transfer to the internal plant occurs by root uptake and translocation from the external surfaces. Each process is considered in turn. In the case of the more recently completed fruit model, the way in which transfer rates were derived follows much the same methodology as the other FARMLAND crops and the specific details are included in the report on the modelling approach (Brown and Sherwood, 2008).

(a) Interception, retention and translocation of deposited radionuclides

When radionuclides deposit from the atmosphere onto agricultural land, part is intercepted by the foliage of vegetation while part will land on the soil. In general, radioactivity is removed from the surfaces of plants by natural processes such as weathering, with a half-life ranging from a few to several tens of days. The interception and retention of radionuclides on plant surfaces varies according to the physico-chemical form of the deposit, the nature of the vegetation and the prevailing conditions. Considerable variation has been observed in measured values of interception factors and removal rates of activity from plant surfaces (Chamberlain, 1970; Chamberlain and Garland, 1991; Pröhl and Hoffman, 1996). An important factor for interception is whether the radionuclides are deposited through dry or wet processes (Müller and Pröhl, 1993; Sinnaeve and Gerber, 1991). However, in assessing the consequences of routine releases of radionuclides to atmosphere, it is not necessary to distinguish between the two types of deposition. The input to the system is the total deposition resulting from both dry and wet processes, and it is appropriate to use interception factors applicable for annual average conditions.

Part of the surface deposits on plants may be absorbed and transferred to other parts of the plant; this process is known as translocation and is far more significant for some elements, for example, caesium, than for others, for example, plutonium. For root vegetables, translocation is particularly important and it is the only mechanism which transfers radionuclides from the surfaces of the portion of the plant above ground to the edible root.

Much of the external contamination on crops at harvest will be removed before consumption of the edible parts by man. Washing and removal of outer leaves of green vegetables leads to a reduction in radionuclide concentration as does the removal of the outer layers of grain in the production of flour. Data on losses through preparation and processing have been widely studied, especially in the context of possible reductions following accidental releases (Sinnaeve and Gerber, 1991; CEC, 1989). A comprehensive review of published information on the effects of processing on radionuclide content of foods has been undertaken (Green and Wilkins, 1995). The distribution of radionuclides in foodstuffs can heavily influence the amount of loss due to processing and preparation as found by a recent study (Green and Poultney, 1999).

In FARMLAND, the interception factor, retention half-times, processing and preparation losses, together with the transfer coefficients representing translocation for crops (green vegetables, grain, root crops and fruit) are all chosen to fit available environmental and experimental data. Consideration has been given to both the time-dependence of the transfer for single deposits and the overall transfer following continuous deposition (Simmonds and Linsley, 1982; Brown and Simmonds, 1995). Data for continuous deposition can be expressed in the form of a normalised specific activity, NSA, defined by Chamberlain (1970) as:

$$\text{NSA}\ (\text{m}^2\ \text{days kg}^{-1}) = \frac{\text{Activity kg}^{-1}\ \text{dry weight of crop}}{\text{Activity deposited per day per m}^2\ \text{of ground}}\ 1 \qquad (3.34)$$

Table 3.17 gives values for NSA obtained from experimental measurements for grain, green vegetables, fruit and root crops, together with those predicted by FARMLAND for caesium, strontium and plutonium. A distinction is made between grain and flour and between prepared and unprepared green vegetables. With the exception of plutonium in green vegetables, the models provide reasonable agreement with the measured data. Other elements are assumed to behave like one of these three; mobile elements behave like caesium while less mobile elements behave like strontium and immobile elements are assumed to behave like plutonium. The few measured NSAs for plutonium are considerably lower than those for strontium and caesium. This is to be expected as plutonium is known to be very immobile in plants and effectively does not translocate. For grain, assuming that only direct contamination of the grain seed occurs, the predicted NSAs for plutonium are in reasonable agreement with those based on measurements. However, for green vegetables, neglecting translocation still leads to a significant overestimate of the measured NSA for plutonium in prepared vegetables. If required, this overestimate could be reduced by lowering the retention half-time for plutonium to 5 days from 14 days (Simmonds and Linsley, 1982). For fruit, translocation is only modelled implicitly through choice of interception factor and retention half-time (Brown and Sherwood, 2008). In all cases the default values for the parameters used to model these processes are inter-related both between themselves and to the yield of crops adopted. They should not, therefore, be changed independently without considerable care.

For pasture, both grazed and grown for hay/silage, only interception and retention are modelled explicitly with translocation only being included implicitly through the choice of values for the other parameters. The interception factors are related to the pasture density based on the relationships observed by Chamberlain (1970). The retention half-life for pasture is based on a review of available data (Brown and Simmonds, 1995). For short term, accidental releases it may be necessary to consider different values for the summer months when cattle and sheep are grazing outdoors and for the winter when cattle are indoors and a longer retention half-life for weathering has been observed (Chadwick and Chamberlain, 1970). However, this is not included in PC-CREAM 08.

(b) Resuspension of activity from soil to external plant surfaces

The general processes of resuspension (see section 3.3.2) will result in the transfer of activity from the soil to the external surfaces of plants. Considerable variation might be expected in the importance of this route of contamination with plant type, in particular with growing habits and method of preparation before consumption. Few data are available in this area, however, and the same general approach is adopted for all surface crops.

The contamination of plant surfaces by resuspension is considered in two stages. The first concerns the resuspension of the radioactivity in the period soon after deposition by wind driven processes. The second involves the resuspension by a variety of processes, of soil particles with which the radioactivity becomes associated within a few months (or years) of deposition. The first process is governed by the time dependent resuspension formula described in section 3.3.2; various approximations have been made to facilitate the incorporation of this relationship into FARMLAND (Brown and Simmonds, 1995) and the same formula is used in the derivation of resuspension transfer factors in the more recent fruit model (Brown and Sherwood, 2008).

The transfer of radionuclides to external plant surfaces by the second process can be determined readily from the quantities of soil typically associated with the edible parts of crops when harvested. The concentration of activity in the soil on plant surfaces is assumed to be the same as that in the well-mixed top 30 cm layer; the sole exception is grass, which is assumed to be derived from undisturbed soil, and the approach adopted for it is discussed in section 3.3.3.4(a). The quantities of soil contaminating various plant surfaces are uncertain and the relevant data are summarised below.

Measurements in the United Kingdom (Reith, 1977) and the USA (McLeod et al, 1980) have shown that a value of 0.01% is typical of the quantity of soil associated with the whole grain seed when expressed in terms of the dry weight of the latter; in exceptional circumstances it could be as much as 0.1%. For vegetables there is even greater uncertainty in assigning an average value to the amount of soil contamination on plant surfaces. There is considerable variation in the physical characteristics of vegetables and this will have a marked influence on the degree of surface contamination; for example leguminous vegetables, such as peas, are protected by a pod, whereas leafy vegetables, such as lettuce, are grown close to the ground and may sometimes be subject to significant contamination by soil (Pinder and McLeod, 1989). A value of 0.1% of the dry plant weight has been suggested (Reith, 1977; Pinder and McLeod, 1989) as an average amount of soil in leafy vegetables and this value has been adopted for all vegetables. It has been suggested that this value is also appropriate for herbaceous and shrub fruits (Brown and Sherwood, 2008) and has been included in the generic fruit model in PC-CREAM 08.

The assumptions outlined in section 3.3.3.3(a), concerning the amount of radioactivity on the surface removed by preparation before consumption, are also adopted here.

(c) Root uptake

The absorption of elements from soil by plants varies considerably depending on a number of factors, notably soil type. There can also be significant variation due to the nature of the plants (for example, root crops compared with grain crops) and the chemical form of the element has a major influence. However, with the exceptions of a few elements, such as strontium and caesium, and to a lesser extent the transuranium elements, these variations have not been investigated in detail. The International Union of Radioecologists (IUR) has compiled a database of root uptake data for a variety of radionuclides, plants and soil types (IUR, 1989). This database has been used together with databases compiled by Ng et al (1982), Simmonds and Crick (1982) and other sources to obtain root uptake transfer data for a number of elements. Work on some radionuclides has been more intensive than for others. Due to the high level of radiological interest in radiostrontium and radiocaesium there is a wealth of data available for these radionuclides including work by Nisbet and Woodman (2000). In some cases it has been possible to distinguish between the transfers between soil and different plant species. For many elements it has not been possible to make this distinction and the transfer has been assumed to be independent of plant type. In some cases the paucity of data is such that the transfers are chosen by analogy with other elements for which data are available.

In addition to the migration process which progressively removes elements from the rooting zone, some contaminants become increasingly unavailable for root uptake as a result of chemical immobilisation processes in the soil. This process, known as fixation, is particularly important for caesium and is included in FARMLAND (Brown and Simmonds, 1995) for this element. For pasture, fixation is included explicitly as shown in Appendix B. However, for root uptake into crops the fixation is included implicitly in the choice of root uptake data.

Data on root uptake tend to be in the form of concentration factors between plants and soil at the end of the growing period. Such data contain no information on the time dependence of the uptake mechanisms and as such cannot be rigorously applied in that context. Their application to a model which is time dependent in character is valid only insofar as the variation in the concentration of the radionuclide in the root zone is small during the growing period. For long-lived radionuclides this assumption is in general valid. Where it is not, the assumption is made in the model that the plant rapidly comes into equilibrium with the soil as determined by the concentration factor. Where the concentration of activity in soil varies rapidly with time the activity in plants will be determined largely by the concentration in soil just prior to harvesting. The derivation of transfer coefficients, which are used in the model from concentration factors, is described in Brown and Simmonds (1995); while these coefficients represent the rate of transfer from soil to plant and vice versa the time dependence is an artefact. The coefficients are chosen solely to ensure that the concentration factor between plant and soil is attained rapidly.

In addition to the mechanisms described in (a) to (c) above, account is taken in the model of removal of activity from the system by harvesting of crops; for most crops a growing period of 120 days (150 days for fruit) is assumed before cropping. Account is also taken of radioactive decay, and the in-growth of daughter products, where appropriate, in each compartment in the system.

(d) Summary of the data adopted to evaluate transfer in plants

The values adopted for the non-element dependent parameters used in the model are summarised in Table 3.18. Data are given for plant yields, interception factor and half-life of retention on plant surfaces, the level of soil contamination on plants before preparation for consumption, growing period before cropping and the depth of the root zone. The values adopted for the element dependent parameters, in particular concentration factors, are summarised in Table 3.19.

Transfer coefficients, which are used in the model, have been derived from data in Tables 3.18 and 3.19 and are summarised in Appendix B.

3.3.3.4 Transfer of radionuclides in animals

It is convenient to consider the transfer of radionuclides to animals in two stages; these comprise the intake of radionuclides into the animal by ingestion or inhalation and the subsequent metabolism of these radionuclides, in particular their transfer to animal tissues (and/or produce) consumed by man. The models developed to describe these transfer processes for grazing animals are described.

(a) Intake of radionuclides by grazing animals

The principal mechanisms involved in the transfer of radionuclides to grazing animals are illustrated schematically below.

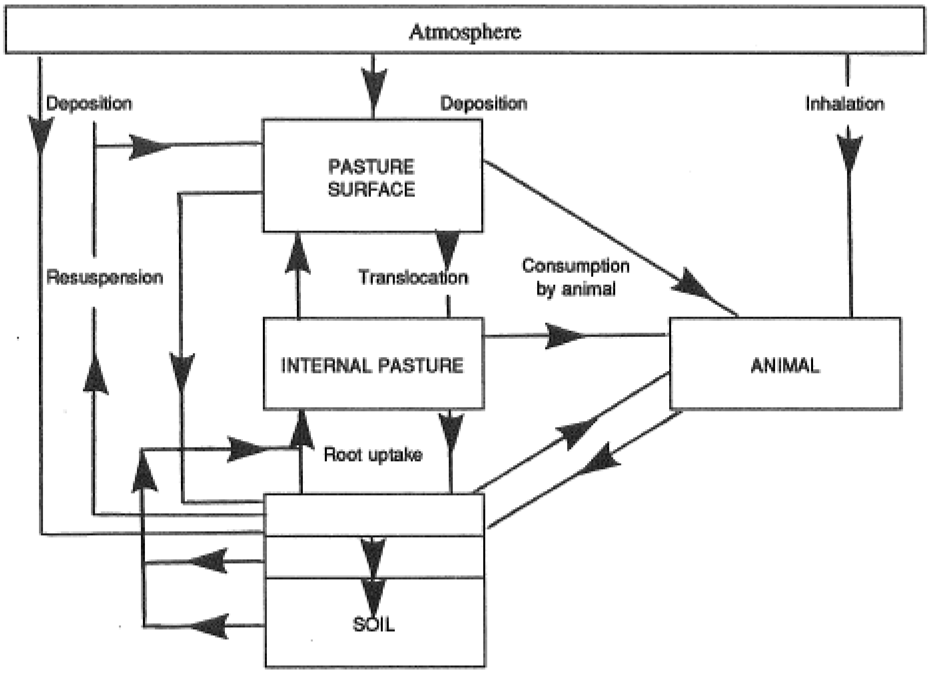

Diagram 3.6 Schematic representation of the principal mechanisms for the transfer of radionuclides in grazing animals

The compartment marked “soil” corresponds to the model for undisturbed soil described in section 3.3.3.2; grass is assumed to be derived solely from undisturbed pasture in this study. The two main routes of intake are ingestion and inhalation. Ingestion is by far the most important pathway for the intake of most radionuclides although inhalation may be significant for those radionuclides whose transfer across the gut of the animal is small.

The consumption of pasture grass is in general the most important mode of intake by ingestion. Three main processes contribute to the contamination of pasture grass: deposition; resuspension; root uptake. Translocation is negligible by comparison. These processes are described in the previous section where the values of the parameters required to estimate the transfer of radionuclides to grass are summarised. The inadvertent consumption of soil, together with grass, by grazing animals is a further pathway which must be considered; for radionuclides which have a low transfer from soil to grass by root uptake the ingestion of soil may be the most important mode of intake. Typically the inadvertent consumption of soil is about 4% and 20% of dry matter intake for cattle and sheep respectively (Thorton and Kinniburgh, 1977; Thorton, 1974); the potential significance of this pathway is evident. Resuspension of activity onto the surface of undisturbed pasture in the period shortly after deposition is governed by the time dependent resuspension formula described in section 3.3.2; the approach is simplified by various approximations to facilitate its inclusion in the model (Brown and Simmonds, 1995).

In general two routes of intake must be considered for the inhalation of activity by animals; the inhalation of activity while the deposition onto pasture is taking place and the subsequent inhalation of resuspended activity. The former may not apply if the source of contamination is not deposition from the atmosphere or if, for example, the animal is introduced to the pasture after the deposition process has ceased. The intake of activity by inhalation while deposition is continuing can be readily evaluated from the breathing rate of the animal and the time dependence of the activity concentration in air. The procedures described in section 3.2 to evaluate concentrations in air of radionuclides and their inhalation by man are equally applicable to animals and are adopted for that purpose; reference should be made to section 3.2 for further details and this aspect is not considered further here. As before, the inhalation of resuspended activity is estimated by the use of simplifying approximations applied to the time dependent resuspension formula (section 3.3.2); the approximations are described elsewhere (Brown and Simmonds, 1995). The time dependent resuspension formula applies to wind-driven resuspension from undisturbed pasture. The habits of grazing animals (in particular the proximity of their head to the ground and the disturbance of the ground during grazing) may result in enhanced but localised resuspended concentrations in air and thus greater intakes by inhalation. Owing to the absence of data no account is taken of this additional resuspension mechanism; it must be recognised however that the inhalation by grazing animals of resuspended activity may consequently be underestimated.

The values adopted for the various parameters concerned with the transfer of radionuclides to grass are summarised in Table 3.18 and 3.19 together with similar data for other plants. The values adopted for the non-element dependent parameters for grazing animals (for example, ingestion and inhalation rates, animal and organ weight, milk yields) are summarised in Table 3.20 and their choice is discussed in more detail elsewhere (Brown and Simmonds, 1995). Consideration is limited to cattle and sheep. The average lifetime of cattle is taken to be six years; this is based on the average lifetime of dairy cattle in the UK. The average lifetime for beef cattle (and dairy cattle in some other parts of the EC) can be considerably shorter and the assumption is recognised as being conservative for those long-lived radionuclides which build up in animal tissue. Sheep are taken to have a mean lifetime of 1 year.

(b) Metabolism of inhaled or ingested radionuclides

The behaviour in the body of radionuclides inhaled or ingested by animals will depend upon the particular chemical element concerned. Data on the metabolism of radionuclides in animals are limited and use is often made of data and models developed to represent the human metabolism. The degree of complexity necessary in modelling the metabolism of a radionuclide in animals depends on the element concerned and the particular circumstances. It is usually sufficient to limit consideration to the transfer of radionuclides to muscle, liver, milk and eggs; these represent the main animal products consumed by people. For radionuclides whose biological half-lives in various organs are small compared with the animals’ lifespan a relatively simple model can be adopted. It is sufficient to model the fractional transfer of ingested or inhaled activity to particular organs and the half-lives of the activity in those organs. For a few radionuclides a more complex model is required if the metabolism is to be represented adequately. Recycling of activity between organs and body fluids may occur with particular implications for the time dependence of the transfer of activity to milk. In such cases both the transfer to and recycling of activity must be modelled and additional organs such as bone or thyroid need to be included. Models of both types are used in this study for cattle and sheep. The simpler model is used for all but the transuranium elements, strontium, iodine and caesium; both models are outlined below.

Simpler metabolic model

The model used to determine the metabolism of all radionuclides other than those of the transuranium elements, strontium, iodine and caesium, is illustrated schematically below:

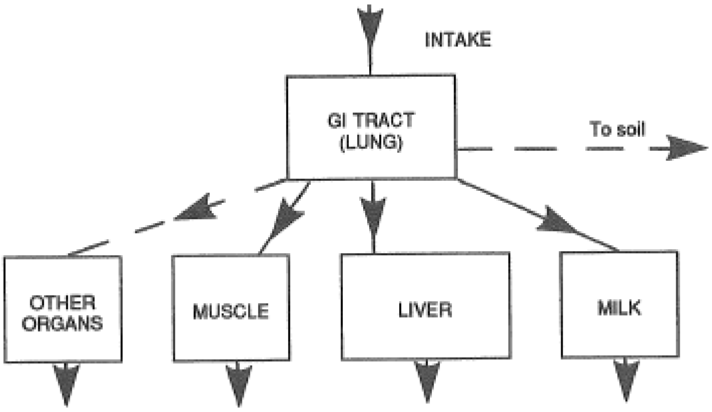

Diagram 3.7 Schematic representation of the simpler metabolic model

The derivation of transfer coefficients is described in Brown and Simmonds (1995) and is only briefly considered here. Most data are for cattle and are in the form of the fraction of the daily intake (assumed to be continuous) ingested appearing in unit mass of meat or milk. The origin and reliability of the data are variable; for example, some data are based on experiments with cattle whereas others are extrapolations from experiments with other species or even derived by analogy with the metabolism of other elements. For a comprehensive compilation of the available data see Brown and Simmonds (1995) and Ng et al (1982). The data adopted in this study are summarised in Table 3.21; the fraction of the daily intake (assumed to be continuous) of stable elements appearing in unit mass of meat and milk are given together with the biological half-life in animal muscle. These data can be used to derive the transfer coefficients for use in the model. Similar data are given in Table 3.21 for transfer to liver and are derived from data recommended for man (Phipps et al, 1991) assuming the fractional transfer from body fluids in liver is the same in cattle as in man. For those elements where no data are available for transfer to liver the concentration and half-life of activity in liver are taken to be equal to those in muscle, although it is recognised that an alternative approach would be to make use of the physiological similarity between elements.

Few data are available on the fractional transfer of inhaled activity to the body organs of grazing animals. The values used here have been derived from metabolic data recommended for man (Phipps et al, 1991; Silk et al, 1997). The fraction of the daily intake by inhalation appearing in a particular organ or milk, from a grazing animal, $F(inh)_c$ is obtained as

$$F(inh)_c = \frac{f(inh)_M}{f(ing)_M} F(ing)_c \qquad (3.35)$$

where $f(inh)_M$ is the fraction of inhaled activity reaching body fluids in man,

$f(ing)_M$ is the fraction of ingested activity reaching body fluids in man,

and $F(ing)_c$ is the fraction of the ingested daily intake appearing in the organ of or milk from cattle (see Table 3.21).

The fractions of ingested or inhaled activity reaching body fluids in man depend on the physico-chemical form of the element considered. However, default forms of each element are assumed for use in PC-CREAM 08 and when inhaled to be in the form of a 1 μm AMAD aerosol. It is further assumed that the transfer of activity across the lung occurs instantaneously; in reality, depending on the compound inhaled, the time constant for transfer may be days, weeks or years. This assumption will overestimate the transfer across the lung of cattle particularly for radionuclides with radioactive half-lives short compared to the time constant for transfer across the lung. The assumption is, however, conservative and considered justified, bearing in mind the many other uncertainties in the data used in the model.

More complex metabolic model

The model used to describe the metabolism of the transuranium elements in grazing animals is illustrated schematically below.

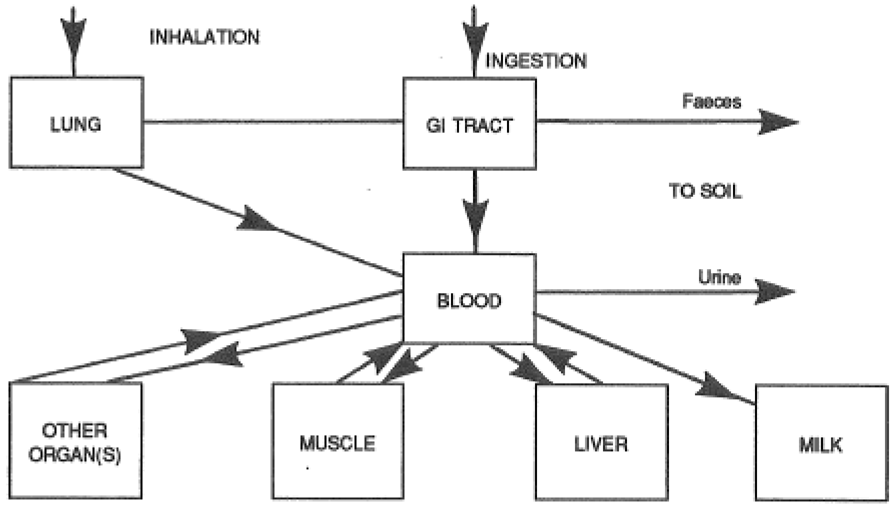

Diagram 3.8 Schematic representation of the more complex metabolic model

Account is taken of the transfer of material to various organs and of the recycling of activity between these organs and body fluids. Organs other than muscle and liver are included where they have a significant effect on animal metabolism and influence the time dependence of the transfer to meat and milk. For the transuranium elements, bone is included as a single organ; while for strontium, bone is again included but a distinction is made between the bone volume and surface to enable the different retention and re-cycling of strontium from the two components to be modelled. For caesium, two different compartments are used to represent soft tissues that are subsequently used for meat; this allows for part of the caesium to be transferred rapidly and part to be retained for a longer period. Iodine concentrates in the thyroid and so this was included as an organ in the metabolic model for iodine which also distinguishes between organic and inorganic iodine in the body.

The transfer of radionuclides from the lung and the GI tract of the animal is based, with simplification where appropriate, on the models recommended by ICRP for man; these models are outlined in Chapter 6 and their modification and simplification for application to grazing animals is discussed elsewhere (Brown and Simmonds, 1995). The time constants and the fraction of each radionuclide transferred from the animal lung and GI tract are based on data given for man assuming each radionuclide to be in the oxide form and as a 1 μm AMAD aerosol when inhaled.

The model for the transuranium elements is based on measurements of the distribution of plutonium among the various organs in cows. These data were used to derive transfer coefficients for the above model as described elsewhere (Brown and Simmonds, 1995; Linsley et al, 1979). In the absence of further data, these transfer coefficients are assumed to be equally valid for the radioisotopes of americium, curium and neptunium. The transfer coefficients derived for cows are also assumed applicable to sheep.

For strontium, caesium and iodine a variety of experimental data were used to derive transfer coefficients for the metabolic models. Cattle and sheep were considered separately. The data and the derivation of the appropriate transfers for the models are described in Brown and Simmonds (1995). As discussed earlier, pigs and chickens are not currently included in PC-CREAM 08.

3.3.3.5 Application of the foodchain models and a summary of selected results

The foodchain models have been applied to evaluate the time dependence of the transfer of activity to foodstuffs following the continuous deposition of activity on land for a year at a rate of 1 Bq m^{-2} s^{-1}. A matrix of results has been generated which contains the time integral of each radionuclide per unit mass of food derived from such land. This matrix (when combined with intake rates or the spatial distributions of agricultural yields, and deposition rates of radionuclides) forms the basis of estimates of the time dependent transfer of activity to man via terrestrial foodstuffs following the release of activity to atmosphere. Some selected results are presented and features of interest noted. Separate consideration is given to transfer to plants and animal produce.

(a) Transfer to plants

Results are given in Table 3.22 for the transfer of seven radionuclides to plants. The results are expressed in terms of the time integrals of activity per unit mass of plant (Bq y kg^{-1}) to various times after the deposition commenced and to infinity.

For surface plants, (green vegetables, grain and fruit) a large fraction of the integrated activity is accumulated in the first year primarily as a result of deposition of activity onto plant surfaces. Only in the case of strontium-90 does the time integral of activity increase significantly in subsequent years; this is due to the relatively high rate of root absorption of strontium from soils. For the short-lived iodine-131 ($t_{½}$ ~ 8 days) there is, as expected, no further contribution after the first year when the deposition has ceased.

In root crops, the time integrals of activity continue to increase while activity remains in the root zone. After the first year the only mechanism of importance for transfer of activity to root crops is absorption from the soil; absorption continues until activity is removed either by migration out of the root zone or by radioactive decay.

For plutonium-239, americium-241 and ruthenium-106 the time integral of activity in root crops is significantly lower than that in the surface crops. This is because the translocation from the surface of the plants to the root is negligible for these radionuclides. In contrast, for iodine-129 and -131, together with caesium-137, translocation is significant and concentrations in root crops are of the same order or greater than in other crop types.

The relative importance and time dependence of the important transfer mechanisms for strontium-90, caesium-137 and plutonium-239 to grain and green vegetables are illustrated in Figures 3.6, 3.7 and 3.8, respectively.

The mechanisms considered are surface contamination from direct deposition, initial resuspension before the surface layer of the deposition becomes uniformly mixed during cultivation, the subsequent resuspension of soil and absorption by root uptake. Initial resuspension is comparatively insignificant in all cases.

For strontium-90 the absorption from soil by the roots of plants becomes an increasingly important transfer mechanism with time. The time integral of activity by root uptake exceeds that due to direct deposition within a few years of deposition ceasing for both green vegetables and grain. This time integral continues to increase for over 100 years, at which time the contribution exceeds that due to direct deposition by about an order of magnitude. The eventual decline in root uptake of strontium-90 is predominantly due to its radioactive decay. Therefore, the transfer of strontium-90 to crops is not particularly sensitive to the uncertainty in rate of migration out of the root zone.

Plutonium-239 is relatively poorly absorbed from soil into plants and the direct deposition of activity makes by far the greatest contribution to the time integrals of activity in green vegetables and grain. Root uptake and resuspension of soil continue to contribute to the time integrals of activity for extended periods until the activity has migrated from the root zone. Considerable uncertainty is associated with the assumed half-life of 100 years for the migration of activity from the root zone. The total transfer of plutonium-239 to green vegetables and grain is, however, determined by direct deposition and the results are therefore relatively insensitive to the assumed half-life for migration. If crops growing on previously contaminated soils were being considered then the uncertainty in the rate of migration in soil would be important.

Most radionuclides considered in this study fall between the extremes represented by plutonium-239 and strontium-90 in terms of the relative importance of root uptake. This is seen in Figure 3.7 for caesium. For long-lived radionuclides such as technetium-99 and iodine-129 with significant root uptake rates, the assumed rate of migration of activity from surface soil will clearly have an impact on the time integral of activity at long times. However, for most of the radionuclides considered in this study the major contribution to the time integral of activity in green vegetables and grain is the direct deposition on plant surfaces. For caesium-137 the time integral due to direct deposition is greater than for strontium-90, which is itself greater than for plutonium-239. This reflects the level of translocation from the plants surface to the internal part of the plant which is greatest for caesium.

(b) Transfer to animal products

The time integrals of activity in meat, liver and milk derived from cows grazing pasture on which activity is deposited continuously at a rate of 1 Bq m^{-1} s^{-1} for one year are given in Table 3.23 for selected radionuclides. Similar results are given for sheep in Table 3.24; while the absolute values of the respective results differ they both exhibit the same general characteristics. No account is taken in the time integrals given in Tables 3.23 and 3.24 of the inhalation of activity by the animals while it is being deposited onto pasture. The most notable feature in Tables 3.23 and 3.24 is that by far the majority of activity is transferred to the various food products within about 50 y of the initial deposition with little further transfer later. For short lived radionuclides the transfer is essentially complete within a short time of the deposition process terminating (for example, see iodine-131). The temporal variation of the time integral of activity differs between radionuclides; this is a consequence of the varying relative importance of the different processes which contribute to the transfer of activity to the animal.

The time dependence and relative importance of the various transfer processes are illustrated in Figures 3.9, 3.10 and 3.11 for the transfer of strontium-90, caesium-137 and plutonium-239, respectively, to cow muscle. The transfer to other products would show similar characteristics. The processes considered are intake of pasture grass contaminated by direct deposition, resuspension and root uptake, intake by the inadvertent ingestion of soil, and the inhalation of resuspended activity.

Direct deposition onto, and root uptake by, pasture grass make by far the greatest contribution to the transfer of strontium to animal products. After about 3 years the contribution due to root uptake is the greatest for strontium-90 as seen in Figure 3.9. The contribution due to inadvertent ingestion of soil is comparatively small and that due to inhalation of resuspended material is negligible. Due to the low levels from the latter pathway its contribution could not be shown in Figure 3.9.

The time dependence of the transfer of strontium-90 to meat varies according to the transfer process. The transfer by direct deposition onto grass is essentially complete after about three years, two years after deposition ceases. The transfer continues due to the re-mobilisation of strontium deposited in the animals' bone but only for a short time. Root uptake continues for a longer period while strontium-90 is found in the top 15 cm of soil. The time variation of transfer by root uptake is determined by a combination of the migration from the top 15 cm of soil and the radioactive half-life of strontium-90 of 29.1 years.

For caesium-137 the total transfer to animal products is dominated by direct deposition onto pasture at all times. Root uptake and the inadvertent ingestion of soil make similar contributions to the total transfer, particularly at long times when fixation has reduced the amount of caesium available for root uptake (see section 3.3.3.3). The inhalation of resuspended material is again relatively unimportant.

The results for plutonium-239 shown in Figure 3.11 show that, as for caesium-137, direct deposition onto pasture grass dominates the total transfer to animal products. However, the contribution from root uptake is much smaller for plutonium than for caesium. For plutonium inadvertent ingestion is significantly more important than root uptake.

Plutonium has a long biological half-life in liver and bone leading to its remobilisation and subsequent transfer to meat, and milk. This process occurs over a lengthy period after intake. It is effectively only stopped at the animal's death, which is assumed to be 6 years for cattle. These factors are reflected in the time dependence of the transfers illustrated in Figure 3.11.

The inhalation by animals of activity as it is being deposited on pasture represents a further route leading to the contamination of animal products, however, this is not modelled in PC-CREAM 08. It is only likely to be significant for those radionuclides which have a low transfer in the terrestrial environment and across the animal's gut. However, even for plutonium which is of this type, the transfer to animal products due to inhalation is small compared to that due to ingestion.

3.3.3.6 The transfer of tritium and carbon-14 in the terrestrial environment

The transfer of tritium and carbon-14 between the atmosphere and the terrestrial environment is somewhat more complex than that described and modelled for other radionuclides in previous sections. This additional complexity is primarily a consequence of the fundamental roles played by hydrogen and carbon in biological systems. The models described earlier are not appropriate for tritium and carbon-14 and a relatively simple specific activity approach is adopted to evaluate the transfer through the terrestrial environment to man.

It can be assumed that the terrestrial environment and man come into rapid equilibrium with the carbon-14 in the atmosphere and that the specific activity of carbon taken in by man by inhalation or ingestion is equal to that in the atmosphere at the point of interest. For a release of carbon-14 to atmosphere the specific activity can be determined by the degree of atmospheric dispersion (see section 3.2) and the carbon concentration in the atmosphere; the latter is taken as 0.15 g m^{-3}. A similar assumption can be made for tritium; the specific activity of tritium taken into the body can be taken as equal to that in atmospheric water vapour, again determined by the degree of atmospheric dispersion with the concentration of water vapour in the atmosphere taken as 8 g m^{-3} (annual average value). The intake of carbon-14 and tritium in man by inhalation and ingestion can be determined from the respective intakes of carbon and water (see section 5.2) by the various routes; the dose in man can then be determined from these intakes from the committed doses per unit intake given in Chapter 6.

An alternative approach, and the one used in PC-CREAM 08, is the use of the specific activity models for the estimation of tritium and carbon-14 activity concentrations in foodstuffs. These activity concentrations may then be used in the usual manner to estimate doses via food ingestion and activity concentrations in air predicted by the dispersion calculations used in the estimation of inhalation dose. Table 3.25 gives the factors relevant to the specific activity in food models in PC-CREAM 08.

A dynamic tritium model has been developed (Higgins et al, 1996) to predict the time-dependent behaviour of tritium in terrestrial foodchains. The model includes the formation of organically bound tritium (OBT) which is less mobile in the environment than tritiated water, the form assumed in the specific activity model. A comparison has been made between the results of the specific activity model and of the dynamic model run for a continuous and constant source. The comparison showed that predictions using the dynamic model are generally slightly higher than those for the specific activity model by up to 25% depending on the foodstuff, due mainly to the inclusion of OBT. However, in the case of grain, where the organic fraction is much larger, the dynamic model predicts a higher total tritium activity concentration by almost a factor of 10. The comparison has thus shown that it cannot, as previously thought, be assumed that the specific activity model for tritium will lead to a conservative estimate in all foodstuffs and that in the case of grain a multiplying factor of 10 may be used to account for OBT.

One important point to note about the specific activity models is the absence of information on the temporal distribution of the dose which in reality may be delivered over an extended period. It is implicit in the model that the dose is only delivered while the specific activity, and thus the discharge, is maintained.

3.3.3.7 The application of the models in southern Europe

The types of food grown and consumed together with the agricultural practices adopted show strong regional variations. The prevailing climate has a pronounced effect on agriculture and may affect the transfer processes outlined in this section of the report. Differences in the transfer of radioactivity in the foodchain between northern and southern France have been considered by Métivier and Bénédetti (1994). Large differences are found in the types of food produced and consumed. For example, there are few cattle in the south and little butter or cheese from cows' milk are produced or eaten. Cheese from goats' milk and oils, such as olive oil, are widely produced and eaten, however. There are also large differences in the times at which crops are planted and harvested and in the feeding regimes for animals. These differences in agricultural practice could be significant for accidental releases where there is a single deposit at a particular time of year (Sinnaeve and Gerber, 1991). However, for routine releases as considered in this study, the deposit is continuous throughout the year and such differences in agricultural practice are insignificant in estimating concentrations in terrestrial foods. It may, however, be important to consider the food appropriate for the region for example, goats rather than cattle.

Any differences in transfer parameters due to the region appear to be small compared with the variation due to other factors, such as soil type. Therefore in this study the default values provided are considered equally appropriate for southern and northern European conditions.

3.3.3.8 Use of the models when the source of the radioactivity is irrigation

Although this exposure route is not currently modelled in PC-CREAM 08 it is considered in this section and Chapter 4. Freshwater, into which radionuclides are discharged, may be used to irrigate agricultural land and provide a source of radionuclides for terrestrial foods. Various types of irrigation are widely used in the EC, notably in the drier southern areas. The types of irrigation methods adopted and their extent are discussed in detail by Métivier and Bénédetti (1994). The type of irrigation which has potentially the greatest radiological significance is spray irrigation. In this case large quantities of water are sprayed over crops leading to radionuclides depositing on both the crops and the soil. Other types of irrigation, for example, via channels, lead to radionuclides entering the soil only initially and then subsequently transferring to plants by root uptake and resuspension.

It is possible to use the models described in the previous sections to estimate the concentrations of radionuclides in terrestrial foods when the input is via irrigation. Spray irrigation may be treated as an atmospheric source with deposition onto plants and soil and the default parameter values may be used in the absence of other data. However, if the spray irrigation is carried out at a high rate, it is similar in effect to heavy rain and there will be reduced interception by plants (Müller and Pröhl, 1993; Sinnaeve and Gerber, 1991; Métivier and Bénédetti, 1994). It may then be appropriate to use a reduced interception factor. For other types of irrigation it is again possible to use models as described but with input only into the appropriate soil compartment and not directly onto plants.

In Métivier and Bénédetti (1994) an alternative method is given for estimating concentrations of radionuclides in terrestrial foods following irrigation. This method uses measured concentrations of radionuclides in foods related to the concentration in the irrigation water. These empirical relationships can then be used to estimate the concentrations in food for a particular case as described.

3.3.3.9 Validation and limitations of the terrestrial foodchain models given in this study

There have been numerous validation and verification studies carried out on FARMLAND since the model was conceived and reviews of several of the validation studies have led to model parameters being refined. The results of such studies are summarised in a review of such studies of environmental models (Simmonds, 1998) and are discussed in more detail in a summary of verification and validation work involving FARMLAND by Brown (1995). Additional studies (Linkov et al, 2006; Ould-Dada et al, 2006) were used in validation and verification of the new FARMLAND fruit model and a summary of how the model performs is included in the fruit model report (Brown and Sherwood, 2008).

The terrestrial foodchain models described in this report are provided for use in assessing the radiological impact of routine releases of radionuclides to the environment. The required endpoints are radiation doses on an annual basis or longer and so it is possible to use annual average data and approximations. The use of such averages may not be appropriate in other applications and although the models are flexible enough to be used in a variety of situations, the way they are used and the parameter values adopted may need to be modified.

In particular, agricultural practices such as times of planting and harvesting of crops are not modelled explicitly. Rather, most crops are assumed to grow for 120 days (150 days for fruit) with a 245 day (210 day for fruit) fallow period and continuous deposition throughout. For an accidental release, deposition occurs only over a limited time and the transfer through foodchains together with the associated effects, will vary markedly depending on when this occurs relative to the growing cycle of the crop (Simmonds, 1985a and 1985b; Sinnaeve and Gerber, 1991). It is therefore necessary to include times of planting and harvesting explicitly; the method of doing this for FARMLAND is described in Brown and Simmonds (1995). For grazing animals it may also be important to consider seasonal variations in their husbandry, in particular the use of stored feed (Simmonds, 1985b). Again FARMLAND can be modified to take such factors into account (Brown and Simmonds, 1995).

As discussed in section 3.3.3.3 no account is taken of the form of the deposit in modelling the interception and retention of radioactivity on the plant's surface. This is justified for continuous releases where the input to the model is the total deposit from dry and wet deposition processes and the interception factor is an appropriate average number. However, for single deposits which will occur as predominantly due to dry deposition, or due to wet deposition, it may be necessary to distinguish between the deposition types (Sinnaeve and Gerber, 1991). This could be included in the models outlined here through the use of appropriate interception factors. An alternative approach taking into account rainfall rate and other plant-related factors is adopted by ECOSYS (Müller and Pröhl, 1993) and could be adopted if such detailed modelling is required for a particular application.

3.3.4 External irradiation due to surface deposition

The external irradiation to which man is subjected from surface deposition has been evaluated, for atmospheric releases, for an undisturbed soil into which penetration of the activity occurs by natural processes. The external irradiation due to the contamination in the top 30 cm of soil has been calculated for photons emitted by the appropriate radionuclides. External irradiation by electrons is presented for a surface deposit. External irradiation due to the contamination in deeper soil (below 30 cm) has been neglected; it is almost zero for electrons and very small for photons compared with the external irradiation due to contamination in the upper layer (the top 30 cm). For strontium in PC-CREAM 08, account is taken of the more rapid downwards migration from the top cm of the soil; based on the work done originally to develop the pasture/soil models for strontium in Haywood et al (1980).

3.3.4.1 External irradiation due to photons

The external irradiation due to photons emitted by radionuclides distributed in soil has been evaluated by first calculating the absorbed dose in air at a height of 1 m above the soil surface. This is accomplished by calculating the contribution to the absorbed dose rate in air from the radionuclides in an annulus of the soil at a certain depth; integrating the dose from the annulus and then summing the dose contributions from the different soil layers gives the total absorbed dose rate in air (see Figure 3.12). The calculation takes into account build-up in the soil and in the air together with attenuation in the soil and in the air. The equation takes the form (Kowe et al, 2007):

$$dD(r,l,E) = \frac{PTE}{2}\left(\frac{\mu_{ab}}{\rho}\right)_{air} C_A B \left(\frac{rdr}{y^2}\right) e^{-\mu_s x} e^{-\mu_a (y-x)} \qquad (3.36)$$

where

dD(r,l,E)	is the component of dose in Gy per year due to photons of energy E MeV from an annular source between r and r+dr at a depth of l in the soil (see Figure 3.12),
l	is the depth of a disk annulus,
P	energy conversion constant ($1.6\ 10^{-12}$ J MeV^{-1}),
T	time conversion constant (3600 s h^{-1})
$(\mu_{ab} / \rho)_{air}$	mass absorption coefficient of air (m^2 kg^{-1}) for photons of energy E,
ρ	density of air (kg m^{-3}),
B	is the build up factor for scattered radiation,
C_A	is the number of photons of initial energy E (MeV) emitted per second per m^2 of surface and cm depth soil layer considered,
x, y and r	are lengths (metres) indicated in Figure 3.12,
μ_a and μ_s	are the linear attenuation coefficients for the photons of energy E in air and material respectively (m^{-1}).

The build-up factor is used to account for the effects of photon scattering which occur in the material of the contaminated layer, shielding layer and the air layer. This scatters photons from their initial trajectory, which was away from the reference point, back to the reference point. The amount of scattering and thus the size of the build-up factor for each annulus is dependent on the shield material, and naturally varies with photon energy and attenuation depth of the annulus. Generally, build-up is higher for low atomic number materials. The air build up library is taken from Kowe et al (2007). The build-up data for various energies are taken from Tables B2 to B6 of Appendix B of Chilton (1968).

To calculate the exposure from a whole layer of contaminated material, the above equation (the dose from a potentially buried annulus) requires integration over the radius of the annulus (r = 0 to infinity) and over the vertical thickness of the layer. If equation 3.36 is written in terms of a variable φ, the number of mean free paths (though air and material) through which a gamma ray must pass in travelling from dr at a depth l to the reference point at height h above the material air interface, and variable dependencies are included, the following is obtained (Kowe et al, 2007):

$$D = \frac{PTE}{2}\left(\frac{\mu_{ab}}{\rho}\right)_{air} C_A \int_{l_1}^{l_2} \int_{c(h+l)}^{\infty} B(\varphi,l,E)\frac{e^{-\varphi(l,E)}}{\varphi(l,E)} d\varphi\, dl \qquad (3.37)$$

where

$$\phi = \left(\frac{h\mu_a + l\mu_s}{h+l}\right)y$$

Attenuation in soil is calculated from information on the major elements in soil and cross-section data for these elements for a range of photon energies. Calculations for a particular radionuclide take into account the photons of different energy emitted and their intensities.

The dose conversion library used in PC-CREAM 08 is based on ICRP Publication 74 (1996). The library considers rotational irradiation geometry. Rotational irradiation geometry has been shown to be the most appropriate for exposure in a standing position from contamination deposited to ground (Jacob et al, 1986). The library provides air absorbed dose conversions (Gy cm^2), whole body dose conversions (effective dose or effective dose equivalent) (Sv cm^2) and tissue or organ dose conversions (equivalent dose or dose equivalent) for four tissues (organs), gonads, breast, thyroid and skin (Sv cm^2). The ICRP Publication 74 dose conversion factors are shown in Table 3.26. Integrated effective doses and skin doses following deposition of some of the radionuclides considered in the terrestrial models are given in Table 3.27.

Individuals indoors will be protected to some degree from gamma-emitting material deposited on the ground outside and on the building in which they are located and on adjacent buildings. The degree of protection will depend upon the thicknesses of the walls, floors and roofs, the window area and the location of the individual within the building (Brown, 1988; Brown and Jones, 1993). Most estimates of the protection afforded by buildings are based upon theoretical studies. Studies suggest that the protection is different for material deposited under dry conditions as opposed to that deposited under wet conditions. Models which simulate the movement of activity on surfaces in the urban environment and calculate the external doses to individuals within buildings have been developed (Crick and Brown, 1990). These models could be used for site-specific studies and to obtain parameter values for use in generic studies. In PC-CREAM 08, the gamma dose rate indoors from deposited activity is assumed to be 0.1 of the dose rate outdoors in a rural environment for the same deposition density on soil.

The model described above is implemented in GRANIS and included in PC-CREAM 08. The stand-alone version of this code has been compared with analytical methods, a number of computer codes and published results. The differences between GRANIS and the other codes are generally less than a factor of two across the energy spectrum 0.1 to 4 MeV. For more information on this see Kowe et al (2007). The model for the migration of radionuclides in soil forms part of the GRANIS external dose model. Busby (1998) demonstrated that the soil model could successfully predict radionuclide concentration at depth in soil over the time period of available data.

3.3.4.2 External irradiation due to electrons

External irradiation from beta-emitters deposited on the ground will be dominated by the activity close to the surface, due to the small range of the electrons in soil. At energies below about 1 MeV attenuation in the air is also significant, resulting in large variations in absorbed dose rate in air with height above the surface. Dose rates to the basal layer of the skin, that is at a depth of 70 µm, have been computed for a height 0.8 m above the surface for a range of radionuclides deposited on the surface (Holford, 1989). Values for some of the radionuclides considered in the terrestrial models are given in Table 3.28. The variation in dose with height above the surface means that these values should be used with caution. The values presented in Table 3.28 do not take into account any shielding provided by clothing or nearby objects or buildings. The uncertainty in the application of these dose rates to real situations must be acknowledged. The calculation of the external dose due to beta radiation is performed in the ASSESSOR part of PC-CREAM 08.

3.4 Representation of the distribution of population and agricultural production within the European Union

The evaluation of collective doses due to atmospheric releases of radioactive material requires information on population distribution and agricultural production in the area of interest. The data used in PC-CREAM 08 are the same as those used in PC-CREAM 98 and have been derived from data held in the accident consequence model PC COSYMA (Jones et al, 1995). The PC COSYMA data are based on gridded data that have been generated by the Commissariat à l'Energie Atomique (CEA, France) using a latitude and longitude co-ordinate system. The data and methods used to generate the CEA grids are discussed in Bonnefous and Déspres (1991); Bonnefous and Déspres (1992); Bonnefous and Leonard (1989); and Simmonds, Lawson and Mayall (1995). To carry out an assessment of collective doses PC-CREAM 08 requires the population distribution and production of fresh milk, milk products, cow meat, cow liver, sheep meat, sheep liver, green vegetables, root crops and grain in the form of a polar grid centred on the point of discharge. PC COSYMA (Jones et al, 1995) can be used to generate such polar grids from the underlying latitude and longitude grid.

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3.6 Tables

TABLE 3.1 The common radionuclides considered for release to the atmosphere

Nuclide	Half-life (days)	Nuclide	Half-life (days)
^{3}H	4.51E+03	^{122}Sb	2.70E+00
^{14}C	2.09E+06	^{124}Sb	6.02E+01
^{35}S	8.74E+01	^{125}Sb	1.01E+03
^{41}Ar	7.61E-02	^{129}I	5.73E+09
^{51}Cr	2.77E+01	^{131}I	8.04E+00
^{54}Mn	3.13E+02	^{132}I	9.58E-02
^{59}Fe	4.45E+01	^{133}I	8.67E-01
^{58}Co	7.08E+01	^{135}I	2.75E-01
^{60}Co	1.92E+03	^{131m}Xe	1.19E+01
^{65}Zn	2.44E+02	^{133}Xe	5.24E+00
^{85}Kr	3.91E+03	^{133m}Xe	2.19E+00
^{85m}Kr	1.87E-01	^{135}Xe	3.79E-01
^{87}Kr	5.30E-02	^{135m}Xe	1.06E-02
^{88}Kr	1.18E-01	^{137}Xe	2.67E-03
^{89}Kr	2.21E-03	^{138}Xe	9.84E-03
^{89}Sr	5.05E+01	^{134}Cs	7.53E+02
^{90}Sr	1.06E+04	^{137}Cs	1.10E+04
^{95}Nb	3.51E+01	^{140}Ba	1.27E+01
^{95}Zr	6.40E+01	^{140}La	1.68E+00
^{106}Ru	3.68E+02		

TABLE 3.2 Parameter values for use in calculating wind speed at source height

Terrain	Roughness length z_0 (m)	n
Sea, very short grass	0.01	0.14
Open grassland	0.04	0.17
Low lying crops, eg, root crops	0.1	0.20
Agricultural areas	0.3	0.26
Parks, open suburbia	0.4	0.28
Cities, woodlands	1 o 4	0.39 to 1.1

TABLE 3.3 Coefficients given by Hosker to derive the vertical standard deviation of the plume for the various stability categories

$$\sigma_z = \frac{a\,x^b}{1 + c\,x^d} F(z_o, x)$$

Stability category	a	b	c	d
A	0.112	1.06	5.38×10^{-4}	0.815
B	0.130	0.950	6.52×10^{-4}	0.750
C	0.112	0.920	9.05×10^{-4}	0.718
D	0.098	0.889	1.35×10^{-3}	0.688
E	0.0609	0.895	1.96×10^{-3}	0.684
F	0.0638	0.783	1.36×10^{-3}	0.672

Coefficients for the roughness correction factor, $F(z_o,x)$

Roughness length (m)	f	g	h	j
0.01	1.56	0.0480	6.25×10^{-4}	0.45
0.04	2.02	0.0269	7.76×10^{-4}	0.37
0.1	2.72	0	0	0
0.4	5.16	-0.098	18.6	-0.225
1.0	7.37	-0.0957	4.29×10^{3}	-0.60
4.0	11.7	-0.128	4.59×10^{4}	-0.78

$$F(z_o, x) = \ln\left(f\,x^g[1 + \{h\,x^j\}^{-1}]\right), \; z_o > 0.1\text{m}$$
$$F(z_o, x) = \ln\left(f\,x^g[1 + h\,x^j]^{-1}\right), \; z_o \le 0.1\text{m}$$

TABLE 3.4 Coefficients given by Doury to derive the vertical standard deviation of the plume for various diffusion conditions[1]

Diffusion condition	Time of travel, [2]t Seconds	M	k
Normal	$0 - 2.4\ 10^2$	0.42	0.814
	$2.4\ 10^2 - 3.28\ 10^3$	1.0	0.685
	$> 3.28\ 10^3$	20	0.5
Poor	All	0.2	0.5

Notes

1 The vertical standard derivation of the $\sigma_z = (Mt)^k$.

2 Time of travel is obtained as x/u, that is the distance travelled divided by the mean wind speed. Wind speeds of 1, 2 and 5 m s^{-1} are considered for normal diffusion and 1, 2 and 3 m s^{-1} for poor diffusion conditions.

TABLE 3.5 Typical values of wind speed and depth of mixing layer for use when measured values are not available[1]

(a) Pasquill/Smith/Hosker scheme

Stability category	Typical wind speed at 10 m (m/s)	Typical mixing layer depth (m)
A	1	1300
B	2	900
C	5	850
D	5	800
E	3	400
F	2	100
G	1	100

(b) Doury scheme

Dispersion condition	Mean wind speed, m s^{-1}	Depth of boundary layer (m)
Poor	1,2,3	200
Normal	1,2	2000
Normal	5	1000

Note

1 From Clarke (1979) and Jones (1980)

TABLE 3.6 Values of deposition velocity and washout coefficient (PC-CREAM 08 defaults in brackets)

Type of material	Deposition velocity[1] m s^{-1}	Washout coefficient[2] s^{-1}
Noble gas	0 (0)	0 (0)
Reactive gas	10^{-3} to 10^{-2} (10^{-2})	3.10^{-5} to 3.10^{-4} (10^{-4})
Particles ~ 1 μm AD	10^{-4} to 10^{-3} (10^{-3})	3.10^{-5} to 3.10^{-4} (10^{-4})
Particles ~ 10 μm AD	10^{-2} to 3.10^{-2}	3.10^{-5} to 3.10^{-4}

Notes

1 Applicable to neutral stability and deposition to grass

2 For a rainfall rate of 1 mm h^{-1}

TABLE 3.7 Fractions of material remaining in the plume due to dry deposition for a deposition velocity of 10^{-2} m s^{-1}

Effective release height - 30 m									
		Fraction left in plume							
Stability category	Wind speed at stack height (m s^{-1})	0.5 km	1 km	2 km	5 km	10 km	20 km	50 km	100 km
A	1.32	0.96	0.94	0.92	0.89	0.86	0.81	0.68	0.51
B	2.65	0.98	0.96	0.94	0.90	0.87	0.83	0.73	0.59
C	6.62	0.99	0.98	0.97	0.95	0.93	0.90	0.85	0.78
D	6.62	1.00	0.98	0.97	0.93	0.90	0.86	0.78	0.69
E	3.97	1.00	0.98	0.94	0.86	0.78	0.68	0.51	0.36
F	2.65	1.00	0.99	0.95	0.79	0.61	0.40	0.13	0.19
G	1.32	1.00	1.00	0.99	0.80	0.44	0.14	0.0068	0.000087
Effective release height = 70 m									
A	1.64	0.99	0.97	0.95	0.93	0.90	0.86	0.75	0.59
B	3.28	1.00	0.99	0.97	0.94	0.92	0.88	0.80	0.67
C	8.21	1.00	1.00	0.99	0.97	0.96	0.93	0.89	0.83
D	8.21	1.00	1.00	0.99	0.97	0.94	0.91	0.84	0.77
E	4.93	1.00	1.00	0.99	0.95	0.89	0.81	0.65	0.49
F	3.28	1.00	1.00	1.00	0.98	0.89	0.68	0.28	0.060
G	1.64	1.00	1.00	1.00	1.00	0.97	0.79	0.23	0.081
Effective release height = 100 m									
A	1.80	0.99	0.98	0.97	0.94	0.92	0.88	0.77	0.62
B	3.60	1.00	0.99	0.98	0.96	0.93	0.90	0.82	0.70
C	8.99	1.00	1.00	0.99	0.98	0.97	0.95	0.91	0.85
D	8.99	1.00	1.00	1.00	0.98	0.96	0.93	0.87	0.80
E	5.40	1.00	1.00	1.00	0.98	0.94	0.85	0.72	0.55
F	3.60	1.00	1.00	1.00	1.00	0.94	0.76	0.33	0.083
G	1.80	1.00	1.00	1.00	1.00	1.00	0.94	0.44	0.056

TABLE 3.8 Criteria for neglecting orographic effects[1,2,3]

Neutral and unstable conditions

(i) the gradient of the surrounding terrain should be less than about 1 in 10

(ii) for a ridge upwind of the source
either $h > 1.5\ H$
or $x > 20\ H$ in neutral conditions
$x > 10\ H$ in very unstable conditions

(iii) for an isolated hill upwind of the source
either $h > 1.5\ H$
or $x > 7\ H$

(iv) for a hill or ridge downwind of the source
either $h > H + \sigma_z\ (x)$
or $\sigma_z\ (x) > H$

In stable conditions

(i) the gradient of the surrounding terrain should be less than about 1 in 100

(ii) for an obstacle upwind of the source
either $h > H$
or $x > 40\ H$ in slightly stable conditions
$x > 100\ H$ in very stable conditions

(iii) for an obstacle downwind of the source
either $h > H + \sigma_z\ (x)$
or $\sigma_z\ (x) > H$

Notes

1 The criteria are based on a change of 30% in the wind speed between flat and complex terrain and the 10 m wind speed must be at least 1 m s^{-1}.

2 h = effective release height.
H = obstacle height.
x = distance between obstacle and source.

3 From Jones (1986).

TABLE 3.9 Factors for Calculating Effective Dose from Photon Fluence[1]

Photon energy (Mev)	Absorbed dose in air per unit fluence (Gy m^2 /photon) (ICRP 74, p159)	Ratio of effective dose to air dose for each energy for isotropic irradiation geometry (Sv Gy^{-1}) (ICRP 74, p175)
0.01	$7.43\ 10^{-16}$	$2.71\ 10^{-3}$
0.015	$3.12\ 10^{-16}$	$1.23\ 10^{-2}$
0.02	$1.68\ 10^{-16}$	$3.62\ 10^{-2}$
0.03	$7.21\ 10^{-17}$	$1.43\ 10^{-1}$
0.05	$3.23\ 10^{-17}$	$5.11\ 10^{-1}$
0.1	$3.71\ 10^{-17}$	$7.48\ 10^{-1}$
0.2	$8.56\ 10^{-17}$	$6.79\ 10^{-1}$
0.5	$2.38\ 10^{-16}$	$6.75\ 10^{-1}$
1	$4.47\ 10^{-16}$	$7.19\ 10^{-1}$
1.5	$6.14\ 10^{-16}$	$7.46\ 10^{-1}$
2	$7.54\ 10^{-16}$	$7.74\ 10^{-1}$
4	$1.21\ 10^{-15}$	$8.24\ 10^{-1}$

Note

1 The derivation of these data is described in section 3.2.6.1.

Table 3.10 Beta dose rates per unit concentration in air [1]

Radionuclide	Cloud beta dose $(Sv\ y^{-1})/(Bq\ m^{-3})$	Radionuclide	Cloud beta dose $(Sv\ y^{-1})/(Bq\ m^{-3})$
^{3}H	-	^{132}I	$9.51\ 10^{-7}$
^{14}C	$7.86\ 10^{-9}$	^{133}I	$7.70\ 10^{-7}$
^{35}S	$9.70\ 10^{-9}$	^{135}I	$6.81\ 10^{-7}$
^{41}Ar	$8.92\ 10^{-7}$	^{131m}Xe	$1.28\ 10^{-7}$
^{51}Cr	-	^{133m}Xe	$2.67\ 10^{-7}$
^{54}Mn	-	^{133}Xe	$9.00\ 10^{-8}$
^{59}Fe	$1.26\ 10^{-7}$	^{135m}Xe	$1.85\ 10^{-7}$
^{58}Co	$4.41\ 10^{-8}$	^{135}Xe	$5.51\ 10^{-7}$
^{60}Co	$7.92\ 10^{-8}$	^{137}Xe	$3.92\ 10^{-6}$
^{65}Zn	$2.26\ 10^{-9}$	^{138}Xe	$1.26\ 10^{-6}$
^{85}Kr	$4.24\ 10^{-7}$	^{134}Cs	$2.39\ 10^{-7}$
^{85m}Kr	$4.35\ 10^{-7}$	^{137}Cs	$2.36\ 10^{-7}$
^{87}Kr	$2.86\ 10^{-6}$	^{140}Ba	$4.76\ 10^{-7}$
^{88}Kr	$6.73\ 10^{-7}$	^{140}La	$1.05\ 10^{-6}$
^{89}Kr	$2.97\ 10^{-6}$	^{141}Ce	$1.92\ 10^{-7}$
^{89}Sr	$1.18\ 10^{-6}$	^{144}Ce	$6.19\ 10^{-8}$
^{90}Sr	$3.00\ 10^{-7}$	^{238}Pu	-
^{95}Zr	$1.21\ 10^{-7}$	^{239}Pu	-
^{95}Nb	$8.49\ 10^{-9}$	^{240}Pu	-
^{106}Ru	-	^{241}Pu	-
^{122}Sb	$1.13\ 10^{-6}$	^{241}Am	$2.86\ 10^{-11}$
^{124}Sb	$7.11\ 10^{-7}$	^{242}Cm	-
^{125}Sb	$9.08\ 10^{-8}$	^{243}Cm	$8.54\ 10^{-8}$
^{129}I	$4.81\ 10^{-9}$	^{244}Cm	
^{131}I	$2.73\ 10^{-7}$		

Note

1 USDOE (1988)

TABLE 3.11 Activity concentration in air (Bq m^{-3}) for a discharge of 1 Bq s^{-1} at a release height of 30 m

Stability category: Pasquill D

	Distance (m)			
Radionuclide	$1.00\ 10^{3}$	$1.00\ 10^{4}$	$1.00\ 10^{5}$	$1.00\ 10^{6}$
^{14}C	$3.88\ 10^{-7}$	$8.90\ 10^{-9}$	$3.00\ 10^{-10}$	$3.00\ 10^{-11}$
^{60}Co	$3.85\ 10^{-7}$	$8.80\ 10^{-9}$	$2.91\ 10^{-10}$	$2.45\ 10^{-11}$
^{90}Sr	$3.85\ 10^{-7}$	$8.80\ 10^{-9}$	$2.91\ 10^{-10}$	$2.46\ 10^{-11}$
^{131}I	$3.82\ 10^{-7}$	$7.95\ 10^{-9}$	$2.16\ 10^{-10}$	$3.47\ 10^{-12}$
^{137}Cs	$3.85\ 10^{-7}$	$8.80\ 10^{-9}$	$2.91\ 10^{-10}$	$2.46\ 10^{-11}$
^{239}Pu	$3.85\ 10^{-7}$	$8.80\ 10^{-9}$	$2.91\ 10^{-10}$	$2.46\ 10^{-11}$
^{241}Am	$3.85\ 10^{-7}$	$8.80\ 10^{-9}$	$2.91\ 10^{-10}$	$2.46\ 10^{-11}$

Note

Deposition velocity = 10^{-3} ms^{-1} for all radionuclides except for:

^{131}I = 10^{-2} ms^{-1}

^{14}C = 0.0 ms^{-1}

TABLE 3.12 Deposition rates (Bq m^{-2} s^{-1}) for a discharge of 1 Bq s^{-1} at a release height of 30 m

Stability category: Pasquill D

	Distance (m)			
Radionuclide	$1.00\ 10^{3}$	$1.00\ 10^{4}$	$1.00\ 10^{5}$	$1.00\ 10^{6}$
^{14}C	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
^{60}Co	$3.88\ 10^{-10}$	$8.80\ 10^{-12}$	$2.91\ 10^{-13}$	$2.45\ 10^{-14}$
^{90}Sr	$3.88\ 10^{-10}$	$8.80\ 10^{-12}$	$2.91\ 10^{-13}$	$2.46\ 10^{-14}$
^{131}I	$3.82\ 10^{-9}$	$7.95\ 10^{-11}$	$2.16\ 10^{-12}$	$3.47\ 10^{-14}$
^{137}Cs	$3.88\ 10^{-10}$	$8.80\ 10^{-12}$	$2.91\ 10^{-13}$	$2.46\ 10^{-14}$
^{239}Pu	$3.88\ 10^{-10}$	$8.80\ 10^{-12}$	$2.91\ 10^{-13}$	$2.46\ 10^{-14}$
^{241}Am	$3.88\ 10^{-10}$	$8.80\ 10^{-12}$	$2.91\ 10^{-13}$	$2.46\ 10^{-14}$

Note

Deposition velocity = 10^{-3} ms^{-1} for all radionuclides except for:

^{131}I = 10^{-2} ms^{-1}

^{14}C = 0.0 ms^{-1}

TABLE 3.13 Finite cloud gamma dose (Sv y^{-1}) for a discharge of 1 Bq s^{-1} at a release height of 30 m

Stability category: Pasquill D

	Distance (m)			
Radionuclide	$1.00\ 10^{3}$	$1.00\ 10^{4}$	$1.00\ 10^{5}$	$1.00\ 10^{6}$
^{14}C	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
^{60}Co	$6.88\ 10^{-13}$	$2.74\ 10^{-14}$	$1.07\ 10^{-15}$	$8.99\ 10^{-17}$
^{90}Sr	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
^{131}I	$1.12\ 10^{-13}$	$3.72\ 10^{-15}$	$1.11\ 10^{-16}$	$1.77\ 10^{-18}$
^{137}Cs	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
^{239}Pu	$2.64\ 10^{-17}$	$7.70\ 10^{-19}$	$2.66\ 10^{-20}$	$2.25\ 10^{-21}$
^{241}Am	$6.97\ 10^{-15}$	$2.03\ 10^{-16}$	$6.91\ 10^{-18}$	$5.84\ 10^{-19}$

Note

Using the finite cloud model

TABLE 3.14 Cloud beta equivalent dose rates in skin (Sv y^{-1}) for a discharge of 1 Bq s^{-1} at a release height of 30 m

Stability category: Pasquill D

	Distance (m)			
Radionuclide	$1.00\ 10^{3}$	$1.00\ 10^{4}$	$1.00\ 10^{5}$	$1.00\ 10^{6}$
^{14}C	$3.05\ 10^{-15}$	$7.00\ 10^{-17}$	$2.36\ 10^{-18}$	$2.36\ 10^{-19}$
^{60}Co	$3.05\ 10^{-14}$	$6.97\ 10^{-16}$	$2.30\ 10^{-17}$	$1.94\ 10^{-18}$
^{90}Sr	$1.16\ 10^{-13}$	$2.64\ 10^{-15}$	$8.73\ 10^{-17}$	$7.37\ 10^{-18}$
^{131}I	$1.04\ 10^{-13}$	$2.17\ 10^{-15}$	$5.89\ 10^{-17}$	$9.48\ 10^{-19}$
^{137}Cs	$9.10\ 10^{-14}$	$2.08\ 10^{-15}$	$6.88\ 10^{-17}$	$5.81\ 10^{-18}$
^{239}Pu	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
^{241}Am	$1.10\ 10^{-17}$	$2.52\ 10^{-19}$	$8.34\ 10^{-21}$	$7.03\ 10^{-22}$

Table 3.15 Time integrated resuspended activity concentration in air for a deposition rate of 1 Bq m^{-2} s^{-1}

Time (y)	Time integrated resuspended activity concentration in air (Bq s m^{-3})						
	^{60}Co	^{90}Sr	^{103}Ru	^{131}I	^{137}Cs	^{239}Pu	^{241}Am
1	$2.31\ 10^{7}$	$2.35\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$2.35\ 10^{7}$	$2.36\ 10^{7}$	$2.36\ 10^{7}$
5	$2.96\ 10^{7}$	$3.21\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$3.22\ 10^{7}$	$3.28\ 10^{7}$	$3.28\ 10^{7}$
50	$3.49\ 10^{7}$	$6.14\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$6.19\ 10^{7}$	$8.50\ 10^{7}$	$8.29\ 10^{7}$
500	$3.49\ 10^{7}$	$7.47\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$7.60\ 10^{7}$	$5.37\ 10^{8}$	$3.83\ 10^{8}$
10000	$3.49\ 10^{7}$	$7.47\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$7.60\ 10^{7}$	$8.69\ 10^{9}$	$6.62\ 10^{8}$
1E+08	$3.49\ 10^{7}$	$7.47\ 10^{7}$	$1.48\ 10^{7}$	$9.56\ 10^{6}$	$7.60\ 10^{7}$	$3.46\ 10^{10}$	$6.62\ 10^{8}$

TABLE 3.16 Values of transfer coefficients used in the models for migration in soil

Transfer coefficient	Value in d^{-1}
Well mixed soil[2]	
k_{11}	$1.90\ 10^{-5}$
Undisturbed land[3]	
k_{12}	$6.64\ 10^{-4}$ [4]
k_{23}	$1.72\ 10^{-4}$
k_{34}	$1.07\ 10^{-4}$
k_{43}	$4.03\ 10^{-6}$
k_{45}	$3.80\ 10^{-5}$

Notes

1 Values from Brown and Simmonds (1995).

2 See diagram 3.3, section 3.3.3.2.

3 See diagram 3.4, section 3.3.3.2.

4 $k_{12} = 1.27\ 10^{-3}\ d^{-1}$ for strontium.

TABLE 3.17 Comparison of measured and predicted normalised specific activities (NSA) for crops

Crop/product	NSA ($m^2d\ kg^{-1}$ dry weight)					
	Strontium		Caesium		Plutonium	
	Based on measurements	Model prediction	Based on measurements	Model prediction	Based on measurements	Model prediction
Grain	2.7 - 4.5[2]	1.5	4.7 - 6.9[2]	7.1	0.52 - 0.8[2]	0.69
Flour	0.5 - 1.4[2]	0.9	5.6 - 6.2[2]	6.5	0.02 - 0.05[2]	0.07
Green vegetables:						
before processing	34 - 57[2]	30.1	-	30.1	-	30.3
after processing	3.5 - 16.8[2]	6.7	3.0 - 10.7[2]	7.7	0.27 - 0.78[2]	6.1
Root crops[1]	0.55 - 1.7	0.06	4.6 - 8.6	7.1	-	-
Fruit[3]	1.5 – 2.0	1.7	5.1 – 10.7	26.6	0.93	0.54

Notes

[1] See Brown and Simmonds (1995).

[2] See Simmonds and Linsley (1982).

[3] See Mayall (1995).

TABLE 3.18 Element independent parameters for crops and pasture

	Value				
Parameter	Green vegetables	Grain	Pasture	Potatoes	Fruit
Yield, fresh wt kg/km^2	$1\ 10^{6}$	$4\ 10^{5}$	$5\ 10^{5}$[6]	$3\ 10^{6}$	$1.69\ 10^{6}$
Interception Factor	0.3	0.3[1] 0.012[2]	0.25	0.4	0.74 (plant) 0.007 (fruit)
Half-life on plant surface, d	14	14[1] 14.4[2]	14[3] 28[4]	14	14 (plant) 14 (fruit)
Soil on plant surface % of dry plant weight	0.1[5]	0.01[5]	4	0.1[5]	- (plant) 0.1 (fruit)
Depth of soil, cm	30[7]	30[7]	15[8]	30[7]	30[7]
Fraction of activity retained after preparation[9]	0.2	0.1	-	1.0	1.0

Notes

1 Whole cereal plant.

2 Grain seed.

3 Summer.

4 Winter.

5 Before preparation and processing.

6 This is the yield from 3 harvests and is expressed as dry weight.

7 Depth of well mixed soil from which root uptake occurs.

8 Depth of undisturbed soil from which root uptake occurs.

9 Applies to surface contamination only.

TABLE 3.19 Equilibrium soil-to-plant concentration ratios (wet weight plant:dry weight soil)

Element	Crop					Trans-location[4]	Ref[1]
	Green vegetables	Grain	Potatoes	Pasture	Fruit		
Sulphur	6 10^{-1}	6 10^{-1}	6 10^{-1}	6 10^{-1}	6 10^{-1}	m	SC
Chromium	3 10^{-4}	3 10^{-4}	3 10^{-4}	3 10^{-4}	3 10^{-4}	i	SC
Manganese	1 10^{-1}	1 10^{-1}	1 10^{-1}	1 10^{-1}	1 10^{-1}	s	IUR
Iron	2 10^{-4}	4 10^{-4}	3 10^{-4}	4 10^{-4}	4 10^{-4}	s	SC
Cobalt	1 10^{-2}	5 10^{-3}	1 10^{-2}	1 10^{-2}	5 10^{-3}	s	IUR
Zinc	1	1	5 10^{-1}	1	1	s	IUR
Strontium	2 10^{-1}	1 10^{-1}	4 10^{-2}	5 10^{-2} [2]	2 10^{-2}	s	IUR +
Yttrium	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	s	NG
Zirconium	1 10^{-4}	1 10^{-4}	1 10^{-4}	1 10^{-4}	1 10^{-4}	s	EC
Niobium	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	s	SC
Ruthenium	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	i	EC
Antimony	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	1 10^{-2}	s	SC
Tellurium	3 10^{-3}	3 10^{-3}	1 10^{-3}	5 10^{-3}	3 10^{-3}	m	EC
Iodine	2 10^{-2}	2 10^{-2}	2 10^{-2}	2 10^{-2}	2 10^{-2}	m	EC
Caesium[3]	1 10^{-2}	1 10^{-2}	6 10^{-3}	3 10^{-2}	3 10^{-3}	m	IUR +
Barium	1 10^{-2}	1 10^{-2}	5 10^{-3}	1 10^{-2}	1 10^{-2}	s	EC
Lanthanum	3 10^{-3}	3 10^{-3}	3 10^{-3}	3 10^{-3}	3 10^{-3}	s	SC
Cerium	1 10^{-3}	1 10^{-3}	1 10^{-3}	1 10^{-3}	1 10^{-3}	i	EC
Plutonium	1 10^{-5}	2 10^{-5}	5 10^{-5}	1 10^{-4}	1 10^{-5}	i	IUR +
Americium	5 10^{-5}	5 10^{-5}	8 10^{-5}	1 10^{-3}	1 10^{-5}	i	IUR +
Curium	5 10^{-5}	2 10^{-5}	3 10^{-5}	1 10^{-3}	2 10^{-5}	i	IUR

Notes

1 Key to references: SC = Simmonds and Crick (1982); IUR = IUR (1989); NG = Ng (1982a, 1982b); EC = work done for EC (1991) but not published.

In addition (+):

Nisbet and Shaw (1994) for strontium, caesium and plutonium values.

Nisbet and Woodman (2000) for strontium and caesium values.

Green et al (1997) for strontium, caesium, plutonium and americium values.

Carini (2001) for strontium, caesium, plutonium and americium values (fruit only).

2 This value applies to uptake from the lower layers of soil, for the top 1 cm a value of 2 10^{-1} is appropriate.

3 For crops other than pasture, fixation of caesium is incorporated implicitly in the root uptake values.

4 Mobility of each radionuclide for translocation has been classified as mobile (m), semi-mobile (s) and immobile (i).

TABLE 3.20 Non-element dependent parameters for animals[1]

Parameter	Cattle	Sheep
Amount eaten per day (kg dry wet/d)		
pasture	13	1.5
grain	-	-
Mean life (y)	6	1
Soil consumption as % of dry matter intake	4	20
Weight of muscle (kg)		
simple model[2]	230[4]	18[4]
complex model[3]	360[5]	30[5]
Weight of liver (kg)		
simple model[2]	6	0.8
complex model[3]	6	1.0
Milk production rate (litres per day)	10	-
Number of animals per km^2	400	500
Inhalation rate m^3 s^{-1}	$1.5\ 10^{-3}$	$1.0\ 10^{-4}$

Notes

1 These values are taken from Brown and Simmonds (1995).

2 The simple model values are for all elements other than caesium, strontium and iodine.

3 The complex model values are for caesium, strontium and iodine only.

4 This is the carcass weight, the weight of lean meat is 150 kg for cattle and 15 kg for sheep.

5 This is the weight of all soft tissues, the weight of lean meat is 150 kg.

TABLE 3.21 Equilibrium transfer factors for cattle and sheep[1]

	Cattle			Sheep		Biological ½ life, y	
	F_m[2]	F_f meat[3]	F_f liver[4]	F_f meat[3]	F_f liver[4]	Meat	Liver
Sulphur	2 10^{-2}	3 10^{-1}	3 10^{-1}	5	2	3 10^{-1}	3 10^{-1}
Chromium	1 10^{-4}	5 10^{-3}	5 10^{-3}	5 10^{-2}	5 10^{-2}	9 10^{-2}	9 10^{-2}
Manganese	2 10^{-4}	5 10^{-4}	2 10^{-2}	5 10^{-3}	2 10^{-1}	6 10^{-2}	7 10^{-2}
Iron	5 10^{-5}	1 10^{-3}	4	1 10^{-2}	3 10^{1}	5	5
Cobalt	1 10^{-4}	1 10^{-4}	1 10^{-2}	1 10^{-3}	1 10^{-1}	5 10^{-1}	5 10^{-1}
Zinc	3 10^{-3}	2 10^{-3}	2 10^{-3}	2 10^{-2}	2 10^{-2}	8 10^{-1}	8 10^{-1}
Strontium	2 10^{-3}	3 10^{-4}	3 10^{-4}	3 10^{-3}	3 10^{-3}	-(5)	-(5)
Yttrium	2 10^{-5}	1 10^{-3}	1 10^{-2}	1 10^{-2}	1 10^{-1}	4 10^{1}	4 10^{1}
Zirconium	6 10^{-7}	1 10^{-7}	1 10^{-7}	1 10^{-6}	1 10^{-6}	2 10^{-2}	2 10^{-2}
Niobium	4 10^{-7}	3 10^{-7}	3 10^{-7}	3 10^{-6}	3 10^{-6}	3 10^{-1}	3 10^{-1}
Ruthenium	1 10^{-6}	1 10^{-3}	1 10^{-3}	1 10^{-2}	1 10^{-2}	7 10^{-1}	7 10^{-1}
Antimony	1 10^{-4}	1 10^{-3}	1 10^{-1}	1 10^{-2}	1	5 10^{-2}	5 10^{-2}
Tellurium	5 10^{-4}	5 10^{-3}	5 10^{-3}	5 10^{-2}	5 10^{-2}	5 10^{-2}	5 10^{-2}
Iodine	5 10^{-3}	2 10^{-3}	2 10^{-3}	5 10^{-2}	5 10^{-2}	-(5)	-(5)
Caesium	5 10^{-3}	3 10^{-2}	3 10^{-2}	5 10^{-1}	5 10^{-1}	-(5)	-(5)
Barium	5 10^{-4}	5 10^{-4}	5 10^{-4}	5 10^{-3}	5 10^{-3}	9 10^{-2}	9 10^{-2}
Lanthanum	2 10^{-5}	5 10^{-3}	2 10^{-1}	5 10^{-2}	2	1 10^{1}	1 10^{1}
Cerium	1 10^{-4}	1 10^{-3}	2 10^{-1}	1 10^{-2}	2	1 10^{1}	1 10^{1}
Plutonium	1 10^{-6}	1 10^{-4}	2 10^{-2}	4 10^{-4}	3 10^{-2}	-(5)	-(5)
Americium	1 10^{-6}	1 10^{-4}	2 10^{-2}	4 10^{-4}	3 10^{-2}	-(5)	-(5)
Curium	1 10^{-6}	1 10^{-4}	2 10^{-2}	4 10^{-4}	3 10^{-2}	-(5)	-(5)

Notes

1 Values taken from Sinnaeve and Gerber (1991); ERDA (1975); Ng (1968); Fletcher and Dotson (1971); Baes et al (1984); Adams et al (1978); Ng et al (1977); Voigt et al (1987).

2 F_m denotes the fraction of the daily intake by ingestion transferred to a litre of milk.

3 F_f denotes the fraction of the daily intake by ingestion transferred to a kg of muscle.

4 F_f denotes the fraction of the daily intake by ingestion transferred to a kg of liver.

5 Not required in the FARMLAND model used for this element.

TABLE 3.22 The time integrals of activity per unit mass of various crops

	Time integral of activity per unit mass of plant (Bq y kg^{-1}) [1,2]				
Nuclide	Times (days)	Grain	Green vegetables	Root veg	Fruit
^{90}Sr	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	6.89 10^{4}	1.15 10^{5}	1.03 10^{3}	2.29 10^{4}
	7.30 10^{2}	7.57 10^{4}	1.28 10^{5}	3.75 10^{3}	2.36 10^{4}
	1.83 10^{3}	9.47 10^{4}	1.66 10^{5}	1.14 10^{4}	2.56 10^{4}
	3.65 10^{3}	1.23 10^{5}	2.21 10^{5}	2.27 10^{4}	2.85 10^{4}
	7.30 10^{3}	1.68 10^{5}	3.09 10^{5}	4.07 10^{4}	3.32 10^{4}
	1.10 10^{4}	2.00 10^{5}	3.73 10^{5}	5.38 10^{4}	3.67 10^{4}
	1.83 10^{4}	2.42 10^{5}	4.53 10^{5}	7.06 10^{4}	4.11 10^{4}
	3.65 10^{4}	2.81 10^{5}	5.26 10^{5}	8.59 10^{4}	4.52 10^{4}
	1.83 10^{5}	2.91 10^{5}	5.44 10^{5}	9.01 10^{4}	4.63 10^{4}
	3.65 10^{7}	2.91 10^{5}	5.44 10^{5}	9.01 10^{4}	4.63 10^{4}
^{106}Ru	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	5.37 10^{3}	1.01 10^{5}	1.91 10^{2}	7.19 10^{3}
	7.30 10^{2}	5.73 10^{3}	1.01 10^{5}	5.55 10^{2}	7.36 10^{3}
	1.83 10^{3}	6.05 10^{3}	1.02 10^{5}	8.74 10^{2}	7.51 10^{3}
	3.65 10^{3}	6.09 10^{3}	1.02 10^{5}	9.17 10^{2}	7.53 10^{3}
	7.30 10^{3}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
	1.10 10^{4}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
	1.83 10^{4}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
	3.65 10^{4}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
	1.83 10^{5}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
	3.65 10^{7}	6.10 10^{3}	1.02 10^{5}	9.19 10^{2}	7.54 10^{3}
^{129}I	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	5.04 10^{5}	1.33 10^{5}	1.23 10^{5}	3.61 10^{5}
	7.30 10^{2}	5.05 10^{5}	1.34 10^{5}	1.24 10^{5}	3.61 10^{5}
	1.83 10^{3}	5.09 10^{5}	1.38 10^{5}	1.29 10^{5}	3.63 10^{5}
	3.65 10^{3}	5.16 10^{5}	1.45 10^{5}	1.35 10^{5}	3.67 10^{5}
	7.30 10^{3}	5.29 10^{5}	1.57 10^{5}	1.48 10^{5}	3.73 10^{5}
	1.10 10^{4}	5.40 10^{5}	1.69 10^{5}	1.60 10^{5}	3.79 10^{5}
	1.83 10^{4}	5.61 10^{5}	1.90 10^{5}	1.81 10^{5}	3.89 10^{5}
	3.65 10^{4}	6.03 10^{5}	2.31 10^{5}	2.22 10^{5}	4.09 10^{5}
	1.83 10^{5}	6.96 10^{5}	3.23 10^{5}	3.15 10^{5}	4.55 10^{5}
	3.65 10^{7}	7.02 10^{5}	3.29 10^{5}	3.20 10^{5}	4.58 10^{5}
^{131}I	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	7.30 10^{2}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	1.83 10^{3}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	3.65 10^{3}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	7.30 10^{3}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	1.10 10^{4}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}
	1.83 10^{4}	4.22 10^{4}	4.12 10^{4}	8.61 10^{3}	2.92 10^{4}

	Time integral of activity per unit mass of plant (Bq y kg^{-1}) ^{1,2}				
Nuclide	Times (days)	Grain	Green vegetables	Root veg	Fruit
	$3.65\ 10^{4}$	$4.22\ 10^{4}$	$4.12\ 10^{4}$	$8.61\ 10^{3}$	$2.92\ 10^{4}$
	$1.83\ 10^{5}$	$4.22\ 10^{4}$	$4.12\ 10^{4}$	$8.61\ 10^{3}$	$2.92\ 10^{4}$
	$3.65\ 10^{7}$	$4.22\ 10^{4}$	$4.12\ 10^{4}$	$8.61\ 10^{3}$	$2.92\ 10^{4}$
^{137}Cs	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$5.02\ 10^{5}$	$1.33\ 10^{5}$	$1.22\ 10^{5}$	$3.59\ 10^{5}$
	$7.30\ 10^{2}$	$5.03\ 10^{5}$	$1.33\ 10^{5}$	$1.23\ 10^{5}$	$3.59\ 10^{5}$
	$1.83\ 10^{3}$	$5.05\ 10^{5}$	$1.35\ 10^{5}$	$1.24\ 10^{5}$	$3.59\ 10^{5}$
	$3.65\ 10^{3}$	$5.08\ 10^{5}$	$1.38\ 10^{5}$	$1.26\ 10^{5}$	$3.60\ 10^{5}$
	$7.30\ 10^{3}$	$5.12\ 10^{5}$	$1.42\ 10^{5}$	$1.29\ 10^{5}$	$3.61\ 10^{5}$
	$1.10\ 10^{4}$	$5.16\ 10^{5}$	$1.46\ 10^{5}$	$1.31\ 10^{5}$	$3.61\ 10^{5}$
	$1.83\ 10^{4}$	$5.20\ 10^{5}$	$1.50\ 10^{5}$	$1.33\ 10^{5}$	$3.62\ 10^{5}$
	$3.65\ 10^{4}$	$5.24\ 10^{5}$	$1.54\ 10^{5}$	$1.36\ 10^{5}$	$3.62\ 10^{5}$
	$1.83\ 10^{5}$	$5.25\ 10^{5}$	$1.55\ 10^{5}$	$1.36\ 10^{5}$	$3.63\ 10^{5}$
	$3.65\ 10^{7}$	$5.25\ 10^{5}$	$1.55\ 10^{5}$	$1.36\ 10^{5}$	$3.63\ 10^{5}$
^{239}Pu	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$5.37\ 10^{3}$	$1.05\ 10^{5}$	$1.07\ 10^{0}$	$7.23\ 10^{3}$
	$7.30\ 10^{2}$	$5.38\ 10^{3}$	$1.05\ 10^{5}$	$4.55\ 10^{0}$	$7.24\ 10^{3}$
	$1.83\ 10^{3}$	$5.38\ 10^{3}$	$1.05\ 10^{5}$	$1.48\ 10^{1}$	$7.26\ 10^{3}$
	$3.65\ 10^{3}$	$5.39\ 10^{3}$	$1.05\ 10^{5}$	$3.15\ 10^{1}$	$7.29\ 10^{3}$
	$7.30\ 10^{3}$	$5.41\ 10^{3}$	$1.05\ 10^{5}$	$6.32\ 10^{1}$	$7.34\ 10^{3}$
	$1.10\ 10^{4}$	$5.43\ 10^{3}$	$1.05\ 10^{5}$	$9.28\ 10^{1}$	$7.39\ 10^{3}$
	$1.83\ 10^{4}$	$5.46\ 10^{3}$	$1.05\ 10^{5}$	$1.46\ 10^{2}$	$7.48\ 10^{3}$
	$3.65\ 10^{4}$	$5.52\ 10^{3}$	$1.05\ 10^{5}$	$2.51\ 10^{2}$	$7.67\ 10^{3}$
	$1.83\ 10^{5}$	$5.66\ 10^{3}$	$1.05\ 10^{5}$	$4.87\ 10^{2}$	$8.08\ 10^{3}$
	$3.65\ 10^{7}$	$5.66\ 10^{3}$	$1.05\ 10^{5}$	$5.02\ 10^{2}$	$8.10\ 10^{3}$
^{241}Am	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$5.37\ 10^{3}$	$1.05\ 10^{5}$	$1.72\ 10^{0}$	$7.23\ 10^{3}$
	$7.30\ 10^{2}$	$5.38\ 10^{3}$	$1.05\ 10^{5}$	$7.28\ 10^{0}$	$7.24\ 10^{3}$
	$1.83\ 10^{3}$	$5.39\ 10^{3}$	$1.05\ 10^{5}$	$2.37\ 10^{1}$	$7.26\ 10^{3}$
	$3.65\ 10^{3}$	$5.41\ 10^{3}$	$1.05\ 10^{5}$	$5.01\ 10^{1}$	$7.28\ 10^{3}$
	$7.30\ 10^{3}$	$5.45\ 10^{3}$	$1.05\ 10^{5}$	$9.96\ 10^{1}$	$7.34\ 10^{3}$
	$1.10\ 10^{4}$	$5.48\ 10^{3}$	$1.05\ 10^{5}$	$1.45\ 10^{2}$	$7.39\ 10^{3}$
	$1.83\ 10^{4}$	$5.54\ 10^{3}$	$1.05\ 10^{5}$	$2.25\ 10^{2}$	$7.47\ 10^{3}$
	$3.65\ 10^{4}$	$5.65\ 10^{3}$	$1.05\ 10^{5}$	$3.75\ 10^{2}$	$7.64\ 10^{3}$
	$1.83\ 10^{5}$	$5.85\ 10^{3}$	$1.05\ 10^{5}$	$6.47\ 10^{2}$	$7.93\ 10^{3}$
	$3.65\ 10^{7}$	$5.86\ 10^{3}$	$1.05\ 10^{5}$	$6.56\ 10^{2}$	$7.94\ 10^{3}$

Notes

1 The time integrals per unit mass of plant have been evaluated for a continuous deposition of each radionuclide for 1 year at a rate of 1 Bq m^{-2} s^{-1}.

2 The yields of each crop are given in Table 3.18.

TABLE 3.23 The time integrals of activity in various food products derived from cows grazing contaminated pasture

	Time integral of activity in animal products (Bq y kg^{-1})^{1}			
Nuclide	Times (days)	Meat	Liver	Milk
^{90}Sr	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	9.22 10^{3}	9.22 10^{3}	4.33 10^{4}
	7.30 10^{2}	1.36 10^{4}	1.36 10^{4}	6.35 10^{4}
	1.83 10^{3}	2.06 10^{4}	2.06 10^{4}	9.64 10^{4}
	3.65 10^{3}	2.45 10^{4}	2.45 10^{4}	1.14 10^{5}
	7.30 10^{3}	2.72 10^{4}	2.72 10^{4}	1.27 10^{5}
	1.10 10^{4}	2.86 10^{4}	2.86 10^{4}	1.34 10^{5}
	1.83 10^{4}	2.98 10^{4}	2.98 10^{4}	1.39 10^{5}
	3.65 10^{4}	3.04 10^{4}	3.04 10^{4}	1.42 10^{5}
	1.83 10^{5}	3.04 10^{4}	3.04 10^{4}	1.42 10^{5}
	3.65 10^{7}	3.04 10^{4}	3.04 10^{4}	1.42 10^{5}
^{106}Ru	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	1.26 10^{4}	1.26 10^{4}	2.87 10^{1}
	7.30 10^{2}	1.52 10^{4}	1.52 10^{4}	2.94 10^{1}
	1.83 10^{3}	1.60 10^{4}	1.60 10^{4}	2.98 10^{1}
	3.65 10^{3}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	7.30 10^{3}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	1.10 10^{4}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	1.83 10^{4}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	3.65 10^{4}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	1.83 10^{5}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
	3.65 10^{7}	1.60 10^{4}	1.60 10^{4}	2.99 10^{1}
^{129}I	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	1.04 10^{5}	1.04 10^{5}	1.55 10^{5}
	7.30 10^{2}	1.14 10^{5}	1.14 10^{5}	1.69 10^{5}
	1.83 10^{3}	1.34 10^{5}	1.34 10^{5}	1.99 10^{5}
	3.65 10^{3}	1.54 10^{5}	1.54 10^{5}	2.27 10^{5}
	7.30 10^{3}	1.73 10^{5}	1.73 10^{5}	2.56 10^{5}
	1.10 10^{4}	1.85 10^{5}	1.85 10^{5}	2.74 10^{5}
	1.83 10^{4}	1.98 10^{5}	1.98 10^{5}	2.93 10^{5}
	3.65 10^{4}	2.09 10^{5}	2.09 10^{5}	3.10 10^{5}
	1.83 10^{5}	2.13 10^{5}	2.13 10^{5}	3.15 10^{5}
	3.65 10^{7}	2.13 10^{5}	2.13 10^{5}	3.15 10^{5}
^{131}I	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}	0.00 10^{0}
	3.65 10^{2}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	7.30 10^{2}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	1.83 10^{3}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	3.65 10^{3}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	7.30 10^{3}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	1.10 10^{4}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}
	1.83 10^{4}	2.47 10^{4}	2.47 10^{4}	5.82 10^{4}

	Time integral of activity in animal products (Bq y kg^{-1})[1]			
Nuclide	Times (days)	Meat	Liver	Milk
	$3.65\ 10^{4}$	$2.47\ 10^{4}$	$2.47\ 10^{4}$	$5.82\ 10^{4}$
	$1.83\ 10^{5}$	$2.47\ 10^{4}$	$2.47\ 10^{4}$	$5.82\ 10^{4}$
	$3.65\ 10^{7}$	$2.47\ 10^{4}$	$2.47\ 10^{4}$	$5.82\ 10^{4}$
^{137}Cs	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$7.83\ 10^{5}$	$7.83\ 10^{5}$	$1.54\ 10^{5}$
	$7.30\ 10^{2}$	$8.31\ 10^{5}$	$8.31\ 10^{5}$	$1.63\ 10^{5}$
	$1.83\ 10^{3}$	$8.84\ 10^{5}$	$8.84\ 10^{5}$	$1.73\ 10^{5}$
	$3.65\ 10^{3}$	$9.07\ 10^{5}$	$9.07\ 10^{5}$	$1.77\ 10^{5}$
	$7.30\ 10^{3}$	$9.14\ 10^{5}$	$9.14\ 10^{5}$	$1.78\ 10^{5}$
	$1.10\ 10^{4}$	$9.15\ 10^{5}$	$9.14\ 10^{5}$	$1.79\ 10^{5}$
	$1.83\ 10^{4}$	$9.15\ 10^{5}$	$9.15\ 10^{5}$	$1.79\ 10^{5}$
	$3.65\ 10^{4}$	$9.15\ 10^{5}$	$9.15\ 10^{5}$	$1.79\ 10^{5}$
	$1.83\ 10^{5}$	$9.15\ 10^{5}$	$9.15\ 10^{5}$	$1.79\ 10^{5}$
	$3.65\ 10^{7}$	$9.15\ 10^{5}$	$9.15\ 10^{5}$	$1.79\ 10^{5}$
^{239}Pu	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$8.86\ 10^{2}$	$1.07\ 10^{5}$	$1.61\ 10^{1}$
	$7.30\ 10^{2}$	$1.57\ 10^{3}$	$1.91\ 10^{5}$	$2.84\ 10^{1}$
	$1.83\ 10^{3}$	$2.95\ 10^{3}$	$3.61\ 10^{5}$	$5.33\ 10^{1}$
	$3.65\ 10^{3}$	$3.99\ 10^{3}$	$4.88\ 10^{5}$	$7.21\ 10^{1}$
	$7.30\ 10^{3}$	$4.48\ 10^{3}$	$5.48\ 10^{5}$	$8.09\ 10^{1}$
	$1.10\ 10^{4}$	$4.53\ 10^{3}$	$5.55\ 10^{5}$	$8.19\ 10^{1}$
	$1.83\ 10^{4}$	$4.54\ 10^{3}$	$5.57\ 10^{5}$	$8.21\ 10^{1}$
	$3.65\ 10^{4}$	$4.55\ 10^{3}$	$5.57\ 10^{5}$	$8.21\ 10^{1}$
	$1.83\ 10^{5}$	$4.55\ 10^{3}$	$5.57\ 10^{5}$	$8.21\ 10^{1}$
	$3.65\ 10^{7}$	$4.55\ 10^{3}$	$5.57\ 10^{5}$	$8.21\ 10^{1}$
^{241}Am	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$8.87\ 10^{2}$	$1.08\ 10^{5}$	$1.61\ 10^{1}$
	$7.30\ 10^{2}$	$1.57\ 10^{3}$	$1.92\ 10^{5}$	$2.85\ 10^{1}$
	$1.83\ 10^{3}$	$2.97\ 10^{3}$	$3.63\ 10^{5}$	$5.36\ 10^{1}$
	$3.65\ 10^{3}$	$4.03\ 10^{3}$	$4.93\ 10^{5}$	$7.28\ 10^{1}$
	$7.30\ 10^{3}$	$4.55\ 10^{3}$	$5.58\ 10^{5}$	$8.22\ 10^{1}$
	$1.10\ 10^{4}$	$4.63\ 10^{3}$	$5.68\ 10^{5}$	$8.37\ 10^{1}$
	$1.83\ 10^{4}$	$4.67\ 10^{3}$	$5.72\ 10^{5}$	$8.43\ 10^{1}$
	$3.65\ 10^{4}$	$4.69\ 10^{3}$	$5.75\ 10^{5}$	$8.47\ 10^{1}$
	$1.83\ 10^{5}$	$4.70\ 10^{3}$	$5.75\ 10^{5}$	$8.48\ 10^{1}$
	$3.65\ 10^{7}$	$4.70\ 10^{3}$	$5.75\ 10^{5}$	$8.48\ 10^{1}$

Note

1 The time integrals correspond to the activity in unit mass of the respective foodstuffs following the continuous deposition of activity on land at a rate of 1 Bq m^{-2} s^{1} for one year.

TABLE 3.24 The time integrals of activity in various food products derived from sheep grazing contaminated pasture

	Time integral of activity in animal products (Bq y kg^{-1})[1]		
Nuclide	Times (days)	Meat	Liver
^{90}Sr	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$1.14\ 10^{4}$	$1.14\ 10^{4}$
	$7.30\ 10^{2}$	$1.42\ 10^{4}$	$1.42\ 10^{4}$
	$1.83\ 10^{3}$	$1.80\ 10^{4}$	$1.80\ 10^{4}$
	$3.65\ 10^{3}$	$1.99\ 10^{4}$	$1.99\ 10^{4}$
	$7.30\ 10^{3}$	$2.12\ 10^{4}$	$2.12\ 10^{4}$
	$1.10\ 10^{4}$	$2.19\ 10^{4}$	$2.19\ 10^{4}$
	$1.83\ 10^{4}$	$2.24\ 10^{4}$	$2.24\ 10^{4}$
	$3.65\ 10^{4}$	$2.27\ 10^{4}$	$2.27\ 10^{4}$
	$1.83\ 10^{5}$	$2.27\ 10^{4}$	$2.27\ 10^{4}$
	$3.65\ 10^{7}$	$2.27\ 10^{4}$	$2.27\ 10^{4}$
^{106}Ru	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$2.00\ 10^{4}$	$2.00\ 10^{4}$
	$7.30\ 10^{2}$	$2.25\ 10^{4}$	$2.25\ 10^{4}$
	$1.83\ 10^{3}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$3.65\ 10^{3}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$7.30\ 10^{3}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$1.10\ 10^{4}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$1.83\ 10^{4}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$3.65\ 10^{4}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$1.83\ 10^{5}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
	$3.65\ 10^{7}$	$2.34\ 10^{4}$	$2.34\ 10^{4}$
^{129}I	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$2.98\ 10^{5}$	$2.98\ 10^{5}$
	$7.30\ 10^{2}$	$3.30\ 10^{5}$	$3.30\ 10^{5}$
	$1.83\ 10^{3}$	$3.95\ 10^{5}$	$3.95\ 10^{5}$
	$3.65\ 10^{3}$	$4.46\ 10^{5}$	$4.46\ 10^{5}$
	$7.30\ 10^{3}$	$4.85\ 10^{5}$	$4.85\ 10^{5}$
	$1.10\ 10^{4}$	$5.03\ 10^{5}$	$5.03\ 10^{5}$
	$1.83\ 10^{4}$	$5.23\ 10^{5}$	$5.23\ 10^{5}$
	$3.65\ 10^{4}$	$5.39\ 10^{5}$	$5.39\ 10^{5}$
	$1.83\ 10^{5}$	$5.44\ 10^{5}$	$5.44\ 10^{5}$
	$3.65\ 10^{7}$	$5.44\ 10^{5}$	$5.44\ 10^{5}$
^{131}I	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$7.30\ 10^{2}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$1.83\ 10^{3}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$3.65\ 10^{3}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$7.30\ 10^{3}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$1.10\ 10^{4}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$1.83\ 10^{4}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$3.65\ 10^{4}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$

	Time integral of activity in animal products (Bq y kg^{-1})[1]		
Nuclide	Times (days)	Meat	Liver
	$1.83\ 10^{5}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
	$3.65\ 10^{7}$	$3.17\ 10^{4}$	$3.17\ 10^{4}$
^{137}Cs	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$1.45\ 10^{6}$	$1.45\ 10^{6}$
	$7.30\ 10^{2}$	$1.57\ 10^{6}$	$1.57\ 10^{6}$
	$1.83\ 10^{3}$	$1.77\ 10^{6}$	$1.77\ 10^{6}$
	$3.65\ 10^{3}$	$1.87\ 10^{6}$	$1.87\ 10^{6}$
	$7.30\ 10^{3}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
	$1.10\ 10^{4}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
	$1.83\ 10^{4}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
	$3.65\ 10^{4}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
	$1.83\ 10^{5}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
	$3.65\ 10^{7}$	$1.91\ 10^{6}$	$1.91\ 10^{6}$
^{239}Pu	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$1.75\ 10^{3}$	$1.24\ 10^{5}$
	$7.30\ 10^{2}$	$2.49\ 10^{3}$	$1.77\ 10^{5}$
	$1.83\ 10^{3}$	$3.21\ 10^{3}$	$2.29\ 10^{5}$
	$3.65\ 10^{3}$	$3.54\ 10^{3}$	$2.53\ 10^{5}$
	$7.30\ 10^{3}$	$3.66\ 10^{3}$	$2.61\ 10^{5}$
	$1.10\ 10^{4}$	$3.67\ 10^{3}$	$2.62\ 10^{5}$
	$1.83\ 10^{4}$	$3.67\ 10^{3}$	$2.62\ 10^{5}$
	$3.65\ 10^{4}$	$3.67\ 10^{3}$	$2.62\ 10^{5}$
	$1.83\ 10^{5}$	$3.67\ 10^{3}$	$2.62\ 10^{5}$
	$3.65\ 10^{7}$	$3.67\ 10^{3}$	$2.62\ 10^{5}$
^{241}Am	$0.00\ 10^{0}$	$0.00\ 10^{0}$	$0.00\ 10^{0}$
	$3.65\ 10^{2}$	$1.76\ 10^{3}$	$1.25\ 10^{5}$
	$7.30\ 10^{2}$	$2.49\ 10^{3}$	$1.78\ 10^{5}$
	$1.83\ 10^{3}$	$3.22\ 10^{3}$	$2.30\ 10^{5}$
	$3.65\ 10^{3}$	$3.56\ 10^{3}$	$2.54\ 10^{5}$
	$7.30\ 10^{3}$	$3.69\ 10^{3}$	$2.64\ 10^{5}$
	$1.10\ 10^{4}$	$3.71\ 10^{3}$	$2.65\ 10^{5}$
	$1.83\ 10^{4}$	$3.72\ 10^{3}$	$2.66\ 10^{5}$
	$3.65\ 10^{4}$	$3.72\ 10^{3}$	$2.66\ 10^{5}$
	$1.83\ 10^{5}$	$3.73\ 10^{3}$	$2.66\ 10^{5}$
	$3.65\ 10^{7}$	$3.73\ 10^{3}$	$2.66\ 10^{5}$

Note

1 The time integrals correspond to the activity in unit mass of each food following the continuous deposition of activity on land at a rate of 1 Bq m^{-2} s^{-1} for one year

TABLE 3.25 Specific activity of tritium and carbon-14 in foods

Foodstuff	Water content %	Carbon content %	Concentration in food Bq kg^{-1} per Bq m^{-3} (air concentration)^2	
			^{3}H	^{14}C
Grain	10	36	12.5	2400
Green vegetables	80	8	100	533
Root vegetables	80	8	100	533
Fruit	84.4	8	100	533
Cow's milk	90	4	112.5	267
Cow/sheep/pig/chicken - meat/offal/eggs	70	12	87.5	800.0

Notes

1 Carbon content is taken as being 40% of dry matter.

2 The atmospheric specific activities are:

125.0 Bq (^{3}H) kg^{-1} (H_2O) per Bq m^{-3} (^{3}H)
6667.0 Bq (^{14}C) kg^{-1} (^{12}C) per Bq m^{-3} (^{14}C)

Table 3.26 Dose conversion factors for rotational geometry (ICRP, 1996)

Photon energy (MeV)	Absorbed dose in air per unit photon fluence (Gy cm^2) (ICRP 74, p159)	Dose in tissue per unit photon fluence (Sv cm^2)				
		Effective	Gonads	Breast	Thyroid	Skin
0.01	$7.43\ 10^{-12}$	$2.42\ 10^{-14}$	$2.76\ 10^{-14}$	$6.46\ 10^{-14}$	$2.15\ 10^{-15}$	$1.49\ 10^{-12}$
0.015	$3.12\ 10^{-12}$	$4.77\ 10^{-14}$	$8.89\ 10^{-14}$	$2.33\ 10^{-13}$	$7.08\ 10^{-14}$	$1.03\ 10^{-12}$
0.02	$1.68\ 10^{-12}$	$7.76\ 10^{-14}$	$1.28\ 10^{-13}$	$3.33\ 10^{-13}$	$2.03\ 10^{-13}$	$7.27\ 10^{-13}$
0.03	$7.21\ 10^{-13}$	$1.38\ 10^{-13}$	$1.61\ 10^{-13}$	$3.24\ 10^{-13}$	2.9510^{-13}	$4.19\ 10^{-13}$
0.05	$3.23\ 10^{-13}$	$2.14\ 10^{-13}$	$2.09\ 10^{-13}$	$2.62\ 10^{-13}$	$3.13\ 10^{-13}$	$2.68\ 10^{-13}$
0.1	$3.71\ 10^{-13}$	$3.56\ 10^{-13}$	$3.44\ 10^{-13}$	$3.55\ 10^{-13}$	$4.56\ 10^{-13}$	$3.62\ 10^{-13}$
0.2	$8.56\ 10^{-13}$	$7.31\ 10^{-13}$	$7.13\ 10^{-13}$	$7.49\ 10^{-13}$	$9.49\ 10^{-13}$	$7.93\ 10^{-13}$
0.5	$2.38\ 10^{-12}$	$1.93\ 10^{-12}$	$1.87\ 10^{-12}$	$2.03\ 10^{-12}$	$2.43\ 10^{-12}$	$2.14\ 10^{-12}$
1	$4.47\ 10^{-12}$	$3.71\ 10^{-12}$	$3.55\ 10^{-12}$	$3.90\ 10^{-12}$	$4.61\ 10^{-12}$	$4.09\ 10^{-12}$
1.5	$6.14\ 10^{-12}$	$5.23\ 10^{-12}$	$5.03\ 10^{-12}$	$5.45\ 10^{-12}$	$6.40\ 10^{-12}$	$5.69\ 10^{-12}$
2	$7.54\ 10^{-12}$	$6.57\ 10^{-12}$	$6.36\ 10^{-12}$	$6.80\ 10^{-12}$	$7.95\ 10^{-12}$	$7.08\ 10^{-12}$
4	$1.21\ 10^{-11}$	$1.10\ 10^{-11}$	$1.09\ 10^{-11}$	$1.12\ 10^{-11}$	$1.29\ 10^{-11}$	$1.15\ 10^{-11}$

Table 3.27 Time integral of the effective dose rate and skin dose rate (Sv) from the external irradiation due to photons for a deposition at a rate of 1 Bq m^{-2} s^{-1}

	Annual dose (effective) 1 m above the ground (Sv)					Annual dose (skin) 1 m above the ground (Sv)				
Radionuclide	Time (y)					Time (y)				
	1	50	500	10000	1 10^{8}	1	50	500	10000	1 10^{8}
^{60}Co	7.18 10^{-1}	7.26 10^{0}	7.26 10^{0}	7.26 10^{0}	7.26 10^{0}	7.85 10^{-1}	7.94 10^{0}	7.94 10^{0}	7.94 10^{0}	7.94 10^{0}
^{95}Nb	5.80 10^{-2}	6.71 10^{-2}	6.71 10^{-2}	6.71 10^{-2}	6.71 10^{-2}	6.40 10^{-2}	7.41 10^{-2}	7.41 10^{-2}	7.41 10^{-2}	7.41 10^{-2}
^{95}Zr	1.63 10^{-1}	2.35 10^{-1}	2.35 10^{-1}	2.35 10^{-1}	2.35 10^{-1}	1.80 10^{-1}	2.60 10^{-1}	2.60 10^{-1}	2.60 10^{-1}	2.60 10^{-1}
^{106}Ru	5.16 10^{-2}	1.66 10^{-1}	1.66 10^{-1}	1.66 10^{-1}	1.66 10^{-1}	5.71 10^{-2}	1.84 10^{-1}	1.84 10^{-1}	1.84 10^{-1}	1.84 10^{-1}
^{129}I	2.17 10^{-3}	2.55 10^{-2}	2.61 10^{-2}	2.61 10^{-2}	2.64 10^{-2}	5.62 10^{-3}	6.22 10^{-2}	6.31 10^{-2}	6.31 10^{-2}	6.36 10^{-2}
^{131}I	7.57 10^{-3}	7.82 10^{-3}	7.82 10^{-3}	7.82 10^{-3}	7.82 10^{-3}	8.36 10^{-3}	8.64 10^{-3}	8.64 10^{-3}	8.64 10^{-3}	8.64 10^{-3}
^{137}Cs	1.83 10^{-1}	4.16 10^{0}	4.45 10^{0}	4.45 10^{0}	4.45 10^{0}	2.03 10^{-1}	4.60 10^{0}	4.92 10^{0}	4.92 10^{0}	4.92 10^{0}
^{239}Pu	2.28 10^{-5}	4.45 10^{-4}	5.33 10^{-4}	6.79 10^{-4}	8.95 10^{-1}	1.24 10^{-4}	1.37 10^{-3}	1.47 10^{-3}	1.63 10^{-3}	9.89 10^{-1}
^{241}Pu	2.87 10^{-6}	1.02 10^{-3}	1.23 10^{-3}	1.24 10^{-3}	6.14 10^{-3}	4.24 10^{-6}	1.22 10^{-3}	1.45 10^{-3}	1.46 10^{-3}	6.88 10^{-3}
^{241}Am	4.76 10^{-3}	8.58 10^{-2}	9.23 10^{-2}	9.25 10^{-2}	2.40 10^{-1}	7.15 10^{-3}	1.12 10^{-1}	1.19 10^{-1}	1.19 10^{-1}	2.82 10^{-1}

Includes contributions from in-growth of daughter products on the ground

Table 3.28 Dose rate factors for exposure to skin 0.8 metre above a contaminated surface from irradiation by electrons[1]

Radionuclide	Dose rate to skin ($Sv\ y^{-1}$ per $Bq\ m^{-2}$)	Radionuclide	Dose rate to skin ($Sv\ y^{-1}$ per $Bq\ m^{-2}$)
^{3}H	-	^{122}Sb	$3.01\ 10^{-07}$
^{14}C	$9.16\ 10^{-10}$	^{124}Sb	$1.81\ 10^{-07}$
^{35}S	$1.41\ 10^{-09}$	^{125}Sb	$2.80\ 10^{-08}$
^{41}Ar	$2.66\ 10^{-07}$	^{129}I	$3.47\ 10^{-10}$
^{51}Cr	$2.92\ 10^{-11}$	^{131}I	$8.58\ 10^{-08}$
^{54}Mn	$1.15\ 10^{-10}$	^{132}I	$2.62\ 10^{-07}$
^{59}Fe	$3.79\ 10^{-08}$	^{133}I	$2.32\ 10^{-07}$
^{58}Co	$1.35\ 10^{-08}$	^{135}I	$2.00\ 10^{-07}$
^{60}Co	$2.34\ 10^{-08}$	^{131m}Xe	$4.09\ 10^{-08}$
^{65}Zn	$7.16\ 10^{-10}$	^{133}Xe	$2.62\ 10^{-08}$
^{85}Kr	1.3110^{-07}	^{133m}Xe	$7.86\ 10^{-08}$
^{85m}Kr	$1.38\ 10^{-07}$	^{135}Xe	$1.73\ 10^{-07}$
^{87}Kr	$4.61\ 10^{-07}$	^{135m}Xe	$6.10\ 10^{-08}$
^{88}Kr	$1.49\ 10^{-07}$	^{137}Xe	-
^{89}Kr	-	^{138}Xe	$2.95\ 10^{-07}$
^{89}Sr	$3.16\ 10^{-07}$	^{134}Cs	$7.66\ 10^{-08}$
^{90}Sr	$5.07\ 10^{-07}$	^{137}Cs	$1.20\ 10^{-07}$
^{95}Zr	$3.61\ 10^{-08}$	^{140}Ba	$1.49\ 10^{-07}$
^{95}Nb	$1.43\ 10^{-09}$	^{140}La	$2.95\ 10^{-07}$
^{106}Ru	$4.95\ 10^{-07}$		

Note:

1 Holford (1989).

3.7 Figures

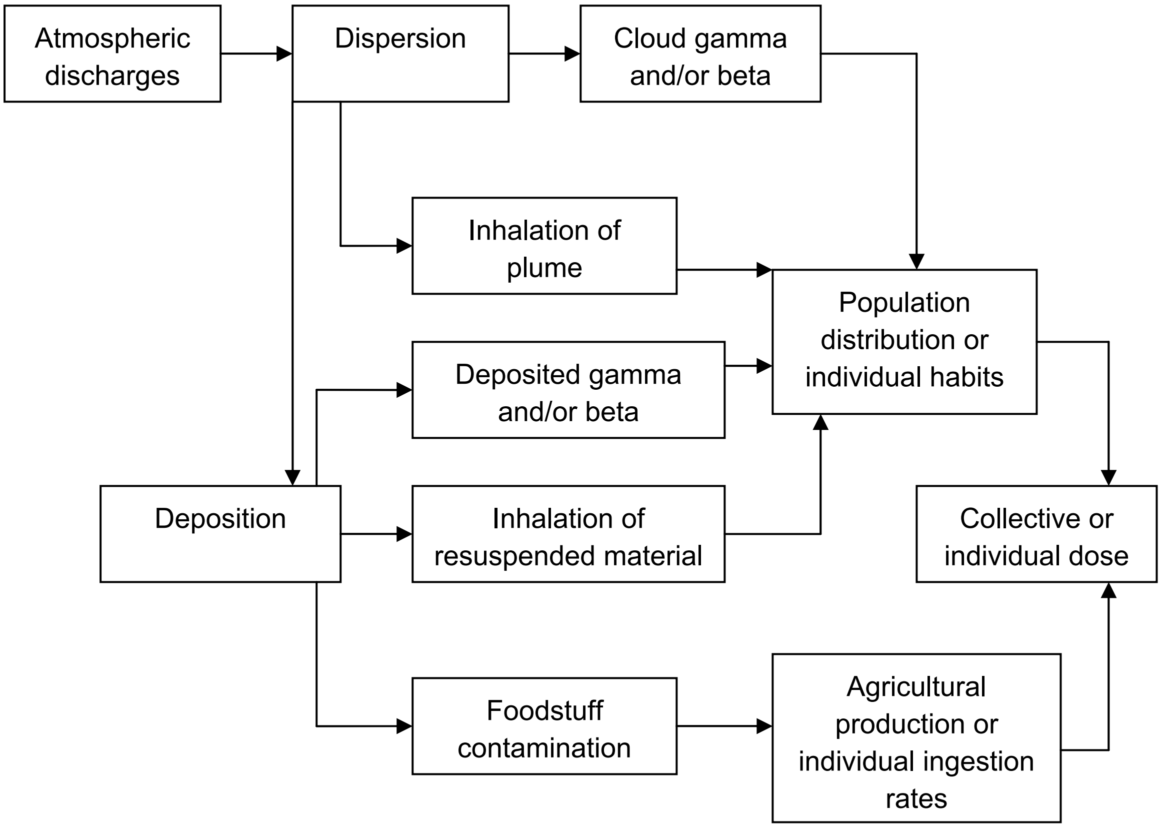


Figure 3.1 Schematic diagram of exposure pathways considered for atmospheric dispersion

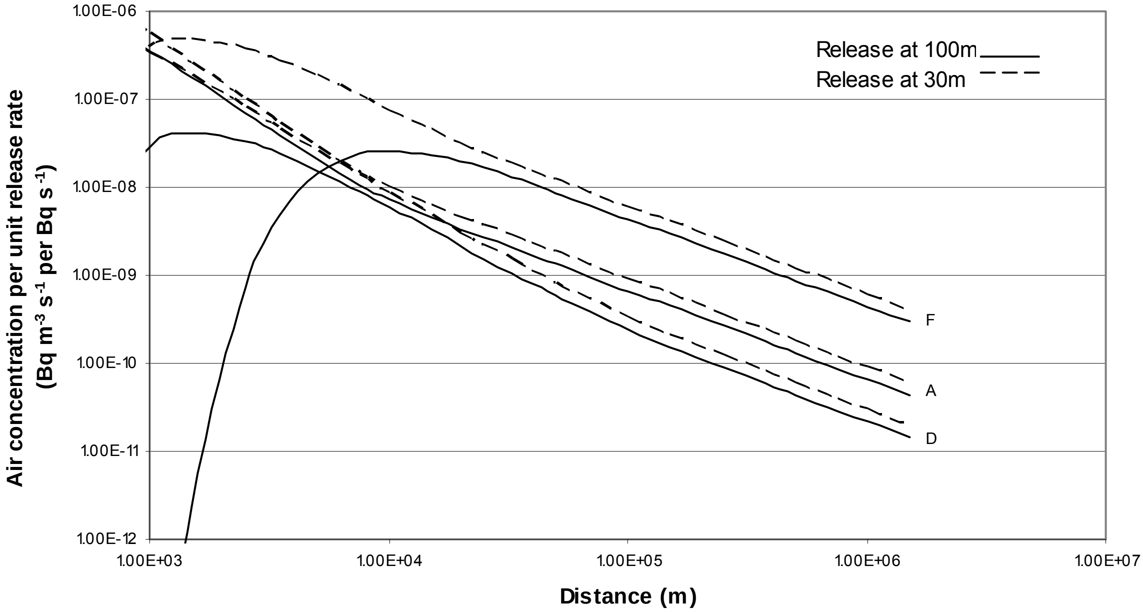


Figure 3.2 The variation of activity concentration in air with distance for a non-depositing radionuclide showing the influence of effective release height and stability category assuming a uniform windrose

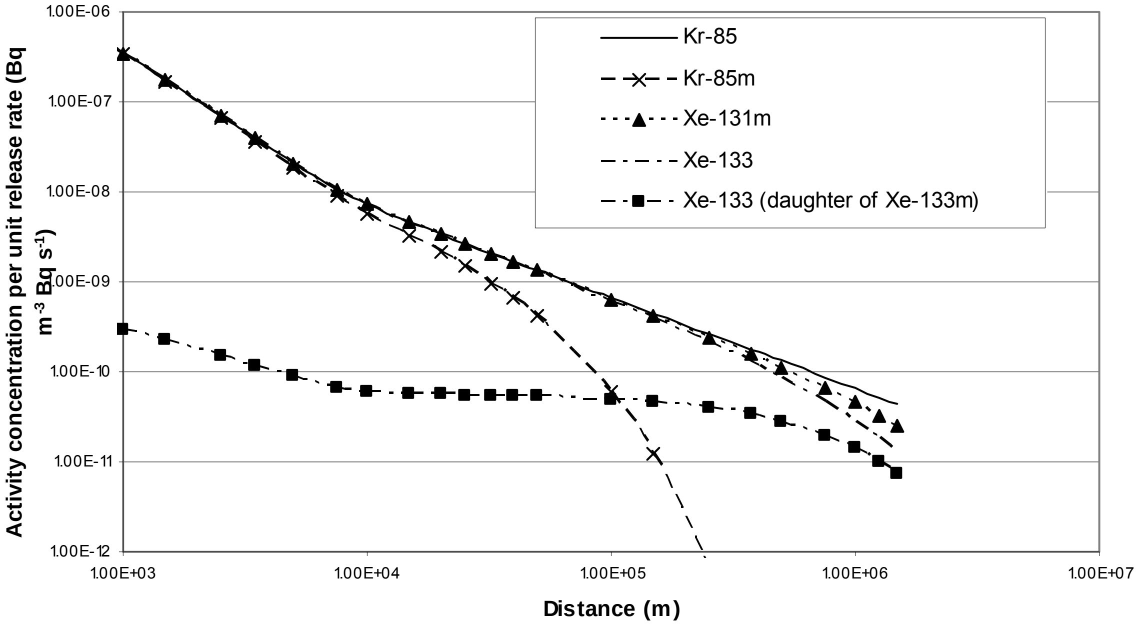


Figure 3.3 The variation of activity concentration in air with distance for noble gases of various half-lives assuming a uniform windrose

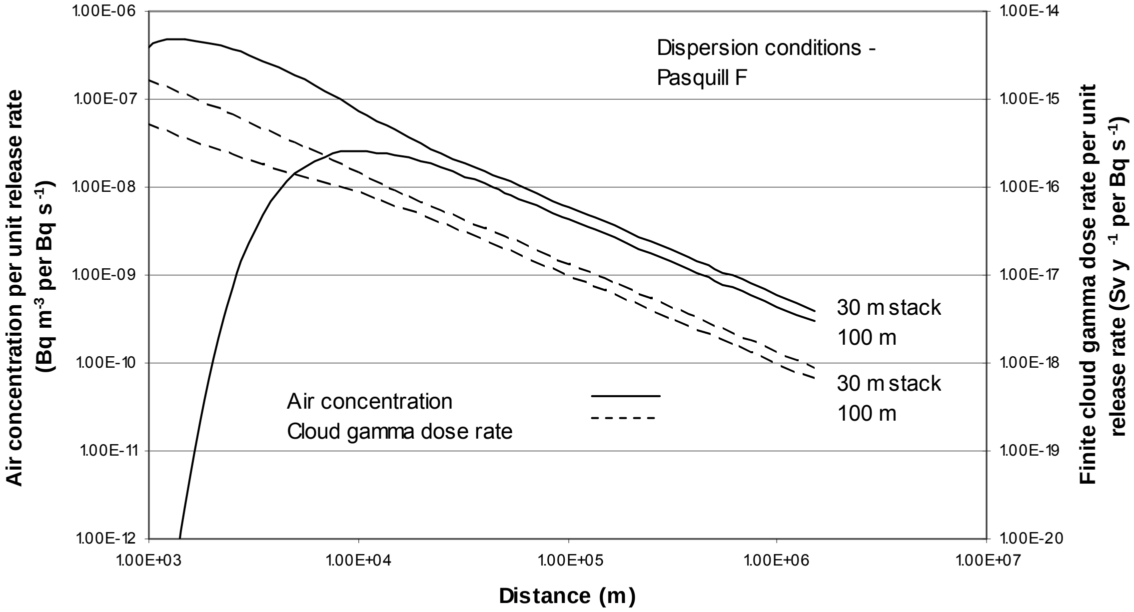


Figure 3.4 The variation of activity concentration in air and external gamma dose rate from the plume with release height assuming a uniform windrose

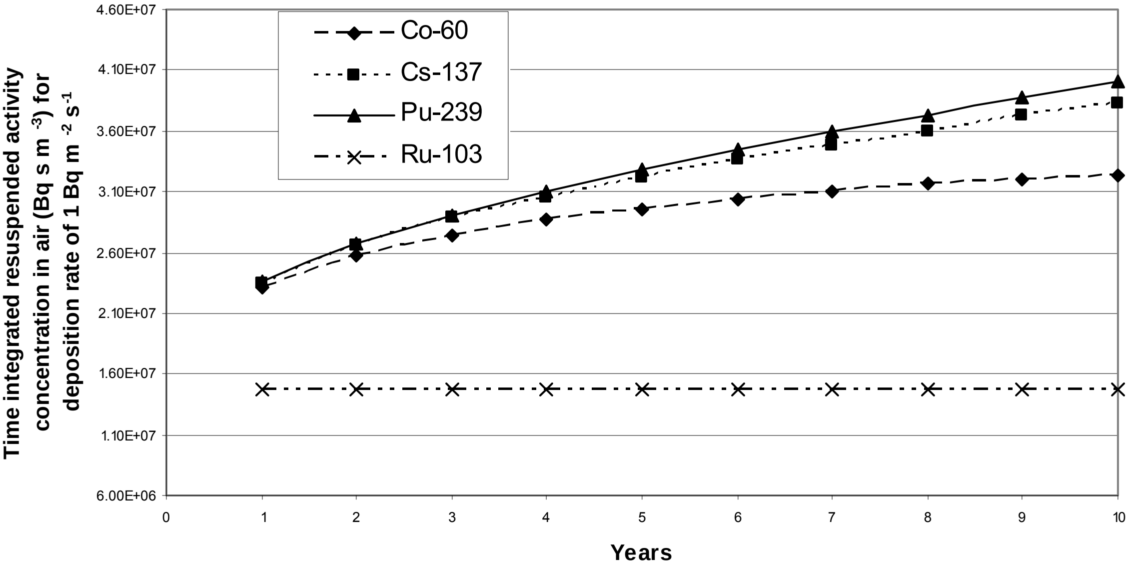


Figure 3.5a Predicted integrated activity concentrations in air following unit deposition rate of 1 Bq m^{-2} s^{-1} for 1 year for the period 1 to 10 years

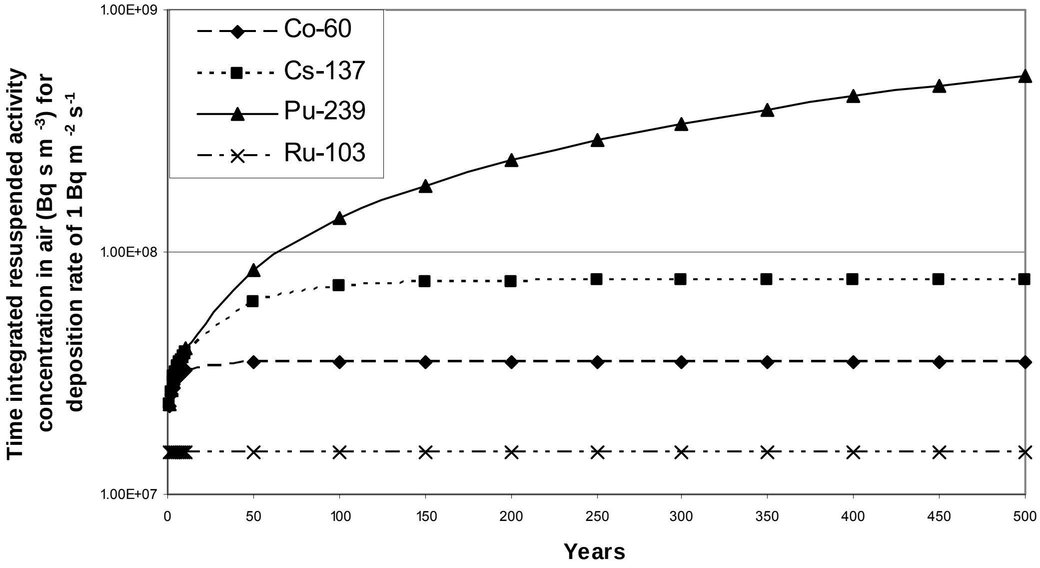
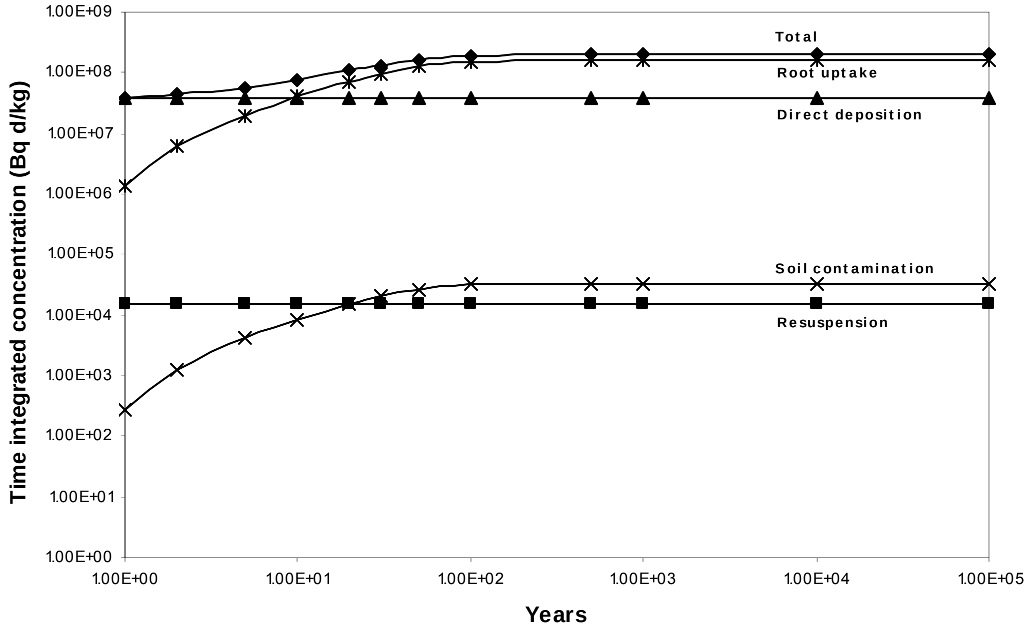


Figure 3.5b Predicted integrated activity concentrations in air following unit deposition rate of 1 Bq m^{-2} s^{-1} for 1 year for the period 1 to 500 years

Green vegetables



Grain

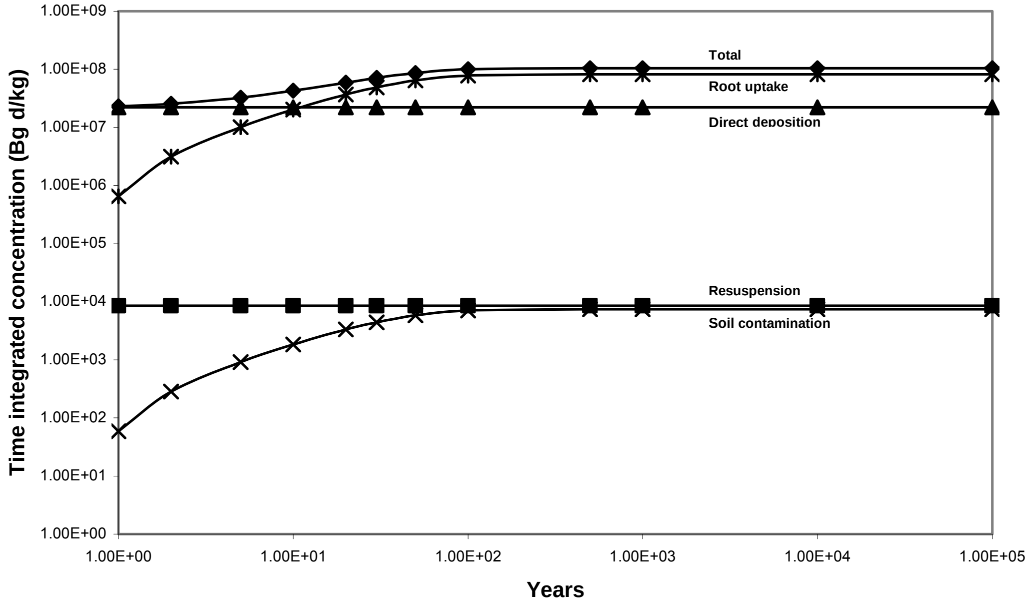


Figure 3.6 The relative importance and time dependence of the important mechanisms for the transfer of strontium-90 to green vegetables and grain

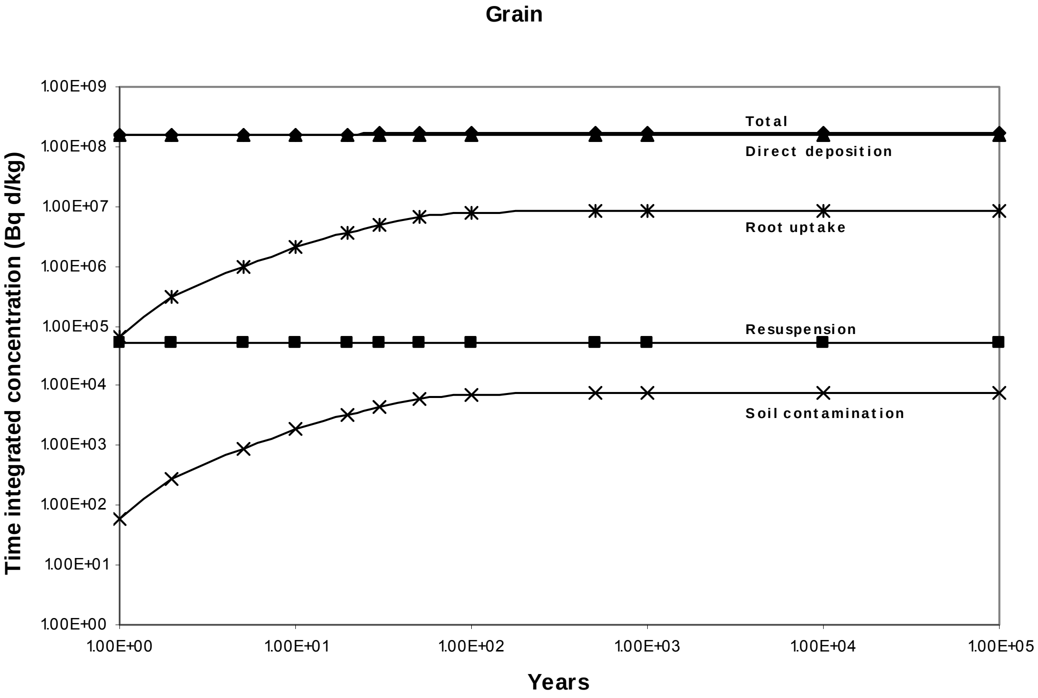
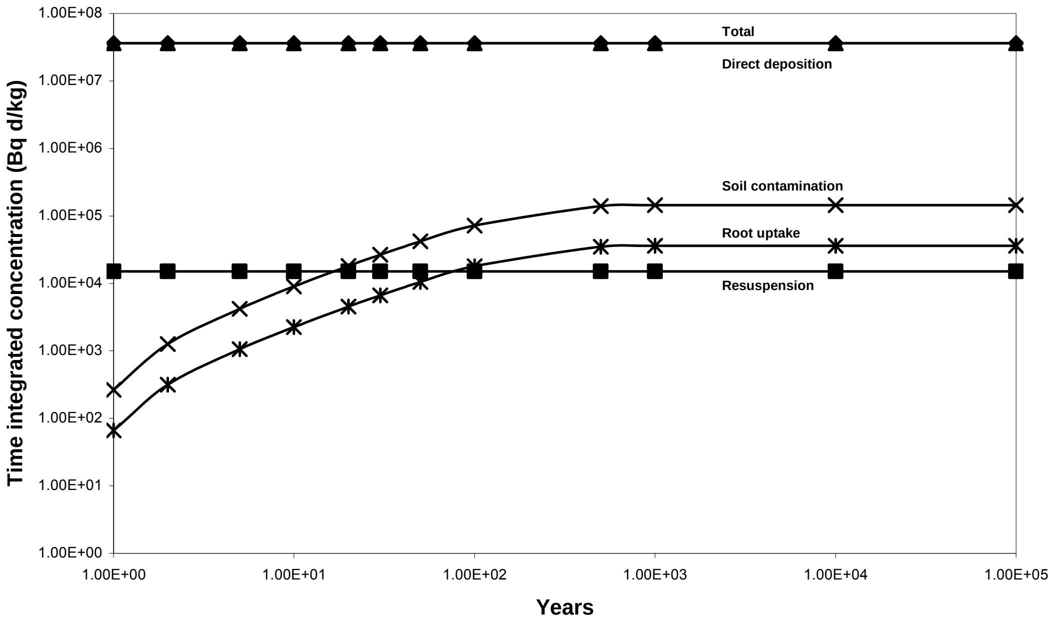


Figure 3.7 The relative importance and time dependence of the important mechanisms for the transfer of caesium-137 to green vegetables and grain

Green vegetables



Grain

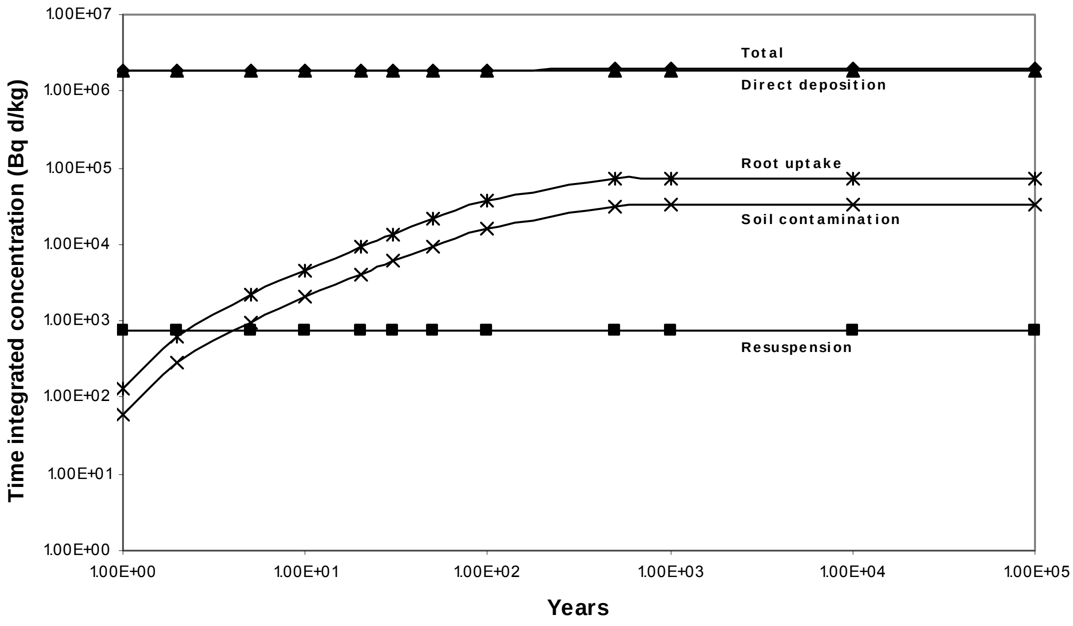


Figure 3.8 The relative importance and time dependence of the important mechanisms for the transfer of plutonium-239 to green vegetables and grain

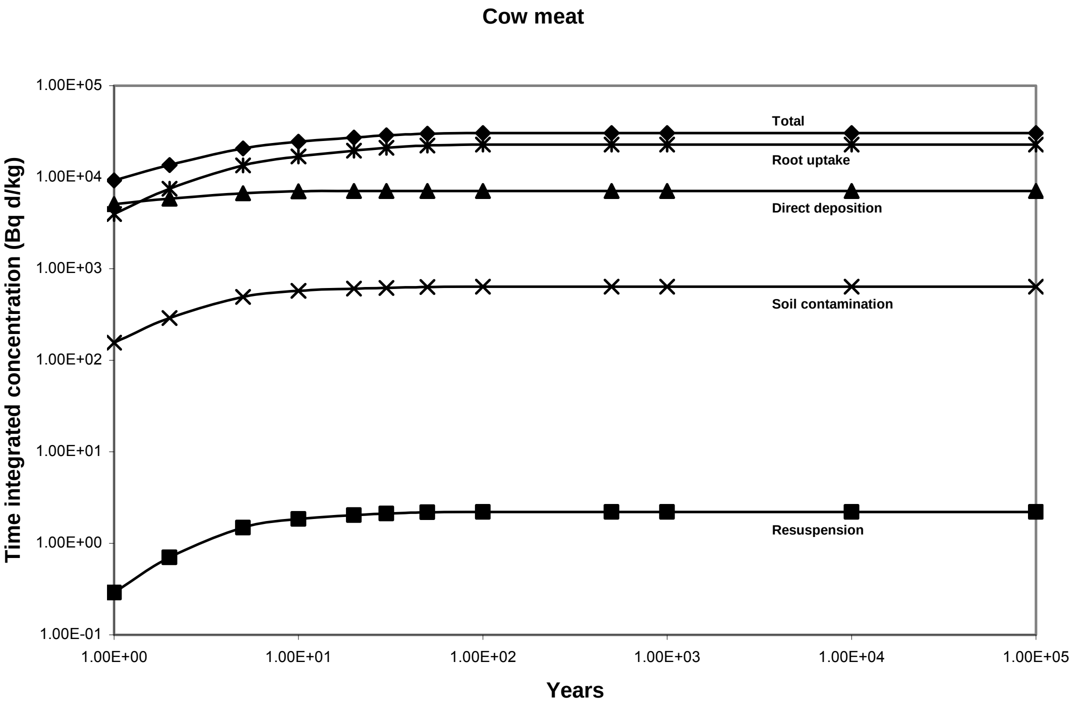


Figure 3.9 The relative importance and time dependence of the important mechanisms for the transfer of strontium-90 to cow muscle

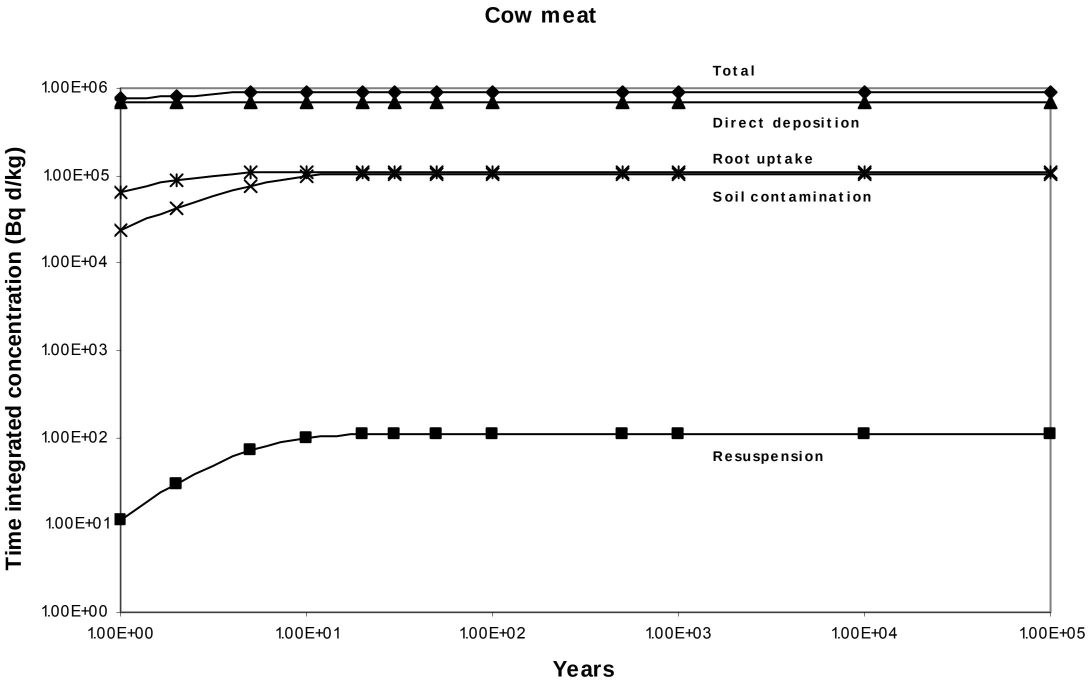


Figure 3.10 The relative importance and time dependence of the important mechanisms for the transfer of caesium-137 to cow muscle

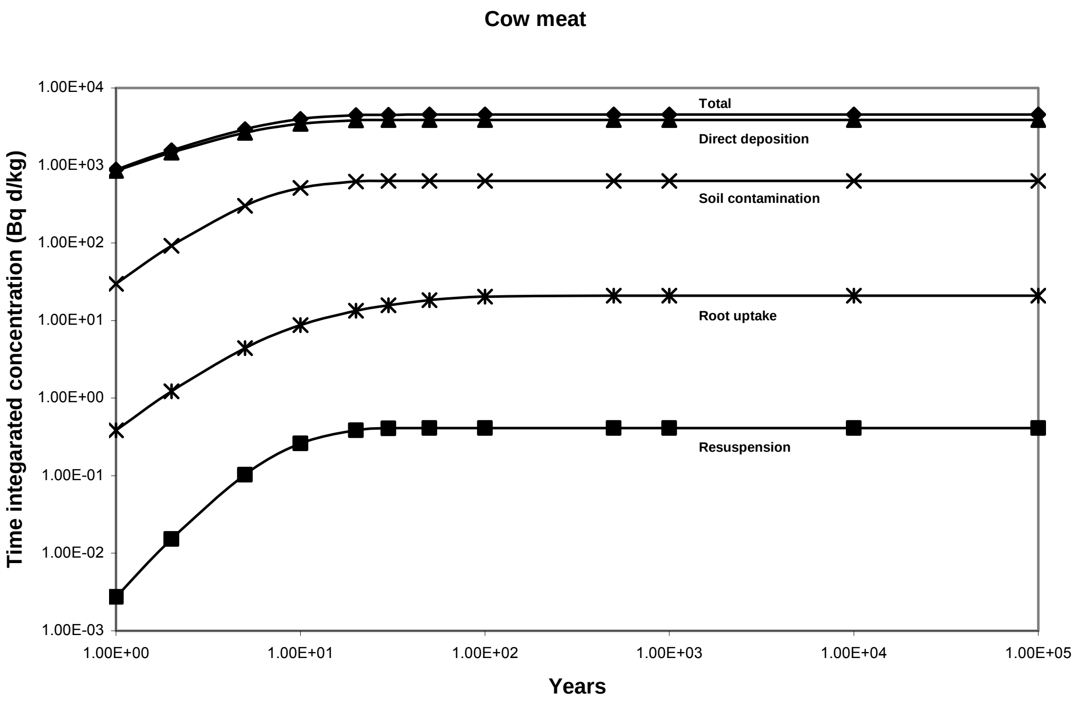


Figure 3.11 The relative importance and time dependence of the important mechanisms for the transfer of plutonium-239 to cow muscle

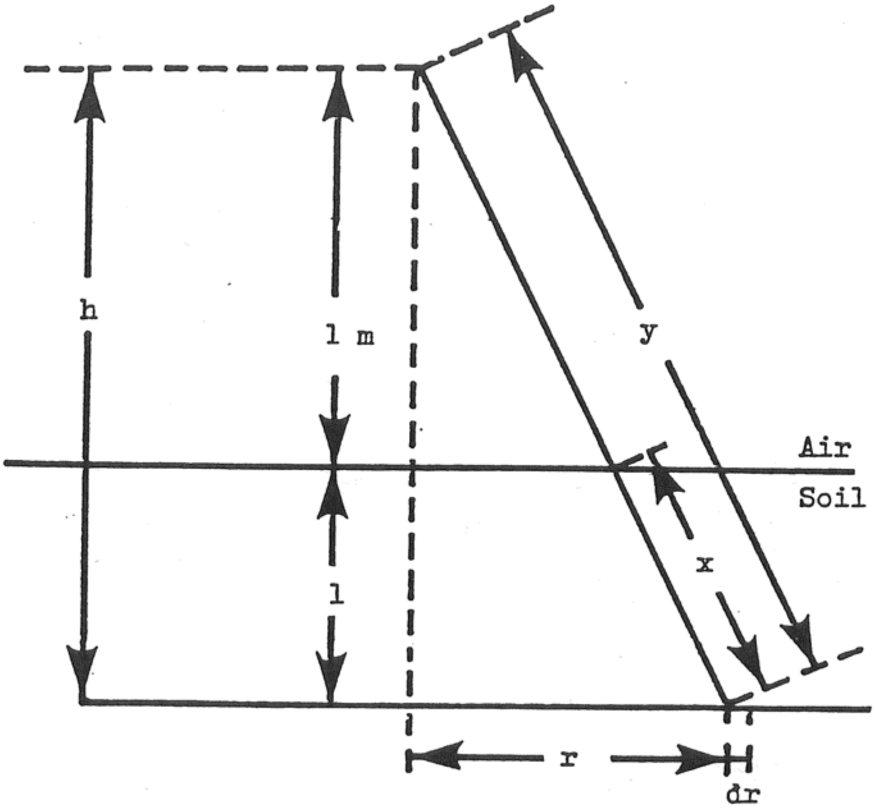


Figure 3.12 Schematic diagram for calculation of external irradiation by photons from radionuclides distributed in soil

4 RELEASES TO THE AQUATIC ENVIRONMENT

4.1 Introduction

Liquid radioactive effluents may be discharged to freshwater (principally rivers), estuarine or marine environments. Radionuclides discharged to rivers are dispersed due to general water movements and sedimentation processes. The principal routes leading to the irradiation of people are: external irradiation from sediments; ingestion of foods derived from the river; drinking water taken from the river; ingestion of foods following the use of river water for irrigation of crops and pasture. Each of these exposure pathways, except the irrigation of crops and pasture with river water, has been considered in PC-CREAM 08. Currently, the subsequent transfer of discharged activity into the sea via an estuary is not included in PC-CREAM 08 but information on estuaries is included in Simmonds, Lawson and Mayall (1995). The dispersion of radionuclides discharged into the marine environment is determined firstly by the local features of the environment, in particular tidal currents and the degree of sedimentation. Subsequent dispersion is influenced by general water movements and sedimentation processes in the larger sea and ocean masses. There are again a number of pathways leading to irradiation of people which include: ingestion of marine foodstuffs, external exposure from beach sediments; inhalation of seaspray. A complete list of pathways considered in CREAM 08 for liquid discharges is given in Table 4.1.

For liquid releases mathematical models have been derived (Simmonds, Lawson and Mayall, 1995) which represent four sectors of the hydrosphere: freshwater bodies (rivers); estuaries; local marine zones; regional marine zones. A discharge into a river may involve the movement of radionuclides through all four sectors, although as already noted, estuaries are currently not included in PC-CREAM 08. For a discharge into the sea it may be only necessary to consider the local and regional marine zones. The models take into account the physical movement and dispersion of water masses, together with radioactive decay. The resulting concentrations in water and sediment then form the input to the calculation of intakes by inhalation and ingestion, exposures from external irradiation and subsequent individual and collective doses.

The radionuclides for which data are presented in this chapter are listed in Table 4.2. These are considered to include the more important radionuclides for aquatic discharges. Other radionuclides could also be included in the models developed provided appropriate parameter values are available.

In the case of discharges to freshwater bodies it may also be necessary to consider the transfer of radionuclides to terrestrial foods through irrigation practices. This transfer can be determined using the terrestrial model described in Chapter 3 section 3.3.3 of this report with the input being predicted water concentrations together with the quantities of water used. Currently, the transfer of radionuclides through irrigation is not included in PC-CREAM 08.

The dispersion models for marine discharges include both northern European waters and the Mediterranean. In each case there is a local model and a regional marine model; the local model acts as an interface between the point of discharge and the regional marine model.

The models described in this chapter are intended for use in determining the radiological consequences of continuous releases to the aquatic environment only. They necessarily contain considerable simplifications of the complex phenomena involved in hydrospheric dispersion. The models represent time dependent behaviour but rely heavily on equilibrium concentration factor data, for example, to estimate the transfer of activity to sediments and aquatic foodstuffs. The flows of water in the marine system considered have been approximated by exchange rates between regional compartments and are considered to represent annual average conditions. This approach can never give strictly accurate representation of seawater movements but is considered adequate for this application given the availability and precision of seafood catch data and the mobility of fish.

A number of different exposure pathways are considered. Some are only relevant for the evaluation of individual dose, while others are particularly important for estimating collective doses. While the generic approach adopted here is useful, its limitations should be recognised, particularly in the case of local individual doses. Where possible site-specific parameter values should be used. Calculated individual doses should not replace calculations based on detailed local habit surveys designed to demonstrate compliance with national regulations.

4.1.1 General modelling principles

In many of the models described in this chapter, compartmental analysis is used to model the movements of activity between parts of the aquatic environment. This technique assumes instantaneous uniform mixing within each compartment, with transfer between compartments being proportional to the inventory of material in the source compartment.

The differential equation which describes the variation of the activity A_i, in compartment i of the model, is of the form:

$$\frac{dA_i}{dt} = \sum_{j=1}^{n} k_{ji} A_j - \sum_{j=1}^{n} k_{ij} A_i - k_i A_i + Q_i \tag{4.1}$$

For all i=1, n where $k_{ii} = 0$

A_i and A_j	are the activities present at time t in compartments i and j (Bq).
k_{ij} and k_{ji}	are the rates of transfer between compartments i and j (s^{-1}).
k_i	is an effective transfer rate from compartment i which takes account of loss of material from the compartment without transfer to another, for example, by radioactive decay (s^{-1}).
Q_i	is a source of continuous input into compartment i, eg, the discharge rate (Bq s^{-1}).

n number of compartments in the system.

The time integral of the inventory in any compartment is given by:

$$A_i' = \int_o^t A_i \, dt \tag{4.2}$$

The time variation and time integrals of the respective inventories are obtained by solution of the sets of simultaneous equations (4.1) and (4.2).

A quantity often used in aquatic models is the volume exchange, R_{ij} in $km^3\ s^{-1}$, from compartment i to compartment j.

$$R_{ij} = k_{ij} V_i \tag{4.3}$$

V_i being the volume of water represented by compartment i (km^3).

The progeny of radionuclides need to be considered where appropriate. For many radioisotopes of concern any radioactive progeny have short half-lives relative to that of the parent; for example, caesium-137 ($t_{½}$ = 30.1 y) and its progeny barium-137m ($t_{½}$ = 2.55 min). If the radioactive half-life of the progeny radionuclide is of the order of a day or less, the behaviour of the progeny in the aquatic environment will essentially be determined by that of its parent. In these cases the two radionuclides can be considered to be in secular equilibrium throughout the environment.

However, for other progeny it is necessary to consider the behaviour of the progeny separately from its parent. Examples of parents and progeny that need to be considered separately are plutonium-241 ($t_{½}$ = 15 y) decaying to americium-241 ($t_{½}$ = 458 y) and zirconium-95 ($t_{½}$ = 0.18 y) decaying to niobium-95 ($t_{½}$ = $9.6\ 10^{-2}$ y).

The in-growth of progeny is modelled by adding for each progeny in a decay chain a further set of compartments, identical to those for the parent. For each set of compartments physical parameters (for example, representing water flow) are the same, but radionuclide-dependent terms such as radioactive decay and sediment sorption are varied between the different sets of compartments according to the particular properties of the radionuclide concerned. Transfer between the sets then occurs between compartments representing the same part of the environment and the transfer rate is the appropriate radioactive decay constant. Thus, progeny are modelled in a “mirror” system of the parent radionuclide.

4.1.2 Calculation of individual and collective doses

The solution of the system of differential equations (4.1 and 4.2) allows the calculation of total activity present in each compartment, as a function of time or the time integral, for a discharge into any one of the compartments. The concentration of activity in the water (or sediment) ($Bq\ m^{-3}$) is then obtained for each compartment by dividing its activity by the compartment volume. These concentrations are then used to calculate intakes by ingestion and inhalation, together with external exposures and hence individual and collective doses. The methods for this are given in the relevant sections below.

4.2 River models

4.2.1 Introduction

Various processes need to be considered in modelling the dispersion of radionuclides released to a river and their subsequent transfer to man. Firstly, those affecting the contamination of the water and sediments: transversal diffusion within the river water; sorption of radionuclides on suspended matter; and sorption of radionuclides on river bank and bed sediment. Secondly, those leading to the contamination of foodstuffs, ie the transfer from water or sediment to fish, drinking water and potentially through irrigation to the terrestrial foodchain.

Rivers are complex, dynamic systems and it is necessary to make various simplifying assumptions in modelling river geometry and processes. Radionuclide interactions complicate the picture further; these are summarised in Figure 4.1. Direct deposition to the water surface and losses by evaporation are generally negligible in comparison to other transfers. The relative importance of the other processes shown in Figure 4.1 depends upon the nature of the effluent released and of the receiving water body. The interaction with particles is complex. The chemical properties of the suspended particles can vary as a function of time depending on a variety of physical (resuspension), chemical and biological processes occurring in water. In the simplest mathematical models radionuclides are assumed to remain in solution and their concentration will decrease by dilution as a result of diffusion and advection. These models will tend to overestimate radionuclide concentrations in water if there is significant adsorption to particles and therefore additional losses from the system due to sedimentation. In addition to the transformation of pollutants in water, more complex models require the parameterisation of terms to describe dispersion, advection and bed-load sediment resuspension under a variety of water flow conditions. Few models have included interaction terms for radionuclides and sediments and fewer still address the effects of different types and concentrations of inorganic and organic complexing agents on radionuclides in river water. All models for assessing the radiological impact of radionuclides released to rivers must include radioactive decay and possibly also the in-growth of progeny.

At the time of radionuclide introduction to a river, advection and dispersion may be the dominant processes. In the long term, biological and chemical processes may become more important. The complexity of aquatic models will depend ultimately on the required accuracy and timescale of predictions. Models designed to predict the real time and small scale changes in radionuclide concentrations will be complex and probably site specific. Their application will be compounded by parameterisation difficulties and lack of relevant data. Universally applicable generic models, although more readily parameterised, cannot be expected to be as accurate as a model developed for a specific river.

In order to estimate doses to individuals and populations appropriate exposure pathways must be modelled and appropriate dosimetric models used. Exposure pathways are considered below. In addition to those pathways arising directly from human interaction with the water body, for example, drinking water or walking on the riverbank, other pathways may arise from the use of the river water for irrigating crops, watering animals or from the use of river sediment as a soil conditioner. Models used for these indirect pathways are likely to be similar to those developed for the transport of radionuclides in the terrestrial environment following deposition from the atmosphere, however, they are not included in PC-CREAM 08.

4.2.2 Modelling approaches

In general there are three different theoretical approaches to river modelling:

1. Simple dilution
2. Hydraulic models
3. Semi-empirical models.

As discussed above the main processes for radionuclide movement in a river are advection and dispersion. However, for some elements interaction with the suspended sediment occurs as chemical sorption and subsequent build up of radioactivity in the sediment occurs. Unfiltered river water, therefore contains activity in the suspended sediment and the water phase. Filtered river water will just contain activity in the water phase. The relationship between the radionuclide concentrations in suspended sediment and in filtered water depend on the sorption characteristic of the radionuclide or element concerned, known as the K_d.

The activity concentration in the unfiltered water, C_{uw}, (Bq m^{-3}) is given by:

$$C_{uw} = C_{fw} + C_{ssl}\alpha \tag{4.4}$$

Where, the activity concentration of solute C_{fw}, also known as the activity concentration in filtered water, is given by:

$$C_{fw} = \frac{C_{uw}}{1 + K_d \alpha} \tag{4.5}$$

and the activity concentration in suspended sediment C_{ssl} (Bq t^{-1}) is simply $C_{fw}\,K_d$, or:

$$C_{ssl} = \frac{C_{uw} K_d}{1 + K_d \alpha} \tag{4.6}$$

Where K_d is the appropriate sediment-water distribution factor ($m^3\ t^{-1}$) and α is the suspended sediment load (t m^{-3})

The sediment distribution coefficient or K_d is widely used in the modelling of radionuclide transfer in aquatic systems. It represents the ratio between concentrations of an element on sediment and concentrations in solution under equilibrium conditions ($m^3\ t^{-1}$) (see section 4.2.3.3 for semi-empirical model). The K_d value gives an indication of the likely behaviour of an element in the aquatic environment. For example, technetium in the freshwater aquatic environment has a relatively low K_d value (200 $m^3\ t^{-1}$); therefore much of the activity remains in solution and is dispersed by the action of the currents. Conversely, plutonium in the freshwater aquatic environment has a relatively high K_d (~100000 $m^3\ t^{-1}$); therefore much of the activity adheres to sediments close to the discharge point and is not dispersed widely. Many other factors must be taken into account, such as the decay rate of the radionuclide and the characteristics of the aquatic environment, such as flows and suspended sediment loads. Values for these parameters can be obtained from in-situ measurements, laboratory studies or derived from the characteristics of the element.

4.2.2.1 Simple dilution

Simple dilution models assume that the effluent is diluted by the total river volume as soon as it is released. No attempt is made to divide the river system into compartments. A major problem with this approach is that sediment interactions are usually ignored, although these processes are important in the transport and removal of radionuclides from river water. The models are nevertheless applicable to radionuclides which do not interact strongly with sediments. More complicated dilution models incorporate terms for the dilution by the flow of the river and radioactive decay of radionuclides during the time of traverse between the point of release and a specified point downstream. For example, Murray and Avogadro (1978) have used this type of approach but have taken sedimentation effects into account.

PC-CREAM 08 includes three types of simple dilution models as screening models and these are discussed below.

(a) Simple screening model.

An estimate of the radionuclide activity concentration in river water at the outfall, averaged over the period of interest can be made as follows:

$$C_{uw} = \frac{Q}{F} \tag{4.7}$$

where, C_{uw} is the activity concentration in unfiltered water at the outfall assuming instantaneous dilution (Bq m^{-3}), Q is the annual discharge rate of the radionuclide (Bq s^{-1}) and F is the volumetric flow rate of the river at the outfall (m^{3} s^{-1}).

The assumptions inherent in the use of such a model are:

- The discharge rate over the period of interest is constant;
- The flow rate of the river over the period of interest is constant;
- There is no dilution of activity in the effluent itself;
- The radioactive effluent is diluted instantaneously and completely in the total flow of the river;
- Radioactive decay is ignored.

A major problem with this approach is that sediment interactions are not modelled explicitly, rather it is assumed that activity concentrations in river sediments are the same as those in suspended sediments, using equation 4.6. The sedimentation processes are important in the transport and removal of radionuclides from river water.

The model may therefore be more applicable for radionuclides which do not interact strongly with sediments, such as tritium or for distances immediately downstream of the discharge point. The model may be used as a screening tool to give cautious estimates of exposures associated with radionuclides in the water column.

(b) Extended screening models

An extension of the simple dilution model incorporates terms for the dilution by the flow of the river, radioactive decay and downstream transit times. This model is more appropriate for short-lived radionuclides with a half life of less than 1 year, eg, ^{32}P or ^{65}Zn.

The assumptions inherent in this approach are:

- The discharge rate over the period of interest is continuous and constant;
- The river flow rate over the period of interest is constant;
- The dilution of the effluent in river water is accounted for using a dilution factor (the dilution of 1 m^3 of effluent in 1000 m^3 of water is represented by a dilution factor of 1000);
- Activity concentrations in water at a point downstream are function of the degree of dilution, the transit time and radioactive decay.

Two types of extended screening models are included in PC-CREAM 08 with complete and incomplete mixing. These can be used to estimate radionuclide concentrations in unfiltered water at any point downstream of the outfall, averaged over the period of interest

Radioactive effluent is generally released into rivers with a discharge rate that is low compared with the much greater flow rate of the river itself. Complete dilution only occurs some tens of kilometres downstream and activity concentrations in water in some areas may be higher than the mean activity concentration calculated assuming that the dilution is complete (Simmonds, Lawson and Mayall, 1995). Radionuclide concentrations in bank and bed sediments are dependent on the activity concentration in the adjacent water column which varies transversely across the river.

Complete mixing
If complete mixing is assumed, the activity concentration in the river water is given by:

$$C_{uw} = \frac{Q}{F} e^{-\lambda t} \tag{4.8}$$

Where λ is the decay constant (s^{-1} or y^{-1}) and t is the transit time at the point, in corresponding units. The transit time in seconds can be calculated using the following equation:

$$t = \frac{x}{v_w} \tag{4.9}$$

Where x is downstream distance (m) of the point and v_w is the water velocity (m s^{-1}). The river water velocity is related to the volumetric flow rate of the river at the outfall F (m^3 s^{-1}) through the following equation:

$$v_w = \frac{F}{wd} \tag{4.10}$$

Where w (m) and d (m) are the width and the depth of the river, respectively.

Using the activity concentration in unfiltered water given by Equation 4.8 it is also possible to estimate activity concentrations in both solution (Equation 4.5) and associated with suspended sediment (Equation 4.6).

Incomplete mixing
For incomplete mixing, activity concentration in the river water is given by:

$$C_{uw} = \frac{Q}{ED} e^{-\lambda t} \tag{4.11}$$

Where E is the effluent flow rate (m^3 s^{-1}) and D is the dilution factor (dimensionless), which accounts for the degree of effluent dilution in river water.

Using the activity concentration in unfiltered water given by Equation 4.11 it is possible to estimate activity concentrations in both solution (Equation 4.5) and associated with suspended sediment (Equation 4.6).

When incomplete mixing is assumed, it is important to check that the dilution factor selected is compatible with incomplete mixing. Consider an example in which a high dilution factor of 2000 is selected, equivalent to 1 m^3 of effluent dispersed in 2000 m^3 of water. This may give rise to activity concentrations lower than those expected at complete mixing, when the radioactive discharge is dispersed in the total flow of the river, ie application of the dilution factor exceeds the total flow of the river. Therefore, if the product of the effluent flow rate, E, and the dilution factor, D, is greater or equal to the river flow, F, then Equation 4.8 for complete mixing should be used.

4.2.2.2 Hydraulic models

PC-CREAM 08 does not include any examples of this type of model. These models have been developed from water flow studies to describe water quality and sediment transport in rivers. The US Nuclear Regulatory Commission (NRC) (USNRC, 1976) has recommended their use in calculating doses from nuclear installations on river sites. Hydraulic models may or may not incorporate interactions with sediments. Zand et al (1976) used a hydraulic model with no sediment interaction to trace the migration of Cl^-, Na^+ and Se^{++} ions in a California creek. The model assumed instantaneous dispersion across the channel, constant unidirectional flow and was solved analytically. Gloyna et al (1963-1972) developed a sediment interaction hydraulic model to describe the transport and dispersion of radionuclides in an experimental flume. This model ignored mass transfer of sediment but permitted adsorption of radionuclides on to bed sediment or aquatic plants. A major drawback of this model is its limitation to rivers with low suspended sediment load, an unlikely case for most rivers.

The inclusion of parameters to account for interaction with sediments, deposition and resuspension leads to the most complicated hydraulic models with severe problems for evaluation of critical parameters and calibration. An example of a model of this type has been developed by Onishi and Wise (1979). This scheme takes account of the advection and dispersion of contaminants in the water and of suspended particles, and the deposition and resuspension of sediments. The adsorption-desorption mechanisms are assumed to be completely reversible and dependent on particle size. Other models account for interaction of contaminants with biota, effects of tributaries, weirs and industrial sources of contaminants (eg, Cox, 2003).

4.2.2.3 Semi-empirical models or dynamic model

Although hydraulic models are a good representation of the processes involved in water and sediment transport they are difficult and expensive to validate, parameterise and apply to radiological problems. For systems such as PC-CREAM 08 generic models are required that are sufficiently accurate but which require the derivation of the minimum number of parameters. It is also possible to use simpler models when considering routine releases of radioactivity where the interest lies in long-term averages.

The semi-empirical model adopted in PC-CREAM 08 offers a compromise; it retains some of the spatial and temporal resolution of hydraulic models but simplifies radionuclide-sediment interaction by employment of the empirically derived distribution coefficient (K_d). The proposed model is that developed by Schaeffer (1976) and described in the original 1979 CEC methodology report (NRPB and CEA, 1979) and Simmonds, Lawson and Mayall (1995). In Schaeffer's model it is assumed that the concentrations of radionuclides in solution decrease exponentially downstream from the point of release due to dilution and adsorption of radionuclides on to sediments. The model assumes a constant and continuous discharge and instantaneous dilution of effluent in the total flow of the river at the point of discharge. Schaeffer also points out that bedload sediment will move downstream at a slower rate than the water and that fluvial sediments could be a major source of radiation dose to people living along the river. The implementation of this model in PC-CREAM 08 is referred to as the "dynamic model".

The Schaeffer parameter k´, represents the potential removal of radionuclides to bedsediment and accounts for exponential losses. Values of k´ must be determined empirically for each radionuclide concerned. As a first approximation values of k´ obtained from measurements in the river Rhône may be used using a simple "rule of thumb" based on the element's freshwater K_d value:

$$\begin{cases} if \quad K_d \leq 10^4\ m^3 t^{-1} \quad then \quad k' = 2\,10^{-6}\ m^{-1} \\ if \quad K_d > 10^4\ m^3 t^{-1} \quad then \quad k' = 1\,10^{-5}\ m^{-1} \end{cases}$$

Originally (Simmonds, Lawson and Mayall, 1995) k´ was given a value of 0 for K_d values of less than 1 10^3 but now the lower k´ value of 2 10^{-6} is considered more appropriate for all values of K_d less than or equal to 1 10^4. Radionuclide specific parameters for use in the river models for decay constant, K_d and k ´ for the common radionuclides used in PC-CREAM 08 are given in Tables 4.2 and 4.3**.**

The general compartmental model structure used in the dynamic model included in PC-CREAM 08 is shown in Diagram 4.1. As seen below the model includes the transfer of activity within each river section between dissolved water, suspended sediment and bed sediment. Transfer of activity then occurs down stream to the next river section for each compartment.

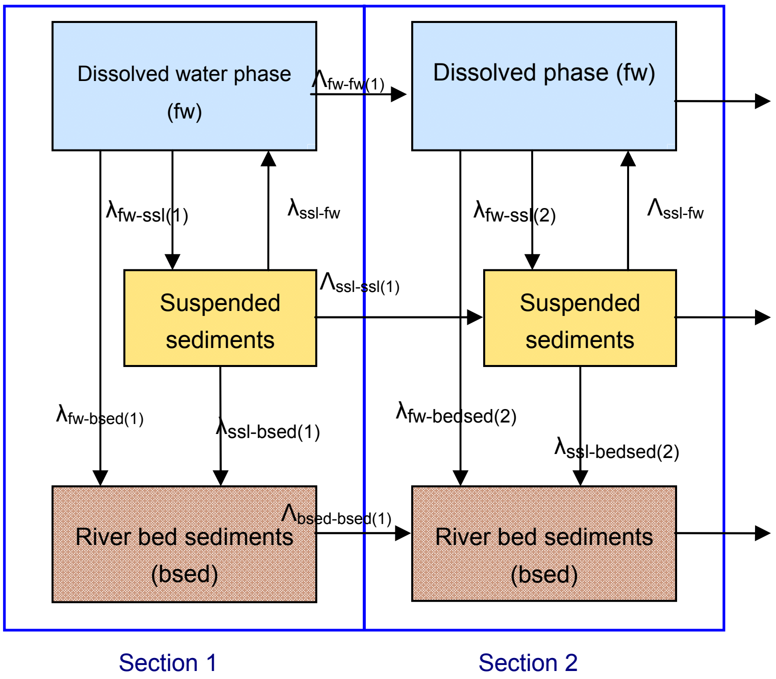


Diagram 4.1 The Schaeffer river compartmental model used in PC-CREAM 08

The transfer rates are given as follows:

Transfer from filtered water to suspended sediment:

$$\lambda_{fw-ssl} = K_d \alpha \, \lambda_{ssl-fw} \tag{4.12}$$

Where λ_{ssl-fw} = transfer of activity from suspended sediment to filtered water= 3.15 10^7 y^{-1}

Transfer from suspended sediment to bed sediment:

$$\lambda_{ssl-bsed} = k' v \tag{4.13}$$

Transfer from filtered water to bed sediment:

$$\lambda_{fw-bsed} = k' v \tag{4.14}$$

Transfer between water compartments along river section. The same transfer rate is also calculated for suspended sediment load.

$$\lambda_{fw-fw} = \frac{v}{l} \tag{4.15}$$

Transfer between bed sediment compartments along river section

$$\lambda_{bsed-bsed} = \frac{v_{bsed}}{l} \tag{4.16}$$

Where v_{bsed} = 1 10^{-4} x velocity of river water (v) (in absence of measurement data)

Where:

K_d = Sediment sorption coefficient (m^3 t^{-1}) (See Table 4.3)

α= Suspended sediment load (t m^{-3}) (for river section)

k' = Schaeffer's sediment interaction coefficient (m^{-1}) (See Table 4.3)

v = velocity of river (m y^{-1}) (for river section)

l = Length of river section (m) (for river section)

λ = Radioactive decay constant y^{-1}

Application of the proposed river model requires the division of the river into sections. Definition of the sections depends on the physical characteristics of the river and also the utilisation of river water and sediments. Within each section river characteristics such as water velocity, bed sediment velocity and amount of suspended sediment are considered constant.

Examples of the application of this river methodology to the rivers Loire and Rhône are given in Appendix 4.1, together with descriptions of the rivers Rhine and Meuse.

Improvements of the Schaeffer model

These improvements are not currently in PC-CREAM 08 and are discussed more fully in Simmonds, Lawson and Mayall, (1995).

a) Plume model

A plume model developed by the Commissariat á l'Energie Atomique (CEA) for large rivers is described in Simmonds, Lawson and Mayall (1995). Radioactive effluent is generally released into rivers from the banks and the discharge rate is low compared with the much greater flow rate of the river itself. Therefore, complete dilution only occurs some tens of kilometres downstream and activity concentrations in water in some areas may be higher than the mean activity concentration calculated assuming that the dilution is complete. Radionuclide concentrations in bank and bed sediments are dependent on the activity concentration in the adjacent water column which varies transversely across the river.

A further model was developed from a screening model in NCRP 123 (NCRP, 1996). This model accounts for river water velocity, width and depth. The model described here is based on a simplified version of the NCRP model, which allows simple scaling or correction factors to be included in the river model.

These correction factors were calculated for example rivers presented in Diagram 4.2. The partial mixing correction factor represents the ratio of the activity concentration in unfiltered water in the plume to the activity concentration in unfiltered water if the same point were assumed to be completely mixed in the total width of the river.

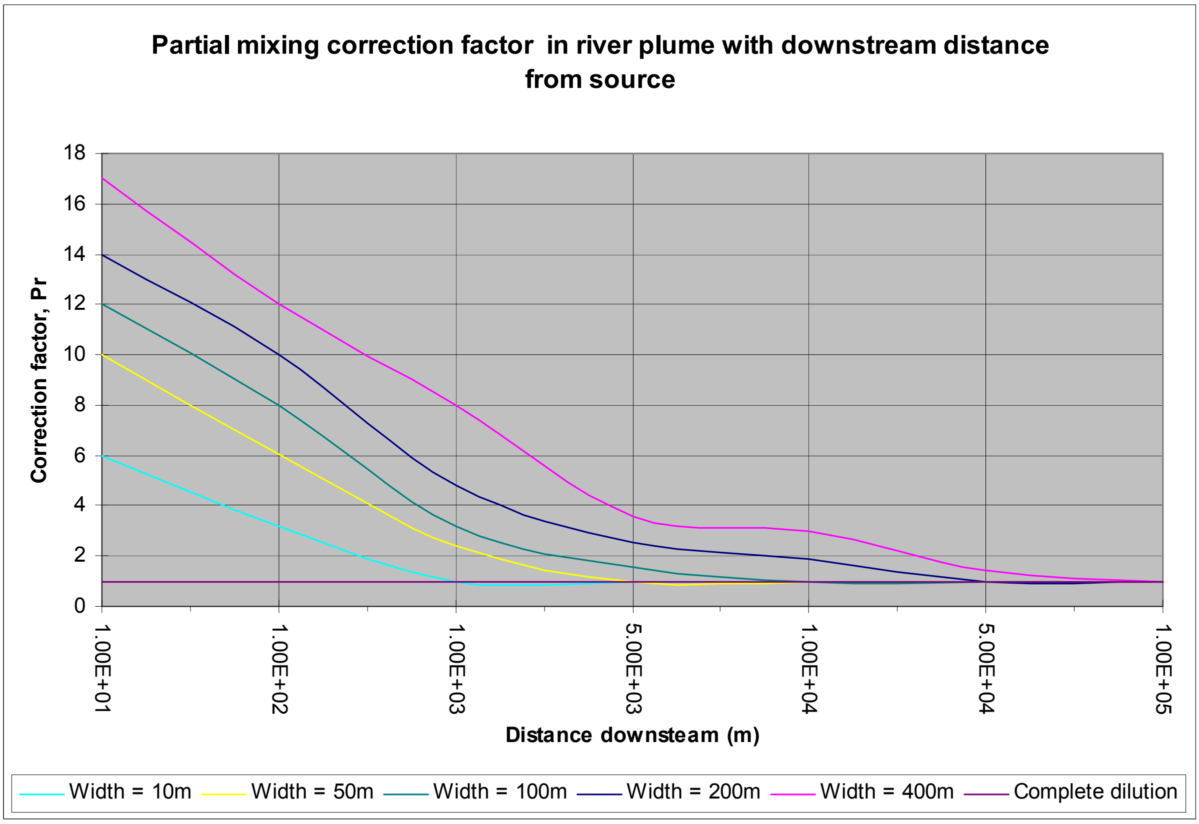


Diagram 4.2 Partial mixing correction factors for implementation of plume river model

b) River bank model

In addition to bed sediment transport proposed by Schaeffer, the modified scheme incorporates a new riverbeach or riverbank compartment. The riverbeach compartment represents the unvegetated, semi-washed zone of sediments on the margins of the main channel. In upland channels this zone is generally boulder-strewn, in lowland reaches it is characterised by fine clays and silts. Alternating reaches of predominantly depositional or erosional nature are a further feature of the riverbeach. In contrast to the high turnover of deposits in the beach zone, the riverbank, at a slightly higher elevation, floods less frequently. Deposition is dominated by dredged materials or the occasional flood, and erosion operates by undercutting or resuspension.

4.2.3 Riverine exposure pathways

The estimation of doses received from the ingestion of drinking water, fish and agricultural products contaminated by irrigation practices follows from the calculation of the radionuclide concentration in the river water. Contamination of river sediment can also lead to doses to people on the river banks.

Ingestion of drinking water

River water, or water from the water-table close to a river, may be extracted, treated and utilised as drinking water. The concentration of activity in drinking water is normally less than the concentration in river water by a factor which varies with the methods of extraction and treatment. Radionuclides present on suspended sediments can easily be removed by filtration and/or coagulation and settling (Dionian and Linsley, 1983). However, the removal of soluble radionuclides by treatment plants is less effective and much depends on the nature of the chemical processes at the treatment plant. Lowland river water is usually treated by slow sand filtration or coagulation and filtration. The effectiveness of these removal processes depends on the radionuclides, the flocculation conditions, composition of the contaminated water and the nature of the contamination. Default removal efficiency values for use in the absence of site specific data are given in Table 4.4 (Brown et al, 2008a; Brown et al, 2008b). Alternatively, it can be assumed that river water is extracted directly from the river and undergoes a single filtration which removes suspended sediment. In this instance the activity in drinking water is assumed to be equal to that of filtered river water with no additional factor applied to allow for treatment.

In PC-CREAM 08 radiation doses for adults, children (10 y) and infants (1y) from ingestion of radionuclides in drinking water are calculated from the activity concentrations in filtered river water obtained from the different models. For the dynamic model the drinking water can be assumed to be extracted from any of the sections included in the model, downstream of the discharge point and there is an option to include water treatment losses. For the simple dilution models the doses from ingestion of drinking water are given by:

$$E = C_{fw}\, \mathrm{I}_w\, \mathrm{H}_{\mathrm{ing}} \qquad (4.17)$$

Where:

E is the individual effective dose from drinking water (Sv y^{-1}).

C_{fw} is the activity concentration in filtered river water (Bq l^{-1}).

I_w is the intake rate of drinking water (l y^{-1}).

H_{ing} is the effective dose per unit intake by ingestion (Sv Bq^{-1}).

For the dynamic river model the doses from ingestion of drinking water are given by:

$$E_i = C_{fw\,i}\, \mathrm{I}_w\, \mathrm{H}_{\mathrm{ing}} Treat \qquad (4.18)$$

Where:

E_i is the individual effective dose for river section, i (Sv y^{-1}).

C_{fwi} is the concentration in filtered river water in river section i (Bq l^{-1}).

I_w is the intake rate of drinking water (l y^{-1}).

H_{ing} is the effective dose per unit intake by ingestion (Sv Bq^{-1}).

Treat = 1- Removal efficiency (R)/100 (dimensionless - optional) – Values for removal efficiency (R) are given in Table 4.4.

Collective doses from ingestion of drinking water are not currently calculated directly in PC-CREAM 08. However, it is possible to use the results from PC-CREAM 08 to estimate collective doses from drinking water as was done for a recent EC study (Harvey et al, 2008). In this study the population was assumed to drink a fraction (50%) of their water from the river, as drinking water was also known to be obtained from ground water wells. The collective dose for drinking water from the relevant rivers was calculated by summing the product of the estimated individual dose from drinking water in each river section and the population living around each river section, (Harvey et al, 2008).

Ingestion of fish

The transfer of activity into fish is calculated using an element dependent concentration factor. These relate the concentration of activity in the edible part of fish to the concentration in filtered water, ie Bq t^{-1} per Bq m^{-3}. Fish concentration factors for a number of elements are given in Table 4.3. The concentration factors assume instantaneous transfer of activity into fish from the compartments of the river model. It is recognised that the time dependence of radionuclide transfer into fish is unlikely to be accurate; activity in fish from short-lived radionuclides may be overestimated due to the assumption of instantaneous transfer from the river water to fish. If data are available then radionuclide specific concentration factors may be used for short lived radionuclides rather than the element dependent values given in Table 4.3 (Smith, 2006).

In PC-CREAM 08 doses from ingestion of fish are calculated as follows:

$$E = Cfood\, I_f\, H_{ing} \tag{4.19}$$

Where E is the individual effective dose from ingestion of fish (Sv y^{-1}).

Cfood is the activity concentration in fish (Bq t^{-1}).

I_f is the intake rate of fish (t y^{-1}).

H_{ing} is the effective dose per unit intake by ingestion (Sv Bq^{-1}).

Where:

$$Cfood = C_{fw} CF \tag{4.20}$$

C_{fw} is the activity concentration of the radionuclide in filtered water (Bq m^{-3})

CF is the element dependent concentration factor for fish (Table 4.3) (Bq t^{-1} per Bq m^{-3})

Collective doses for discharges to rivers are not currently included in PC-CREAM 08. However, if required the collective intake of a given radionuclide from ingestion of fish per section of the river can be calculated by multiplying the calculated activity concentration in the edible fraction of fish by the quantity of fish caught in the river section and the fraction of the fish which is edible.

External exposure

External exposure of people can occur through immersion in the river water, either during bathing or fishing, or from occupancy of the river bank or boats. The maximum amount of time people spend engaged in such activities is site specific and must be determined by habit surveys. Data on the collective occupancy of river water or river banks are sparse. The external exposure pathway currently modelled in PC-CREAM 08 is that from contaminated river bank sediment. This is probably the most significant pathway in terms of dose and likelihood of occurrence (Jones et al, 2002). In the models used in PC-CREAM 08 it is assumed that the activity concentration in the river bank sediment is the same as that in the bed sediment. In the dynamic model the activity concentration in sediment will build up over the time period of the discharges and for long lived radionuclides the estimated activity concentration in the sediment may be greater than that estimated by the simple dilution model.

In PC-CREAM 08 the external doses due to beta and gamma emissions from radionuclides in river bank sediments are calculated using a similar approach. Gamma doses are calculated using the approach developed by Hunt (1984). The following formulae are used to calculate doses in PC-CREAM 08:

For gamma

$$E = C_{bedsed}\ GAMM\ DF\ Conv\ 0.87\ DTW\ Occ \tag{4.21}$$

Where:

E = Effective dose in μSv y^{-1}

C_{bsed}= Activity concentration in sediments (dry) (Bq kg^{-1})

GAMM= Gamma energy (MeV)

DF= Dose rate in sediment (wet) = 0.288 (μGy h^{-1} per Bq g^{-1} per MeV) (Hunt, 1984).

Conv = Conversion from Bq kg^{-1} to Bq g^{-1} (1 10^{-3})

0.87 = Sv per Gy

DTW = dry to wet activity concentration conversion = 0.9

Occ = Occupancy time on sediment (h y^{-1})

For beta:

$$E = C_{bsed}\ Dens\ t\ DF\ wt\ Occ\ Conv \tag{4.22}$$

Where:

E = Effective dose in μSv y^{-1}

C_{bsed}= Activity concentration in sediments (Bq kg^{-1}) dry

Dens = 1500 kg m^{-3}.

t = Thickness of deposit 0.01m

DF = beta skin dose factor at 1m (Sv y^{-1} per Bq m^{-2}) (Holford, 1989)

Wt= skin weighting factor (0.01)

Occ = Occupancy time on sediment (h y^{-1})/8760 h y^{-1}

Conv = Conversion from Sv y^{-1} to (μSv y^{-1}) (1 10^6)

Ingestion of agricultural products contaminated by irrigation and soil conditioning practices

Radioactivity can reach man from crops which have been irrigated with contaminated river water or treated with dredged river bed sediments to act as soil conditioners or fertiliser. PC-CREAM 08 currently does not consider these pathways. Doses can however be calculated from the activity concentrations in unfiltered water and sediment that form part of the output of the PC-CREAM 08 river model, together with the use of the PC-CREAM 08 FARMLAND model.

Spray irrigation

This can be modelled by taking account of the transfer of activity to the external surfaces of the plants, root uptake and translocation, in a manner similar to that described for the deposition of activity from the atmosphere (see Chapter 3). The transfer coefficients used to describe the movement of activity when the source is via irrigation are generally identical to those adopted in Chapter 3 for atmospheric deposition. A possible exception is the fraction of activity intercepted by the plant surfaces during the deposition process. If spray irrigation is carried out at a high rate then the fraction deposited on vegetation will be lower (Pröhl and Hoffman, 1996).

The models in Chapter 3 can be used to estimate the time integrated concentrations (Bq y $kg^{-1)}$ in agricultural produce derived from land irrigated at a rate of 1 Bq m^{-2} s^{-1} for one year. The rate of deposition for each radionuclide is obtained by multiplying the concentration in unfiltered water, in Bq m^{-3}, by the irrigation rate in m^3 s^{-1} per m^2. It is noted that it may be possible for activity to reach man by other routes from irrigation practices such as resuspension of contaminated soil and subsequent inhalation, or external irradiation from contaminated soil. It is assumed that these pathways are negligible in terms of collective dose compared with the ingestion of agricultural products. This is because inhalation and external irradiation doses are dependent on the proximity of individuals to contaminated agricultural soils. Such proximity is unlikely for large sections of the population. However, these pathways may be important when considering doses to specific individuals.

Sediment application

Radionuclides can also be transferred to foodchain pathways by application of river sediments as a soil conditioner. Over subsequent years, these operations result in the accumulation of radionuclides in agricultural soils. Dredging operations can transfer bed sediment directly to agricultural land in which case the rate of deposition of each radionuclide is obtained by multiplying the concentration in river bed sediment, Bq m^{-3}, by the application rate in m^3 y^{-1} per m^{-2}. Alternatively dredged materials may be placed on the riverbank for later transfer to agricultural soils. The rate of radionuclide deposition in this case is the product of the concentration in riverbank sediment, Bq m^{-3}, and the application rate, m^3 y^{-1} m^{-2}. In all cases soil conditioning materials are assumed to be removed from the top layer of the relevant river sediment compartments.

Animals' drinking water (unfiltered river water)

The consumption of unfiltered river water by livestock can also lead to the accumulation of radionuclides in meat and milk products. The concentration of radionuclides in beef for example, is calculated as follows. The annual intake of radionuclides by cattle is derived from the product of the concentration of radionuclides in unfiltered river water, Bq m^{-3}, and the annual consumption of water, m^3 y^{-1}. Using uptake factors, the radionuclide concentration in meat can be derived in Bq kg^{-1} (see Chapter 3).

4.3 Estuary models

Estuaries represent a particularly complex aquatic environment. Within estuaries there is a complex interaction between the tides in the open sea and variable freshwater flows. The salinity varies both in space and time, which may lead to changes in the chemical forms of the radionuclides and hence changes in the uptake of the radionuclides by sediments and biota. Estuaries often exhibit complex patterns of sediment deposition and resuspension.

The degree to which it is necessary to model the behaviour of radionuclides within an estuary depends on the particular application of this methodology. In particular, it depends upon the site of the discharge and the importance of exposure pathways originating within the estuary. For example, a complex estuary model may be desirable if the discharge takes place within or near to the estuary and the internal dynamics of the estuary are therefore important in determining the movement of radionuclides between the marine and freshwater environments. Similarly, a complex estuary model may be desirable when surveys of habits indicate that exposure pathways originating within the estuary make a significant contribution to the calculated doses arising from the discharge. In this case also the significance of the estuary may depend upon whether doses to individuals are being calculated, which are usually highest near to the discharge point, or collective doses, in which case contributions to the doses from freshwater pathways and large-scale marine dispersion may greatly exceed those arising from the estuary.

Models for discharges to estuaries are not currently included in PC-CREAM 08. However, in a recent study for the EC (Harvey et al, 2008) activity concentrations in the marine environment were estimated following discharges to rivers using a simple interface model that is described below. Information is provided on more complex models for discharges of radioactivity to estuaries in Simmonds, Lawson and Mayall (1995).

4.3.1 Simple interface model

Where a complex estuary model is not justified, then a simple interface to represent the movement of radionuclides between the freshwater and marine environment may suffice. This interface should take into account the movement of radionuclides in the water phase and those absorbed onto suspended particulate material. It should also take into account the downstream movement of radionuclides within the bed of the freshwater system and the possible desorption of radionuclides as these bed sediments enter the marine environment. The marine environment near to the estuary will be represented by a local marine compartment (see section 4.4.1) with its associated bed sediments (see section 4.4.2.1). Some of the radionuclides present in the freshwater bed sediments will be partially desorbed as these sediments pass through the estuary and hence they will enter the water phase of the local marine compartment. The radionuclides remaining with the bed sediment are assumed to enter the top sediment compartment associated with the local marine compartment. The fraction of the inventory of each radionuclide in the freshwater bed sediments which is desorbed depends upon the relative values of the freshwater and coastal marine sediment distribution coefficients (K_ds), as described previously (Simmonds, Lawson and Mayall, 1995). Radionuclides in the water column, that is, dissolved in the water phase and adsorbed on suspended sediments, are assumed to reach a new equilibrium instantaneously as they pass from freshwater to seawater, in accordance with the respective K_ds.

The activity concentration in the marine water which has been desorbed from the freshwater sediment is given as:

$$C_{mwdsorb} = C_{bsed} Desorb \tag{4.23}$$

While, the activity concentration in the marine sediment which has been desorbed from the freshwater sediment is given as:

$$C_{mbseddsorb} = C_{bsed}(1 - Desorb) \tag{4.24}$$

Where:

$C_{mwdsorb}$= Activity concentration in unfiltered sea water from activity desorbed from freshwater sediment (Bq m^{-3})

$C_{mbseddsorb}$= Activity concentration in marine sediment from activity desorbed from freshwater sediment (Bq t^{-1})

C_{bsed} = Activity concentration in freshwater bed sediment (Bq t^{-1})

Desorb = Nuclide specific desorption factor (see Table 4.5)

For the interface between the estuary and the marine model, the large size of most of the marine compartments necessitates the use of a small local compartment close to the discharge point if modelling of the small-scale dispersion of radionuclides is to be adequate. The type of local marine compartment near an estuary depends upon the chosen estuary model. When the estuary is represented by a simple interface then a local marine compartment should be used, with the parameter values given in Table 4.6. If the estuary was represented by a multi-compartment model, then one of these compartments should be used to represent the local marine dispersion.

The increasing salinity with distance downstream towards the sea results in changes in the values of the element-specific parameters, biota concentration factors and sediment distribution coefficients. Both the freshwater and marine values of these parameters exhibit a large uncertainty for most elements. One approach is that, wherever the salinity is less than 0.5%, then freshwater values are used and that marine values are used when the salinity exceeds this value (Jackson, 1985).

4.3.2 Estuarine exposure pathways

Human activity in and around estuaries may be higher than in adjacent fresh water or marine areas. At low tide exposed mud flats may be frequented by people digging for fishing bait or bird-watching. In general, the pathways by which people may become exposed to radionuclides in or from estuaries are the same as those in the marine environment (see section 4.4.2.1). The salinity of the water in estuaries generally precludes its use for irrigation purposes.

Generally, the activity transferred to marine water from freshwater sediment is small compared to the activity transferred via water. For most radionuclides this is essentially zero apart from isotopes of phosphorous, sulphur, chlorine, calcium and ruthenium where activity lost from the sediment via desorption is more significant.

4.4 Marine modelling

In order to calculate the collective and individual exposures resulting from discharges of activity to sea, it is necessary to model the dispersion of radionuclides in marine waters, their possible reconcentration in environmental materials and the pathways to man.

The models described below represent the dispersion of radionuclides in European coastal waters, the Atlantic Ocean and other world oceans. For a release of radionuclides at a particular location, the models calculate the time-dependent activity concentrations in the various sea areas taking into account advection and diffusion, radioactive decay and the interaction of radionuclides with suspended and seabed sediments. The calculated concentrations of radionuclides in environmental materials are used in the calculation of collective and individual exposures by taking into account the appropriate habit data.

Activity may be discharged into the marine environment either directly or indirectly via a freshwater body. In the latter case the environmental characteristics of the freshwater system will influence the fraction of activity which eventually reaches the sea; this fraction can be evaluated using the models described in the previous section. For direct discharges to the marine environment, the local environmental conditions may be important in determining the maximum individual exposures and also influence the amount of activity which becomes more widely dispersed, thus exposing populations at a distance from the discharge. For this reason the model chosen to represent dispersion in the marine environment is subdivided into 'local' and 'regional' components which are interfaced. The regional model can be interfaced with a number of local models, each representing discharges from a single site, if the overall radiological impact of, say, a nuclear power generating programme, is required.

The model used in PC-CREAM 08 was adapted from a model developed by Working Group D of the MARINA II project, to assess the radiation exposures to the member states of the European Union from radioactivity in north European waters (Simmonds et al, 2002). This model was based on that developed for the first MARINA project and described in Simmonds, Lawson and Mayall (1995). The original model consisted of 44 compartments representing northern European waters and relevant adjacent seas. The revised model (Simmonds et al, 2002) has an increased number of compartments (72) to better represent transfer in particular marine areas and also an improved representation of the transfer of radionuclides to sediment. The full MARINA II model requires a significant amount of computer resources to run, particularly if progeny are included in a parallel system of compartments. Therefore, in PC-CREAM 08 a simplified version of the model with 55 compartments has been implemented.

4.4.1 Local models

Dispersion on a local scale, up to a few kilometres from the discharge point, is modelled by a single well-mixed water compartment together with associated sediment compartments (see section 4.4.2.1). The local box is interfaced with the regional marine model and exchanges water and suspended sediment with the adjacent regional marine model compartment (Figure 4.2). This approach has been used previously in radiological assessment models (Simmonds et al, 2002; Hunt, 1982). Its adequacy depends upon the values chosen for the local box parameters; the derivations of three 'generic' types of local box (estuarine, coastal and exposed coastal) are given in Camplin, Clark and Delow (1982). If detailed assessments of the local radiological impact of discharges from a particular site are to be undertaken, then site-specific hydrographical data will be required; methods for deriving local box parameters from such data have been described elsewhere (Maul, 1985). In some cases a detailed hydrodynamic model may be indicated (see, for example, Robeau, Patti and Charmascon, 1988) but discussion of such models is beyond the scope of this report.

Local compartment parameter values appropriate for EC sites discharging to the marine environment are given in Table 4.6.

4.4.2 Regional marine model

Figures 4.3 to 4.6 illustrate the various regions used in the regional marine model. As noted above the 72 compartments include in the MARINA II model (Simmonds et al, 2002) has been reduced to 55 in PC-CREAM 08 by consolidating the compartments in the Atlantic, Arctic, Barents and Kara seas. This was considered justified as generally discharges are not made directly to these regions and tests were carried out to ensure that the reduced model gave similar results to the full MARINA II model.

Each of the water compartments has associated suspended sediment, and water compartments in contact with the seabed have underlying seabed sediment compartments. The model for the Mediterranean Sea is described in Appendix 4.3. The limited exchange between the Mediterranean and the Atlantic through the Straits of Gibraltar allows the models to be used to a large extent independently. Therefore, PC-CREAM 08 contains two marine models: a) the Northern European model which includes the compartments shown in Figures 4.3 to 4.6 with the Mediterranean Sea included as a single compartment; b) the Mediterranean model which models the Mediterranean Sea in detail but includes all other sea areas in just two compartments.

4.4.2.1 Description of the model

The model can conveniently be described under three headings; water movements, sediments and exposure pathways.

(a) Water movements

The model describes the significant movements of water in European coastal seas by a system of interlinked compartments. It has increased resolution compared with the previous model (Simmonds, Lawson and Mayall, 1995) particularly in the English Channel, The Atlantic Ocean and the Barents and Kara Seas. In particular the Atlantic Ocean is modelled using three vertical water compartments with exchanges between them. However, as noted above, in order to enable the model in PC-CREAM 08 to run more efficiently some compartments included in the full MARINA II model (Simmonds et al, 2002) in the Atlantic, Arctic, Barents and Kara seas have been consolidated into single compartments.

Tables 4.7 and 4.8 give the parameter values required for implementation of the model. Table 4.7 describes the dimensions of the compartments and the key modelling parameters required for each compartment. Water compartments which include more than one vertical water layer (eg, Atlantic North NE, Kattegat, Belt Sea and Baltic Seas) only include interaction with sediments via the bottom water layer. In general, radionuclide transport is modelled by an advective flux representing the action of currents. However, it should be noted that for deep compartments, especially in the Atlantic Ocean region, turbulent diffusion may also make a significant contribution to radionuclide transport (in particular for vertical transport). Accordingly, turbulent diffusion has been modelled, where necessary, using a diffusion flux, complementary to the advective flux. Table 4.8 gives the advective and diffusive fluxes between the compartments. These are summed to provide a total flux for implementation in the model.

(b) Sediments

As for freshwater, the adsorption of activity by sediments can result in significant depletion of activity from the water phase. Such depletion is due to both the partitioning of the activity between the liquid phase and the solid phase (suspended sediments) and the removal of activity from the water column to bottom sediments. The amount of activity which is in solution is the quantity which needs to be calculated because the concentration factors for sediments and marine organisms are defined with respect to this soluble fraction.

The sediment concentration factor, or distribution coefficient, K_d, is defined as the ratio of the amount of radionuclide per unit weight of dry sediment to the amount per unit volume of water (Bq g^{-1} per Bq cm^{-3} or Bq t^{-1} per Bq m^{-3}). Table 4.9 gives values of the sediment concentration factor, K_d, which have been adopted for the common nuclides used in PC-CREAM 08. For a given element, it is common to find variations in K_d of two orders of magnitude according to the physical and chemical properties of the radionuclides and the sediments. The values given in Table 4.9 are those recommended for coastal waters and deep oceans, unless site-specific information is available.

Depletion by sediments is greater for those nuclides with the higher values of the sediment concentration factor; it will also be greatest in those sea areas with high suspended sediment loads and/or high rates of sediment deposition, particularly if such sediments are of small grain size, such as mud and silts, which have a high surface area to volume ratio. At any given time the activity in the water column is partitioned between the water phase and the suspended sediment material. The activity in the water column which is in solution is calculated in the same way as for freshwater in equation 4.5

The sedimentation model implemented in PC-CREAM 08 and described below accounts for remobilisation of activity from the top sediment layer into the water column and the transfer of activity to deep sediment. The remobilisation model for the top sediment layer was adapted from the COLDOS model (MacKenzie and Nicholson, 1987) and (Mitchell et al, 1999). This approach was used to calculate the transfers between top sediment and water (λ_1 and λ_2 in Diagram 4.3). The transfers between lower sediment layers were modelled using the same approach as in Simmonds, Lawson and Mayall (1995).

Two types of marine environment are defined for the purpose of modelling the transfer of radionuclides to sediments, deep and coastal waters, due to the differences found in the transfer processes in the deep ocean compared to those in the coastal environment. The transfer to sediment can be influenced by many factors, such as sediment exchangeability and abundance of biota. Therefore, for coastal compartments, ie, depths less than 200 m, parameter values for sediment reworking and porewater turnover are listed in Table 4.10 which are different to those for deep waters. The values for the radionuclide dependant sediment distribution factors, K_d used in the equations to calculate transfers between sediment and water for both coastal (less than 200 m) and deep oceans are given in Table 4.9.

It should be noted that activity concentrations are calculated for dry bed sediments in PC-CREAM 08 but include the portion of activity associated with the porewater. If bed sediment concentrations are to be compared with measurements then users should consider removing the activity associated with the porewater by multiplying the dry bed sediment concentration by (1-1/R). The parameter R depends on the porosity, density and sediment partition coefficient of the sediment (Simmonds et al, 1995).

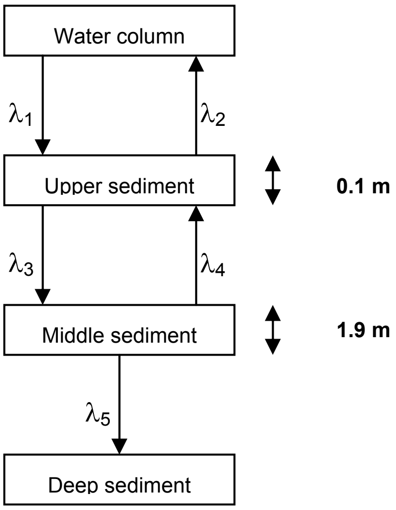


Diagram 4.3 Generic structure for the water–sediment compartment model

For each water compartment directly in contact with the seabed, the compartment structure shown in Diagram 4.3 is used to represent the transfer of radionuclides through sedimentation. The relation between the activity concentration in water and in suspended sediment is the same as given in equation 4.4. The activity in dry bed sediment can be estimated from:

$$C_{mbsed} = \frac{A_{mbsed}}{WV\ WD\ Lt\ \rho\left(1-\varepsilon\right)Conv} \tag{4.25}$$

Where:

C_{mbsed} = Activity concentration in bed sediment (Bq t^{-1})

A_{mbsed} = Activity in bed sediment compartment (Bq)

WV = Volume of water layer (m^3)

WD = the depth of the water layer (m)

L_t = the thickness of the upper sediment layer (m) (see Table 4.11)

ρ = the density of the sediments t m^{-3} (see Table 4.11)

ε = the sediment porosity (unit less) (see Table 4.7)

Conv = the conversion of t to kg = 1000

The following equations are used to determine the transfer rates between the water and sediment compartments presented in Diagram 4.3.

Transfer between water and top sediment (λ_1)

This transfer has four terms representing particle scavenging, molecular diffusion, porewater mixing and particle mixing:

$$\lambda_1 = \frac{1}{WD(1+k_d\alpha)}\left(SRk_d + \frac{D}{L_t} + R_T\varepsilon L_t + R_w\rho k_d(1-\varepsilon)\right) \qquad (4.26)$$

Where: K_d is the sediment distribution coefficient ($m^3\ t^{-1}$)

SR = the sedimentation rate ($t\ m^{-2}\ y^{-1}$)

α = the suspended sediment load ($t\ m^{-3}$)

D = the sediment diffusion coefficient ($m^2\ y^{-1}$) (see Table 4.7)

WD = the depth of the water layer (m)

L_t = the thickness of the upper sediment layer (m) (see Table 4.11)

ε = the sediment porosity (see Table 4.7)

R_T = the porewater turn-over rate (1 y^{-1} for shallow seas (up to 200 m deep) (Mitchell et al, 1999); 0.1 y^{-1} per year for deep ocean (Mackenzie and Nicholson, 1987))

R_W = the sediment reworking rate (5 10^{-3} m y^{-1} for shallow seas (up to 200 m deep) (Mitchell et al, 1999); 5 10^{-4} for deep oceans) (Mackenzie and Nicholson, 1987)

ρ = the density of the sediments ($t\ m^{-3}$) (see Table 4.11).

The diffusion term is fully represented as:

$$\frac{D}{b.\min(b,c)} \qquad (4.27)$$

where b and c are the depths of sediment of the relevant two layers between which diffusion occurs. However, for all the equations given below, the diffusion term of the equation has been evaluated using the sediment layer depths shown in Diagram 4.3. Should these depths change, then the equations will require revision accordingly.

Transfer between top sediment and water (λ_2)

This transfer includes three processes: molecular diffusion, porewater mixing and particle mixing:

$$\lambda_2 = \frac{DF_S}{L_t^2 \varepsilon} + R_T F_S + \frac{R_w(1-F_S)}{L_t} \tag{4.28}$$

where F_s is the inverse of the reciprocal of the retardation coefficient ($1/R$) (Simmonds et al, 1995):

$$F_S = \frac{1}{1 + \frac{k_d \rho (1-\varepsilon)}{\varepsilon}} \tag{4.29}$$

Transfer between top sediment and middle sediments (λ_3)

This transfer has two terms: one for sedimentation and the other for diffusion

$$\lambda_3 = \frac{(1-F_S)SR}{\rho L_t (1-\varepsilon)} + \frac{DF_S}{L_t^2} \tag{4.30}$$

Transfer between middle and top sediments (λ_4)

This transfer only includes diffusion:

$$\lambda_4 = \frac{DF_S}{L_m L_t} \tag{4.31}$$

where L_m is the thickness of the middle sediment layer (1.9 m).

Transfer between middle and deep sediments (λ_5)

This transfer only includes sedimentation

$$\lambda_5 = \frac{(1-F_S)SR}{\rho L_m (1-\varepsilon)} \tag{4.32}$$

4.4.2.2 Exposure pathways

The purpose of the estuary and marine models is to calculate radionuclide concentrations in the filtrate fraction of the water and on suspended sediment. There are potentially a large number of pathways by which man may be irradiated as a result of these radionuclide concentrations. Only a habit survey can determine the relative importance of the different pathways in any specific case, but those pathways which are included in generic assessments are discussed here. The pathways may be conveniently divided into those which arise as a result of seaspray and those which arise more directly from the calculated radionuclide concentrations. A full list of the exposure pathways considered in this methodology is given in Table 4.1.

Exposure pathways arising from seaspray

The transfer of radionuclides to the land in seaspray is well documented (Eakins et al, 1982; Howorth and Eggleton, 1988). A default transfer rate of 1 m^3 of seaspray per year per metre of coastline is recommended. This transfer is only taken into account for the local marine compartment because radionuclide concentrations in other marine compartments are considerably lower. The radionuclide concentration in seaspray is generally taken to be that in the water column of the local marine compartment which includes activity on suspended sediments as well as activity dissolved in the water. In the case of certain actinides the radionuclide concentration in seaspray may be enhanced relative to that in seawater (Eakins et al, 1982). The seaspray only travels a limited distance inland. Empirical formulae for the deposition rate of the seaspray as a function of distance inland from the shore have been developed (Howorth and Eggleton, 1988) or it may be simply assumed that all of the radionuclides in the seaspray are deposited uniformly over a certain distance inland, say 500 m. The choice of model depends upon the level of detail required and the relative importance of this pathway.

Exposure of man may arise either directly by the inhalation of the seaspray, or from a number of terrestrial pathways following deposition of the seaspray on land. The dose to an individual from inhalation of seaspray depends upon the activity concentration in the seaspray, the inhalation rate of the individual, the inhalation dose per unit intake and the amount of time the person is exposed to the spray. The simplest approach is to assume that the person spends this time on a shoreline because this is where the maximum radionuclide concentrations occur in seaspray. Dose per unit intake values for 1 μm AMAD (activity median aerodynamic diameter) particles are used unless measurements of the aerosol size indicate a different value. The collective dose arising from inhalation of seaspray may be similarly calculated using collective shoreline occupancy expressed in man hours per year, for example.

The deposition on land of radionuclides in seaspray can be treated in an analogous manner to the deposition of other airborne radionuclides, ie, the same deposit on the land is assumed to give rise to the same concentrations in soil and crops as if the radionuclides were from an airborne discharge. The terrestrial exposure pathways considered are described in detail in the chapter on atmospheric discharges.

Currently, PC-CREAM 08 only includes the direct inhalation of radionuclides in seaspray in the calculation of individual doses. It does not include the estimation of doses due to deposition onto land or the collective dose due to seaspray.

The formula to calculate inhalation from seaspray in PC-CREAM 08 is based on the empirical approach described in Howorth and Eggleton (1988) and used for the Cumbria study (Wilkins et al, 1994):

$$D_{seaspary} = C_{air}\, R_{inh}\, Occ\, DC_{inh} \tag{4.33}$$

Where $D_{seaspray}$ is the dose to an individual from inhalation of seaspray (Sv y^{-1}), C_{air} is the activity concentration of seaspray (Bq m^{-3}, see below), R_{inh} is the inhalation rate for an individual (adult = 7300 m^3 y^{-1}), *Occ* is the occupancy of an individual in a coastal area (adult = 2000 h y^{-1}), DC_{inh} is the dose coefficient for inhalation to members of the public for 1 µm particles (Sv Bq^{-1}).

C_{air} is derived from an empirical formula based on measurements of soil cores for plutonium, americium and caesium and deposition velocities for plutonium and caesium along the Cumbrian coastlline. The values of parameters given below are those used in PC-CREAM 08.

$$C_{air} = \frac{C_{soil}}{TDV} \tag{4.34}$$

where C_{soil} is the measured cumulative deposition in the soil, (Bq m^{-2}) and TDV is an element dependent total deposition rate measured for plutonium and caesium at 300 m from shoreline (m y^{-1}). Values of TDV are given in Table 4.10. For PC-CREAM 08, the value for plutonium is adopted for all actinides, while the value for caesium is adopted for all other elements.

The cumulative deposition in the soil, C_{soil} is related to the (calculated) cumulative seaborne activity, C_{sea} (Bq m^{-3}), through the equation:

$$C_{soil} = C_{sea} A10^{-\alpha d} (1 + B10^{-\beta d}) \tag{4.35}$$

In PC-CREAM 08 C_{sea} was assumed to represent the unfiltered water concentration per unit discharge (Bq m^{-3} per Bq y^{-1}) in marine water adjacent to the coast; d is the distance inland from the mean high water-mark (km). A (m y^{-1}), B (dimensionless), α (km^{-1}) and β (km^{-1}) are element dependent empirical constants determined for plutonium, americium and caesium. In PC CREAM values for plutonium are assumed to apply to all actinides except americium and curium; values for americium are assumed to apply to americium and curium and ruthenium while values for caesium are assumed to apply to caesium and all other elements excluding those above.

Exposure pathways from ingestion of seafood

Both collective and Individual doses arising from seaspray are generally low in comparison to those arising from the more direct pathways. Of the more direct pathways, the most important in terms of dose is usually the ingestion of seafood (Jones et al, 2002). Radionuclide concentrations in the edible parts of seafood are obtained from the concentrations in the filtrate fraction of seawater using a concentration or bio-accumulation factor. The concentration factor relates the activity per unit mass of edible seafood to the dissolved activity per unit mass of seawater. Its use assumes there is equilibrium between the radionuclide concentrations in the seafood and those in the surrounding water. Generic values of the concentration factors for fish, crustaceans, molluscs and seaweed for the radionuclides considered in this methodology are given in Table 4.12. Wherever possible site-specific data (eg, Pentreath et al, 1988) should be used instead of these generic values.

Doses to individuals from the consumption of seafood are obtained from the calculated radionuclide concentrations in the seafood together with information about which marine areas have supplied the individual's intake and in what proportion. For example, a pessimistic assumption would be to assume that the individual obtained all his intake of seafood from the local marine compartment, where radionuclide concentrations are highest. This assumption must be balanced against the probability of it being true. In PC-CREAM 08 the default assumption is that individual~~s~~ obtain 90% of their fish from the regional marine compartment closest to the discharge point, ie, the compartment encompassing the local marine compartment, and 10% from the local compartment. Also the individual's intake of crustaceans, molluscs and seaweed is assumed to be all from the local compartment. However, it is possible for users to change these factors in estimating both individual and collective doses.

The individual dose for ingestion of seafood is then given by:

$$E_i = \sum_{f=1}^{n} H_{i,f} \tag{4.36}$$

where:

E_i	is the individual effective dose for region i (Sv y^{-1}).
$H_{i,f}$	is the individual dose from ingestion of food type, f, for region i.
n	is the number of foodstuffs considered.

$$H_{i,f} = \left(\left(Cfood_{local}\ I_{i,f}\ f_{local}\right) + \left(Cfood_{regional}\ I_{i,f}\ f_{regional}\right)\right)H_{ing} \tag{4.37}$$

where:

$Cfood_{local}$	is the activity concentration in food in the local compartment (Bq t^{-1})
$Cfood_{regional}$	is the activity concentration in food in the regional compartment (Bq t^{-1})
$I_{i,f}$	is the intake rate of food for region i (t y^{-1}).
f_{local}	is the fraction of food caught for consumption in the local compartment
$f_{regional}$	is the fraction of food caught for consumption in the regional compartment
H_{ing}	is the effective dose per unit intake by ingestion (Sv Bq^{-1}).

Where:

$$Cfood_{local} = C_{fmw\_local}\ CF \tag{4.38}$$

C_{fmw_local}	is the activity concentration of the radionuclide in filtered seawater in the local compartment (Bq m^{-3})
CF	is the nuclide dependent concentration factor for seafood (See Table 4.12) (Bq t^{-1} per Bq m^{-3})

$$Cfood_{regional} = C_{fmw\_regional}\ CF \tag{4.39}$$

C_{fmw}	is the activity concentration of the radionuclide in filtered seawater in the regional compartment (Bq m^{-3})
CF	is the nuclide dependent concentration factor for seafood (See Table 4.12) (Bq t^{-1} per Bq m^{-3})

Collective doses from the ingestion of seafood are calculated using the recorded catches of fish, crustaceans, molluscs, and seaweed as published by the International Council for the Exploration of the Sea (ICES) in annual reports (ICES, 2001) as described in the MARINA II Working Group C report (Hagel, 2002). Most of the catch data presented in Tables 4.13 to 4.15 are taken from the MARINA II Working Group C report (Hagel, 2002). These values take into account imports/exports, non food use and edible fractions of foods (see Table 4.16). Catch data for the Baltic were obtained from the EC MARINA BALT study (Nielsen, 2000) and catch data for the Mediterranean (Appendix 4.3) were obtained from the EC MARINA MED study (Cigna et al, 1994). The edible fractions for seafood for these two regions were not included and so the default values given in Table 4.16 were used. The fish catch data for the local compartment are obtained by multiplying the catch data for the regional compartment by the ratio of the local to regional compartment volumes. For crustaceans, molluscs and seaweed the regional catch and harvest is multiplied by the ratio of the local to regional coastline lengths to obtain data for the local compartment.

The collective dose is then given by:

$$H_c(t) = \sum_{f=1}^{M} IC_f(t)\, H_{ing} \tag{4.40}$$

where:

H_c (t)	is the collective effective dose integrated to time t (man Sv).
H_{ing}	is the effective dose per unit intake by ingestion (Sv Bq^{-1}).
M	is the number of foods.

The collective intake for ingestion of foodstuff f is given by:

$$IC_f(t) = \sum_{i=1}^{n} Cfood_{i,f}(t)\, P_i^{f} \tag{4.41}$$

where:

$IC_f(t)$	is the time integral of the collective intake to time t (Bq).
$Cfood_{i,f}(t)$	is the time integral to time t of nuclide concentration in food f, in region i (Bq y t^{-1}).
P_i^f	is the annual catch of food f in region i (t y^{-1}), including imports/exports and edible fraction.
n	is the number of regions.

$$Cfood_{i,f}(t) = C^{i}_{ifmw}(t)\,CF_f \tag{4.42}$$

where:

$Cfood_{i,f}$ is the time integral to time t of concentration in food f in water region i (Bq y t^{-1}).

C^{i}_{ifmw} is the time integral to time t of concentration of the radionuclide in filtered seawater water in region i (Bq y m^{-3}).

CF_f is the nuclide dependent concentration factor for food f (Bq t^{-1} per Bq m^{-3})

Exposure pathways from sediments

Radionuclide concentrations on beach material may be taken to be the same as those in the top layer of bed sediment in the adjacent marine compartment. However, because most of the bed sediment is comprised of fine muddy particulates, rather than the larger sand grains and gravel which form the beaches used by the majority of people, a reduction factor of 10 may be applied to obtain the radionuclide concentrations on the most-frequented beaches. This reduction factor has been obtained from a comparison of predicted radionuclide concentrations in bed sediment in the Irish Sea and measurements on material from intertidal areas (Charles, Jones and Cooper, 1990). In PC-CREAM 08 the reduction factor of 10 is included when estimating collective doses from external irradiation from beach occupancy but not when estimating individual doses.

People may also inadvertently inhale and ingest beach material. This pathway is not currently included in PC-CREAM 08 as doses are generally insignificant compared to other pathways (Jones et al, 2002). Further details about estimating exposures from these pathways are given in Simmonds, Lawson and Mayall (1995).

External exposure from beach material may be an important pathway for certain radionuclides. The simplest approach to calculating these external doses is to assume that the beach is a uniformly contaminated semi-infinite medium. The calculated dose rate from such a medium (Hunt, 1984) should be combined with an occupancy factor or collective occupancy factor. In PC-CREAM 08 the external doses from freshwater and marine sediments are calculated for both beta and gamma emissions. The gamma doses are calculated using the approach given by Hunt (1984). The following formulae are used to calculate doses in PC-CREAM 08:

For gamma:

$$E = C_{mbsed}\ DTW\ GAMM\ DF\ 1.7\ Conv\ 0.87\ Occ \tag{4.43}$$

Where:

E = Effective dose in μSv y^{-1}

C_{mbsed}= Activity concentration in marine sediments (dry) (Bq kg^{-1})

DTW = dry to wet activity concentration conversion = 0.65/1.7

GAMM= Gamma energy (MeV)

DF= Dose rate in sediment (wet) 0.1584 (μGy h^{-1} per Bq cm^{-3} per MeV) (Hunt, 1984).

1.7 = wet density in g cm^{-3} to convert DF to (μGy h^{-1} per Bq g^{-1}).

Conv = Conversion from Bq kg^{-1} to Bq g^{-1} ($1\ 10^{-3}$)

0.87 = Sv per Gy

Occ = Occupancy time on sediment (h y^{-1}) (See Table 4.17).

This becomes:

$$E = C_{mbsed}\ 0.65\ GAMM\ DF\ Conv\ 0.87\ Occ \qquad (4.44)$$

Where:

DF= Dose rate in sediment (wet) 0.1584 (μGy h^{-1} per Bq g^{-1} per MeV) (Hunt, 1984).

For beta:

$$E = C_{mbsed}\ Dens\ t\ DF\ wt\ Occ\ Conv$$

Where:

E = Effective skin dose in µSv y^{-1} (4.45)

C_{mbsed}= Activity concentration in sediments (Bq kg^{-1}) dry

Dens = 650 kg m^{-3}.

t = Thickness of deposit 0.01m

DF = beta skin dose factor at 1m (Sv y^{-1} per Bq m^{-2}) (Holford, 1989)

Wt= skin weighting factor (0.01)

Occ = Occupancy time on sediment (h y^{-1})/8760 h y^{-1} (See 4.17)

Conv = Conversion from Sv y^{-1} to (μSv y^{-1}) ($1\ 10^{6}$)

In certain cases external exposure from contaminated fishing gear may be an important pathway for a few individuals. A pessimistic estimate of the dose rate to a person handling the fishing gear may be obtained using the method described by Hunt (1984). The time spent in close proximity to the fishing gear is likely to be short.

In PC-CREAM 08 collective doses from beach exposure is calculated for gamma emissions using an estimated value for the collective beach occupancy.

The collective dose is given by:

$$H_c{}^j(t) = \sum_{i=1}^{n} C^{i,j}{}_{imbsed}(t) BF \; 0.65 \; GAMM \; DF \; Conv \; 0.87 \; Occrate_i \; Cl_i \qquad (4.46)$$

where

$H_c^j(t)$	is the collective effective dose integrated to time t from exposure source j (man Sv).
$C^{i,j}{}_{imbsed}$	is the time integral to time t of concentration of the radionuclide, for source j, in sediment in region i (Bq y kg^{-1}).
BF	Beach factor of 0.1, is a reduction factor to allow for the reduced concentration in sandy sediment, where most beach exposure occurs
$Occrate_i$	is the average occupancy rate in region i, (man h y^{-1} m^{-1}) (see Table 4.18).
CL_i	is the coastline length in region i, (m), (see Table 4.19).

An illustrative example application of the marine models is shown in Figures 4.7 to 4.10. These figures show results for a hypothetical discharge of 1 TBq over 1 year to the local marine compartments around Vandellos in Spain, and Sellafield in north-west England. Two radionuclides were considered: caesium-137, which has a low affinity for sediments, and plutonium-239, which has a high affinity for sediments. Figures 4.7 to 4.10 show the temporal variation of activity concentrations in seawater and the top of the seabed. Figures 4.11 and 4.12 show the resulting collective dose to the EU population in the 50th year of continuous discharge for caesium-137 and plutonium-239, respectively.

4.4.3 Exchange between the oceans and the atmosphere

Three radionuclides, tritium, carbon-14 and iodine-129, are sufficiently long-lived and mobile that the exchange of these radionuclides between the oceans and the atmosphere should be taken into account when considering collective doses over long timescales. This exchange, and the further exposure pathways arising from it, is discussed in Chapter 5 which covers global circulation models.

4.5 Model limitations

The models contained in the methodology, CREAM, and the computer system PC-CREAM 08 for the dispersion of radionuclides in rivers, estuaries and the marine environment are all intended for use in assessing the consequences of continuous releases. They are generic and intended for use throughout the EC. Much use is made of annual averages for parameters such as water flow. This is considered adequate for assessing the radiological consequences of routine releases where radiation doses on an annual basis or integrated over longer timescales are required. Similarly the use of equilibrium concentration factors to relate concentrations of radionuclides in water to those in sediments or aquatic foods is considered adequate. However, such models may not be appropriate if short term releases are considered or if information is required on the variation in concentrations over short times (less than a year). In these cases more complex, site specific models would be more appropriate.

Wherever possible the models have been validated by comparing model predictions with sets of environmental data not used in their development. For example, a simplified version of the river model described in section 4.2 was used in an assessment of releases of radionuclides to the river Molse Nete in Belgium (Lawson et al, 1991). As part of this study, the predicted concentrations of radionuclides in the bed sediment and predicted external gamma dose rates were compared with measurements carried out over a number of years. Agreement between the predictions and measurements was generally within a factor of five (Lawson et al, 1991). Also PC-CREAM has been used in a study for the Environment Agency (Hilton et al, 2002), in which doses were calculated for multiple discharges into the river Thames. A comparison was made between measured and predicted concentrations in fish, freshwater plants and sediment. This showed agreement between predictions and measurements within a factor of four for concentrations in fish and freshwater plants for tritium, caesium-137 and strontium-90.

The marine model has also been tested extensively by comparing predictions with environmental measurements for a number of different cases. As part of the MARINA II project (Simmonds et al, 2002) the 72 compartment model was used to compare results for releases of caesium-137, technetium-99 and plutonium-239 for a number of different sea areas and media, against measured data, between 1990 and 2000. Most model predictions for filtered water were within a factor of two of the measured values. For sediments model predictions were within a factor of two to five of the measured values. One important development for this model was the improvement in modelling sediment remobilisation. This is illustrated in Figure 4.13 for the Irish Sea. The MARINA II model was also compared with the more complex CSERAM model developed by CEFAS (Aldridge, 1998). Comparisons with this model were made in two studies carried out by the HPA (Bexon et al, 2003; Jones et al, 2003). The models were generally found to be in good agreement with the observed environmental activity concentrations. Comparisons of the models are shown in Figures 4.14 and 4.15.

A comparison was made of the simplified marine model used in PC-CREAM 08 and the full model used in MARINA II (Simmonds et al, 2002). The differences in activity concentrations between the two models are not significant enough to affect dose calculations to any great extent. A release into the Portuguese Continental Shelf (PCS) shows greatest variation due to the changes to the Atlantic compartments. However, this scenario models a direct discharge into the PCS and the use of a local compartment would result in smaller differences. Individual dose calculations in PC-CREAM 08 will be estimated using the local and regional compartments closest to the release point and these show less than 10% variation when compared to the full model. For collective dose calculations, the differences are seen at greatest distances from the release point and therefore, due to dilution, water and sediment activity concentrations are several orders of magnitude lower than nearby compartments. It is therefore unlikely that the collective dose calculation will be greatly affected by these model simplifications.

4.6 References

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4.7 Tables

Table 4.1 Relevant exposure pathways for discharges to aquatic systems

Exposure pathway	Freshwater	Marine
Ingestion	Fish	Fish
	Treated/untreated drinking water	Crustacea
		Molluscs
		Seaweed
	Terrestrial foods from land irrigated with river water**	Terrestrial foods from land subject to seaspray deposition*
	Terrestrial foods from land treated with river sediments*	Terrestrial foods from land treated with marine sediments*
	Terrestrial foods from land treated with river plants*	Terrestrial foods from land treated with marine plants*
	Animal products where animals' drinking water has been taken from a river or lake*.	
	River water whilst swimming (inadvertent)*	Sea water whilst swimming (inadvertent)*
	River bank sediments (inadvertent)*	Marine sediments (inadvertent)*
Inhalation	Resuspended river sediments*	Resuspended marine sediments*
		Seaspray
External irradiation	River bank sediments	Marine sediments
	Fishing gear*	Fishing gear
	River / lake water while swimming*	Sea water while swimming*
	River / lake water while boating*	

* Pathways not considered in PC-CREAM 08

** Can be calculated using the FARMLAND model

Table 4.2 Radionuclides considered for assessing the radiological impact of aquatic releases (Common radionuclides in PC-CREAM 08) (ICRP, 1993)

Radionuclide	Half Life (y)	Radionuclide	Half Life (y)
^{3}H	$1.23\ 10^{1}$	^{131}I	$2.20\ 10^{-2}$
^{14}C	$5.73\ 10^{3}$	^{134}Cs	$2.06\ 10^{0}$
^{32}P	$3.92\ 10^{-2}$	^{137}Cs	$3.00\ 10^{1}$
^{33}P	$6.96\ 10^{-2}$	^{140}Ba	$3.49\ 10^{-2}$
^{35}S	$2.40\ 10^{-1}$	^{140}La	$4.60\ 10^{-3}$
^{51}Cr	$7.59\ 10^{-2}$	^{141}Ce	$8.90\ 10^{-2}$
^{54}Mn	$8.56\ 10^{-1}$	^{144}Ce	$7.79\ 10^{-1}$
^{55}Fe	$2.70\ 10^{0}$	Pm-147	$2.62\ 10^{0}$
^{59}Fe	$1.22\ 10^{-1}$	Eu-152	$1.33\ 10^{1}$
^{57}Co	$7.42\ 10^{-1}$	Eu-154	$8.80\ 10^{0}$
^{58}Co	$1.94\ 10^{-1}$	Eu-155	$4.96\ 10^{0}$
^{60}Co	$5.27\ 10^{0}$	^{210}Pb	$2.23\ 10^{1}$
^{63}Ni	$9.60\ 10^{1}$	^{210}Po	$3.78\ 10^{-1}$
^{65}Zn	$6.68\ 10^{-1}$	^{226}Ra	$1.60\ 10^{3}$
^{75}Se	$3.29\ 10^{-1}$	$^{230\mathrm{Th}}$	$7.70\ 10^{4}$
^{89}Sr	$1.38\ 10^{-1}$	$^{234\mathrm{Th}}$	$6.60\ 10^{-2}$
^{90}Sr	$2.91\ 10^{1}$	^{234}U	$2.44\ 10^{5}$
^{95}Zr	$1.75\ 10^{-1}$	^{235}U	$7.04\ 10^{8}$
^{95}Nb	$9.63\ 10^{-2}$	^{238}U	$4.47\ 10^{9}$
^{99}Tc	$2.13\ 10^{5}$	^{237}Np	$2.14\ 10^{6}$
^{103}Ru	$1.08\ 10^{-1}$	^{238}Pu	$8.78\ 10^{1}$
^{106}Ru	$1.01\ 10^{0}$	^{239}Pu	$2.41\ 10^{4}$
^{122}Sb	$7.40\ 10^{-3}$	^{240}Pu	$6.54\ 10^{3}$
^{124}Sb	$1.65\ 10^{-1}$	^{241}Pu	$1.44\ 10^{1}$
^{125}Sb	$2.77\ 10^{0}$	^{241}Am	$4.32\ 10^{2}$
^{123m}Te	$3.28\ 10^{-1}$	^{242}Cm	$4.46\ 10^{-1}$
^{125m}Te	$1.59\ 10^{-1}$	^{243}Cm	$2.85\ 10^{1}$
^{129}I	$1.57\ 10^{7}$	^{244}Cm	$1.81\ 10^{1}$

Table 4.3 – Freshwater sediment distribution factors (K_d), values of k′ and freshwater fish concentration factors for common radionuclides in PC-CREAM 08

Element	K_d ($m^3\ t^{-1}$)	Reference	k′ (m^{-1})	Fish ($m^3\ t^{-1}$)	Reference
H	3 10^{-2}	Kane (1984)	0	1 10^{0}	NCRP (1996)
C	2 10^{3}	Booth (1976)	2 10^{-6}	5 10^{4} [a]	IAEA (1994)
P	5 10^{1}	IAEA (2001)	2 10^{-6}	5 10^{4} [b]	IAEA (2001)
S	2 10^{2}	Booth (1976)	2 10^{-6}	8 10^{2}	IAEA (2001)
Cr	2 10^{4}	Booth (1976)	1 10^{-5}	2 10^{2}	IAEA (2001)
Mn	5 10^{4}	Zeevaert (1987)	1 10^{-5}	4 10^{2}	IAEA (2001)
Fe	1 10^{4}	Booth (1976)	2 10^{-6}	2 10^{2}	IAEA (2001)
Co	2 10^{4}	Zeevaert (1987)	1 10^{-5}	3 10^{2}	IAEA (2001)
Ni	1 10^{4}	Coughtrey et al (1985)	2 10^{-6}	1 10^{2}	IAEA (2001)
Zn	1 10^{3}	Booth (1976)	2 10^{-6}	1 10^{3}	IAEA (2001)
Se	4 10^{3}	Coughtrey et al (1985)	2 10^{-6}	2 10^{2}	IAEA (2001)
Sr	2 10^{3}	Booth (1976)	2 10^{-6}	6 10^{1} [c]	IAEA (1994)
Zr	6 10^{4}	Booth (1976)	1 10^{-5}	3 10^{2}	IAEA (2001)
Nb	1 10^{2}	Booth (1976)	2 10^{-6}	3 10^{2}	IAEA (2001)
Tc	1 10^{1}	Kane (1984)	2 10^{-6}	2 10^{1}	IAEA (2001)
Ru	7 10^{3}	Zeevaert (1987)	2 10^{-6}	1 10^{1}	IAEA (2001)
Sb	5 10^{2}	Zeevaert (1987)	2 10^{-6}	1 10^{2}	IAEA (2001)
Te	3 10^{1}	Booth (1976)	2 10^{-6}	4 10^{2}	IAEA (2001)
I	3 10^{2}	Coughtrey et al (1985)	2 10^{-6}	4 10^{1}	IAEA (2001)
Cs	1 10^{4}	Brach-Papa et al, 2005	2 10^{-6}	2 10^{3}	IAEA (2001); NCRP (1996)
Ba	2 10^{3}	Subgroup IIa element(Alkaline earth) similar to strontium	2 10^{-6}	4 10^{0}	IAEA (2001)
La	6 10^{4}	Booth (1976)	1 10^{-5}	3 10^{1}	NCRP (1996)
Ce	1 10^{4}	IAEA (2001)	2 10^{-6}	3 10^{1}	IAEA (2001)
Pm	5 10^{3}	IAEA (2001)	2 10^{-6}	3 10^{1}	IAEA (2001)
Eu	1 10^{4}	Similar to cerium (Lanthanides)	2 10^{-6}	5 10^{1}	IAEA (2001)
Pb	1 10^{4}	Kane (1984)	2 10^{-6}	3 10^{2}	IAEA (2001)
Po	4 10^{4}	Shahul Hameeed et al (1997)	1 10^{-5}	5 10^{1}	IAEA (2001)
Ra	5 10^{2}	IAEA (2001)	2 10^{-6}	5 10^{1}	IAEA (2001)
Th	5 10^{6}	Kane (1984)	1 10^{-5}	1 10^{2}	IAEA (2001)
U	5 10^{1}	IAEA (2001)	2 10^{-6}	1 10^{1}	IAEA (2001)
Np	5 10^{2}	Kane (1984)	2 10^{-6}	3 10^{1}	IAEA (2001)
Pu	1 10^{5}	IAEA (2001)	1 10^{-5}	3 10^{1}	IAEA (2001)
Am	4 10^{5}	Coughtrey et al (1985)	1 10^{-5}	3 10^{1}	IAEA (2001)
Cm	1 10^{5}	Coughtrey et al (1984), Matsunaga et al (1998)	1 10^{-5}	3 10^{1}	IAEA (2001)

Notes:

a This is PC-CREAM default. Recommended UK value of 5 10^3 given in Thompson et al (1972) and Smith (2006).

b This is PC-CREAM default. For UK lowland phosphate-receiving rivers and other rivers with a high phosphorus content a value of 5 10^3 is considered more appropriate (Hilton et al, 2002).

c This is PC-CREAM default. For areas sedimentary bedrock with hard water use low values (15). For regions with igneous rocks, sandy or organic soils with soft water use high values (75). IAEA (2001)

d This is PC-CREAM default value. For areas sedimentary bedrock with hard water use low values (2 10^3) . For regions with igneous rocks, sandy or organic soils with soft water use high values (1 10^4) IAEA (2001).

Table 4.4 – Removal efficiencies, R of radionuclides for water treatment plants (% removal) (Brown et al, 2008a; Brown et al, 2008b)

Element	Flocculation / coagulation / clarification	Sand Filtration (Rapid & Slow)
H[a]	0	0
C[a]	0	0
P[a]	0	0
S[b]	40	10
Cr[c]	40	10
Mn[c]	40	10
Fe[c]	40	10
Co	40	10
Ni[c]	40	10
Zn[c]	40	10
Se	40	10
Sr	10	10
Zr	70	10
Nb	70	10
Tc	40	40
Ru	40	10
Sb[d]	40	10
Te	40	10
I	10	10
Cs	10	10
Ba	10	40
La	70	40
Ce	70	70
Pm[e]	70	70
Eu[e]	70	70
Pb[f]	40	10
Po[f]	40	10
Ra	10	40
Th[g]	70	10
U	70	0
Np[g]	70	10
Pu	70	10
Am	70	10
Cm[g]	70	10

Notes:

a - Hammond, 2009; b - Assumed to be largely removed as sulphate or liberated as SO_2 etc.

c - Assume same as Co, transition metals; d - Similar to Te; e - lanthanides same as Ce

f - Similar to Sb; g - Similar to Am

Table 4.5 – Sediment desorption parameters for Estuary modelling

Elements	Desorption fraction
H, C, Na, Ca, Cr, Mn, Fe, Co, Ni, Zn, Se, Sr, Y, Zr, Nb, Tc, Ag, Sn, Sb, Te, I, Ba, Cs, Ce, Pr, Pm, Eu, Pb, Po, Ra, Ac, Th, Pa, U, Np, Pu, Am, Cm	Zero
P, S, Cl, Ca, Ru	0.3

Table 4.6 – Site and local marine compartment parameters

Local marine compartment No	Site Name	Type of site [a]	Discharging state	River discharged to, if inland site [b]	Regional marine compartment	Volume exchange rate with regional compartment ($m^3\ y^{-1}$)	Volume (m^3)	Depth (m)	α ($t\ m^{-3}$)	SR ($t\ m^{-2}\ y^{-1}$)
74	Almaraz	(i)	Spain	Tajo (Modelled as Loire 3)	26	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
75	Barsebaeck	(c)	Sweden	-	46	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$7.50\ 10^{-4}$
76	Belleville	(i)	France	Loire (1)	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
77	Berkeley	(c)	UK	-	22	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
78	Beznau	(I)	Switzerland	Rhine (1)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
79	Biblis	(I)	Germany	Rhine (4)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
80	Blayais	(c)	France	-	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
81	Borssele	(c)	Netherlands	-	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
82	Bradwell	(c)	UK	-	38	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
83	Brokdorf	(c)	Germany	-	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
84	Brunsbuettel	(c)	Germany	-	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
85	Capenhurst	(c)	UK	-	20	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-4}$	$5.00\ 10^{-3}$
86	Cattenom	(i)	France	Moselle (Rhine 7)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
87	Chapelcross	(c)	UK	-	15	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-3}$
88	Chinon	(i)	France	Loire (3)	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
89	Chooz	(i)	France	Meuse	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
90	Dampierre	(i)	France	Loire (1)	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
91	Doel	(c)	Belgium	-	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.10\ 10^{-4}$
92	Dodewaard	(i)	Netherlands	Rhine (10)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
93	Dounreay	(c)	UK	-	12	$1.60\ 10^{11}$	$3.20\ 10^{9}$	40	$1.00\ 10^{-6}$	$1.00\ 10^{-4}$
94	Dungeness	(c)	UK	-	37	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$

Local marine compart-ment No	Site Name	Type of site [a]	Discharging state	River discharged to, if inland site [b]	Regional marine compart-ment	Volume exchange rate with regional compartment ($m^3\ y^{-1}$)	Volume (m^3)	Depth (m)	α ($t\ m^{-3}$)	SR ($t\ m^{-2}\ y^{-1}$)
95	Emsland	(i)	Germany	Ems – (Modelled as Rhine 10)	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
96	Fessenheim	(i)	France	Rhine (1)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
97	Flamanville	(c)	France	-	30	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
98	Golfech	(i)	France	Garonne - (Modelled as Loire 3)	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
99	Gosgen	(i)	Switzerland	Rhine (1)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
100	Grafenrheinfeld	(i)	Germany	Main – (Rhine 5)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
101	Gravelines	(c)	France	-	39	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$2.00\ 10^{-4}$
102	Grohnde	(i)	Germany	Weser- (Modelled as Rhine 7)	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
103	Hartlepool	(c)	UK	-	40	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
104	Harwell	(i)	UK	Thames (3)	38	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
105	Heysham	(c)	UK	-	20	$8.00\ 10^{9}$	$1.00\ 10^{8}$	10	$1.00\ 10^{-5}$	$4.90\ 10^{-3}$
106	Hinkley Point	(c)	UK	-	22	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
107	Hunterston	(c)	UK	-	11	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
108	Jose Cabrera	(i)	Spain	Tajo (Modelled as Loire 1)	26	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
109	Kahl	(i)	Germany	Main – (Rhine 5)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
110	Karlsruhe WAK	(i)	Germany	Rhine (2)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
111	Kruemmel	(i)	Germany	Elbe (Modelled as Rhine 10)	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$

Local marine compart-ment No	**Site Name**	**Type of site** [a]	**Discharging state**	**River discharged to, if inland site** [b]	**Regional marine compart-ment**	**Volume exchange rate with regional compartment ($m^3\ y^{-1}$)**	**Volume (m^3)**	**Depth (m)**	**α ($t\ m^{-3}$)**	**SR ($t\ m^{-2}\ y^{-1}$)**
112	Cap de la Hague	(c)	France	-	31	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.02\ 10^{-4}$
113	Leibstadt	(i)	Switzerland	Rhine (1)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
114	Muelheim-Kaerlich	(i)	Germany	Rhine (8)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
115	Muhlberg	(i)	Switzerland	Rhine (1)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
116	Neckarwestheim	(i)	Germany	Nekar (Rhine 3)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
117	Nogent	(i)	France	Seine (Modelled as Loire 2)	33	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
118	Obrigheim	(i)	Germany	Nekar (Rhine 3)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
119	Oldbury	(c)	UK	-	22	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
120	Paluel	(c)	France	-	34	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-5}$	$5.00\ 10^{-5}$
121	Penly	(c)	France	-	34	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
122	Philippsburg	(i)	Germany	Rhine (2)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
123	Rheinsberg	(i)	Germany	Elbe (Modelled as Rhine 10)	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
124	Ringhals	(c)	Sweden	-	44	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$7.50\ 10^{-4}$
125	Risø	(c)	Denmark	-	46	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$5.00\ 10^{-4}$
126	Sellafield	(c)	UK	-	18	$5.00\ 10^{11}$	$2.00\ 10^{9}$	20	$5.00\ 10^{-6}$	$1.00\ 10^{-2}$
127	Sizewell	(c)	UK	-	38	$1.10\ 10^{10}$	$3.00\ 10^{8}$	10	$8.00\ 10^{-5}$	$1.00\ 10^{-4}$
128	Springfields	(c)	UK	-	20	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$5.00\ 10^{-3}$
129	St Laurent des eaux	(i)	France	Loire 2	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
130	Stade	(c)	Germany	-	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$

Local marine compart-ment No	Site Name	Type of site [a]	Discharging state	River discharged to, if inland site [b]	Regional marine compart-ment	Volume exchange rate with regional compartment ($m^3\ y^{-1}$)	Volume (m^3)	Depth (m)	α ($t\ m^{-3}$)	SR ($t\ m^{-2}\ y^{-1}$)
131	Tihange	(i)	Belgium	Muese (Modelled as Rhine 8)	39	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
132	Torness	(c)	UK	-	40	$8.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
133	Trawsfynydd	(i)	UK	Lake Trawsfynydd	19	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
134	Trillo	(i)	Spain	Tajo (Modelled as Loire 1)	26	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$2.00\ 10^{-4}$
135	Unterweser	(c)	Germany	-	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
136	Winfrith	(c)	UK	-	32	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.02\ 10^{-4}$
137	Wuergassen	(i)	Germany	Weser (Modelled as Rhine 8)	41	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
138	Wylfa	(c)	UK	-	16	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-3}$
139	Devonport	(c)	UK	-	29	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-4}$	$2.00\ 10^{-4}$
140	Rosyth	(c)	UK	-	12	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
141	Faslane	(c)	UK	-	11	$1.00\ 10^{11}$	$5.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$1.00\ 10^{-4}$
142	Barrow	(c)	UK	-	20	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$5.00\ 10^{-3}$
143	Cardiff	(c)	UK	-	22	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
144	Aldermaston	(i)	UK	Thames (5)	38	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
145	Amersham	(i)	UK	Thames (7)	38	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$
146	Loviisa	(c)	Finland	-	54	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-4}$
147	Olkiluoto	(c)	Finland	-	49	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-4}$
148	Oskarshamn	(c)	Sweden	-	50	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-4}$
149	Forsmark	(c)	Sweden	-	49	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-4}$
150	Greifswald	(c)	Germany	-	51	$4.00\ 10^{10}$	$2.00\ 10^{9}$	20	$1.00\ 10^{-5}$	$5.00\ 10^{-4}$

Local marine compart-ment No	**Site Name**	**Type of site** [a]	**Discharging state**	**River discharged to, if inland site** [b]	**Regional marine compart-ment**	**Volume exchange rate with regional compartment ($m^3\ y^{-1}$)**	**Volume (m^3)**	**Depth (m)**	**α ($t\ m^{-3}$)**	**SR ($t\ m^{-2}\ y^{-1}$)**
151	Asco	(i)	Spain	Ebro (Modelled as Rhone 7)	60	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
152	Cofrentes	(i)	Spain	Jucar (Modelled as Rhone 7)	60	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
153	Sta Maria de Garona	(i)	Spain	Ebro (Modelled as Rhone 1)	60	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
154	Vandellos	(c)	Spain	-	60	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
155	Bugey	(i)	France	Rhone 1	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
156	Creys-Malville	(i)	France	Rhone 1	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
157	Cruas Meysse	(i)	France	Rhone 5	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
158	Marcoule	(i)	France	Rhone 7	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
159	St Alban	(i)	France	Rhone 4	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
160	Tricastin	(i)	France	Rhone 6	63	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
161	Gundremmingen	(i)	Germany	Danube (Modelled as Rhine 1)	68	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
162	Isar	(i)	Germany	Danube (Modelled as Rhine 1)	68	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$1.00\ 10^{-6}$	$7.50\ 10^{-5}$
163	Calder Hall	(c)	UK	-	18	$5.00\ 10^{11}$	$2.00\ 10^{9}$	20	$5.00\ 10^{-6}$	$1.00\ 10^{-2}$
164	Civaux	(i)	France	Vienne (Loire 3a)	24	$4.00\ 10^{9}$	$2.00\ 10^{8}$	10	$2.00\ 10^{-4}$	$1.00\ 10^{-4}$

Notes:

a - I = inland site; c = coastal site

b- numbering denotes the river section in which discharge occurs (see Appendix C)

Table 4.7 - Parameter values for regional marine model compartments

Compartment No	Compartment name	Volume (m^3)	Depth (m)	α ($t\ m^{-3}$)	SR ($t\ m^{-2}\ y^{-1}$)	D ($m^2\ y^{-1}$)	ε
1	Other oceans	$8.98\ 10^{17}$	3800	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
2	Atlantic North N.E. (surface 0-1000m)	$1.02\ 10^{16}$	1000	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
3	Atlantic North N.E. (middle 1000-2000m)	$1\ 10^{16}$	1000	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
4	Atlantic North N.E. (deep 2000-4000m)	$2.1\ 10^{16}$	2000	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
5	Other Atlantic	$3.83\ 10^{17}$	$3.22\ 10^{3}$	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
6	Arctic Ocean	$1.69\ 10^{16}$	1200	$1.\ 10^{-7}$	$1.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
7	Arctic South	$2.17\ 10^{16}$	$2.20\ 10^{3}$	$1.\ 10^{-8}$	$3.\ 10^{-6}$	$3.15\ 10^{-3}$	0.3
8	Spitzbergen	$8\ 10^{13}$	1200	$1.\ 10^{-7}$	$1.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
9	Kara and Barents sea	$3.85\ 10^{14}$	$1.33\ 10^{2}$	$1.\ 10^{-7}$	$1.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
10	Norwegian Waters	$1\ 10^{15}$	1200	$1.\ 10^{-7}$	$1.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
11	Scottish Waters W.	$1\ 10^{13}$	110	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
12	Scottish Waters E.	$3\ 10^{12}$	110	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
13	Irish Sea N.W.	$4.08\ 10^{11}$	93	$2.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
14	Irish Sea N.	$6.1\ 10^{10}$	34	$2.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
15	Irish Sea N.E.	$5.2\ 10^{10}$	24	$3.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
16	Irish Sea W.	$6.62\ 10^{11}$	63	$3.\ 10^{-6}$	$1.\ 10^{-3}$	$3.15\ 10^{-2}$	0.75
17	Irish Sea S.E.	$1.62\ 10^{11}$	31	$2.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
18	Cumbrian Waters	$3.8\ 10^{10}$	28	$1.\ 10^{-5}$	$6.\ 10^{-3}$	$3.15\ 10^{-2}$	0.75
19	Irish Sea S.	$1.1\ 10^{12}$	57	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
20	Liverpool and Morecambe Bays	$3.2\ 10^{10}$	13	$3.\ 10^{-6}$	$6.\ 10^{-3}$	$3.15\ 10^{-2}$	0.75
21	Celtic Sea	$2.02\ 10^{13}$	150	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
22	Bristol Channel	$1\ 10^{12}$	50	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
23	Bay of Biscay	$6.5\ 10^{14}$	4000	$1.\ 10^{-7}$	$1.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
24	French Continental Shelf	$3.5\ 10^{13}$	350	$5.\ 10^{-7}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
25	Cantabrian Sea	$3\ 10^{13}$	760	$1.\ 10^{-6}$	$2.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
26	Portuguese Continental Shelf	$1.5\ 10^{13}$	490	$1.\ 10^{-6}$	$2.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
27	Gulf of Cadiz	$2.3\ 10^{14}$	1700	$2.\ 10^{-7}$	$5.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
28	Mediterranean	$3.91\ 10^{15}$	1400	$1.\ 10^{-6}$	$8.\ 10^{-5}$	$3.15\ 10^{-2}$	0.75
29	English Channel W.	$1.41\ 10^{12}$	77.7	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
30	Channel Islands	$6.99\ 10^{11}$	47.2	$3.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
31	Cap de la Hague	$6.16\ 10^{11}$	66.8	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
32	Lyme Bay	$2.01\ 10^{11}$	39.5	$3.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
33	Baie de la Seine	$2.62\ 10^{11}$	34.3	$3.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
34	Sam’s Beach	$9.94\ 10^{10}$	25.3	$3.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
35	Central Channel S.E.	$4.08\ 10^{11}$	49	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
36	Central Channel N.E.	$3.02\ 10^{11}$	49	$1.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
37	Isle of Wight	$1.53\ 10^{11}$	29.5	$5.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
38	North Sea S.W.	$4.5\ 10^{11}$	31	$6.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
39	North Sea S.E.	$9.5\ 10^{11}$	37	$6.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
40	North Sea Central	$1.28\ 10^{13}$	50	$6.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75

Compartment No	Compartment name	Volume (m^3)	Depth (m)	α (t m^{-3})	SR (t m^{-2} y^{-1})	D (m^2 y^{-1})	ε
41	North Sea E.	$1.2\ 10^{12}$	22	$6.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
42	North Sea N.	$5.6\ 10^{13}$	240	$6.\ 10^{-6}$	$1.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
43	Skagerrak	$6.78\ 10^{12}$	210	$1.\ 10^{-6}$	$5.\ 10^{-3}$	$3.15\ 10^{-2}$	0.75
44	Kattegat (surface 0-20m)	$3.2\ 10^{11}$	20	$1.\ 10^{-6}$	$8.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
45	Kattegat (bottom 20-120m)	$2\ 10^{11}$	100	$1.\ 10^{-6}$	$8.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
46	Belt Sea (surface 0-14m)	$1.5\ 10^{11}$	14	$1.\ 10^{-6}$	$8.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
47	Belt Sea (bottom 14-44m)	$1.4\ 10^{11}$	30	$1.\ 10^{-6}$	$8.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
48	Bothnian Bay	$1.48\ 10^{12}$	41	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
49	Bothnian Sea	$4.89\ 10^{12}$	62	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
50	Baltic Sea W. (surface 0-49m)	$3.79\ 10^{12}$	49	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
51	Baltic Sea E. (surface 0-53m)	$6.97\ 10^{12}$	53	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
52	Baltic Sea W. (bottom 49-159m)	$7.7\ 10^{11}$	110	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
53	Baltic Sea E. (bottom 53-163m)	$1.53\ 10^{12}$	110	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
54	Gulf of Finland	$1.1\ 10^{12}$	37	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75
55	Gulf of Riga	$4.05\ 10^{11}$	23	$1.\ 10^{-6}$	$5.\ 10^{-4}$	$3.15\ 10^{-2}$	0.75

Table 4.8 Exchange rates for regional marine model

Exchange rate (m^3 y^{-1})	From compartment number	To compartment number	Exchange rate (m^3 y^{-1})	From compartment number	To compartment number
$6.94\ 10^{14}$	1	5	$6.00\ 10^{11}$	16	19
$3.19\ 10^{13}$	2	3	$5.68\ 10^{11}$	17	14
$5.89\ 10^{14}$	2	5	$4.33\ 10^{11}$	17	16
$3.47\ 10^{14}$	2	7	$2.30\ 10^{11}$	17	18
$2.00\ 10^{14}$	3	2	$7.50\ 10^{10}$	17	19
$3.22\ 10^{13}$	3	4	$1.29\ 10^{11}$	17	20
$1.40\ 10^{14}$	3	5	$2.05\ 10^{11}$	18	15
$1.30\ 10^{14}$	4	5	$1.45\ 10^{11}$	18	17
$3.16\ 10^{13}$	4	7	$3.50\ 10^{10}$	18	20
$1.09\ 10^{15}$	5	1	$1.32\ 10^{12}$	19	16
$8.60\ 10^{14}$	5	2	$7.50\ 10^{10}$	19	17
$4.73\ 10^{13}$	5	3	$6.00\ 10^{11}$	19	21
$3.16\ 10^{13}$	5	4	$1.09\ 10^{11}$	20	17
$6.16\ 10^{13}$	5	7	$5.50\ 10^{10}$	20	18
$1.46\ 10^{14}$	6	7	$2.60\ 10^{12}$	21	7
$9.45\ 10^{12}$	6	9	$1.32\ 10^{12}$	21	19
$1.00\ 10^{14}$	7	8	$2.00\ 10^{12}$	21	22
$1.04\ 10^{13}$	7	11	$1.50\ 10^{14}$	21	23
$1.03\ 10^{13}$	7	21	$1.40\ 10^{14}$	21	24
$4.60\ 10^{13}$	7	42	$8.65\ 10^{12}$	21	29
$8.00\ 10^{13}$	8	6	$2.00\ 10^{12}$	22	21
$2.00\ 10^{13}$	8	7	$5.70\ 10^{14}$	23	2
$3.15\ 10^{13}$	9	10	$1.00\ 10^{14}$	23	3
$1.89\ 10^{13}$	9	7	$1.50\ 10^{14}$	23	21
$3.15\ 10^{13}$	10	7	$5.80\ 10^{14}$	23	24
$6.00\ 10^{13}$	10	9	$3.90\ 10^{14}$	23	25
$9.00\ 10^{11}$	11	7	$1.40\ 10^{14}$	24	21
$1.07\ 10^{13}$	11	12	$5.80\ 10^{14}$	24	23
$2.00\ 10^{11}$	11	13	$7.50\ 10^{13}$	24	25
$5.00\ 10^{11}$	12	11	$1.10\ 10^{14}$	25	2
$8.00\ 10^{12}$	12	40	$3.90\ 10^{14}$	25	23
$2.40\ 10^{12}$	12	42	$7.50\ 10^{13}$	25	24
$9.20\ 10^{11}$	13	11	$1.50\ 10^{13}$	25	26
$3.33\ 10^{11}$	13	14	$4.60\ 10^{14}$	26	2
$5.00\ 10^{11}$	13	16	$1.30\ 10^{13}$	26	25
$8.33\ 10^{11}$	14	13	$6.00\ 10^{13}$	26	27
$1.83\ 10^{11}$	14	15	$4.10\ 10^{14}$	27	2
$1.73\ 10^{11}$	14	17	$1.00\ 10^{14}$	27	3
$2.88\ 10^{11}$	15	14	$5.80\ 10^{13}$	27	26
$1.00\ 10^{11}$	15	18	$5.29\ 10^{13}$	27	28
$7.20\ 10^{11}$	16	13	$5.06\ 10^{13}$	28	27
$9.33\ 10^{11}$	16	17	$3.69\ 10^{12}$	29	21

Exchange rate ($m^3\ y^{-1}$)	From compartment number	To compartment number	Exchange rate ($m^3\ y^{-1}$)	From compartment number	To compartment number
$2.95\ 10^{12}$	29	30	$8.96\ 10^{12}$	40	42
$3.30\ 10^{12}$	29	31	$2.68\ 10^{11}$	40	43
$1.27\ 10^{12}$	29	32	$1.50\ 10^{11}$	41	39
$1.51\ 10^{12}$	30	29	$1.63\ 10^{12}$	41	40
$6.25\ 10^{12}$	30	31	$1.73\ 10^{12}$	42	7
$8.97\ 10^{11}$	31	29	$6.00\ 10^{13}$	42	10
$4.81\ 10^{12}$	31	30	$1.00\ 10^{11}$	42	12
$2.73\ 10^{12}$	31	32	$2.05\ 10^{12}$	42	40
$4.97\ 10^{12}$	31	35	$2.58\ 10^{13}$	42	43
$1.07\ 10^{12}$	31	36	$1.68\ 10^{11}$	43	40
$1.56\ 10^{11}$	32	29	$1.00\ 10^{10}$	43	41
$3.41\ 10^{12}$	32	31	$3.23\ 10^{13}$	43	42
$6.29\ 10^{11}$	32	37	$1.50\ 10^{12}$	43	45
$7.70\ 10^{11}$	33	34	$2.00\ 10^{12}$	44	43
$2.61\ 10^{12}$	33	35	$1.00\ 10^{11}$	44	45
$1.70\ 10^{11}$	34	33	$9.30\ 10^{11}$	45	44
$6.53\ 10^{12}$	34	35	$7.20\ 10^{11}$	45	47
$6.07\ 10^{11}$	34	39	$1.20\ 10^{12}$	46	44
$1.24\ 10^{12}$	35	31	$7.00\ 10^{11}$	46	47
$3.21\ 10^{12}$	35	33	$9.30\ 10^{11}$	47	46
$6.38\ 10^{12}$	35	34	$2.70\ 10^{11}$	47	51
$9.45\ 10^{12}$	35	36	$2.20\ 10^{11}$	47	53
$2.43\ 10^{12}$	35	39	$2.75\ 10^{11}$	48	49
$2.86\ 10^{11}$	36	31	$1.75\ 10^{11}$	49	48
$8.43\ 10^{12}$	36	35	$7.15\ 10^{11}$	49	50
$6.89\ 10^{12}$	36	37	$6.97\ 10^{12}$	50	51
$1.92\ 10^{12}$	36	39	$1.07\ 10^{11}$	50	52
$1.95\ 10^{11}$	37	32	$5.25\ 10^{11}$	51	49
$6.85\ 10^{12}$	37	36	$6.97\ 10^{12}$	51	50
$5.47\ 10^{11}$	37	39	$2.08\ 10^{11}$	51	53
$6.09\ 10^{11}$	38	39	$5.95\ 10^{11}$	51	54
$3.81\ 10^{11}$	38	40	$3.12\ 10^{11}$	51	55
$1.60\ 10^{11}$	39	34	$2.20\ 10^{11}$	52	46
$1.63\ 10^{11}$	39	35	$1.07\ 10^{11}$	52	50
$1.47\ 10^{11}$	39	36	$2.20\ 10^{11}$	52	53
$7.60\ 10^{10}$	39	37	$2.08\ 10^{11}$	53	51
$2.94\ 10^{11}$	39	38	$4.40\ 10^{11}$	53	52
$5.12\ 10^{11}$	39	40	$7.20\ 10^{11}$	54	51
$5.01\ 10^{12}$	39	41	$3.44\ 10^{11}$	55	51
$1.00\ 10^{11}$	40	12			
$6.96\ 10^{11}$	40	38			
$5.40\ 10^{10}$	40	39			
$2.66\ 10^{12}$	40	41			

Table 4.9 – Marine Sediment K_d Bq t^{-1} per Bq m^{-3})

Element	Coast	Ref	Ocean	Ref
H	$1.0\ 10^{0}$	IAEA (1985)	$1.0\ 10^{0}$	IAEA (1985)
C	$2.0\ 10^{3}$	IAEA (1985)	$2.0\ 10^{3}$	IAEA (1985)
P	$1.0\ 10^{2}$ [a]	IAEA (2001)	$1.0\ 10^{2}$	IAEA (2001)
S	$5.0\ 10^{-1}$	IAEA (1985)	$1.0\ 10^{0}$	IAEA (1985)
Cr	$5.0\ 10^{4}$	IAEA (1985)	$5.0\ 10^{4}$	IAEA (1985)
Mn	$2.0\ 10^{5}$	IAEA (1985)	$2.0\ 10^{8}$	IAEA (1985)
Fe	$5.0\ 10^{4}$	IAEA (1985)	$5.0\ 10^{7}$	IAEA (1985)
Co	$2.0\ 10^{5}$ [b]	IAEA (1985)	$1.0\ 10^{7}$	IAEA (1985)
Ni	$1.0\ 10^{5}$	IAEA (1985)	$1.0\ 10^{6}$	IAEA (1985)
Zn	$2.0\ 10^{4}$	IAEA (1985)	$2.0\ 10^{5}$	IAEA (1985)
Se	$1.0\ 10^{3}$	IAEA (1985)	$1.0\ 10^{3}$	IAEA (1985)
Sr	$1.0\ 10^{3}$	IAEA (1985)	$2.0\ 10^{2}$	IAEA (1985)
Zr	$1.0\ 10^{6}$	IAEA (1985)	$5.0\ 10^{5}$	IAEA (1985)
Nb	$5.0\ 10^{5}$	IAEA (1985)	$2.0\ 10^{5}$	IAEA (1985)
Tc	$1.0\ 10^{2}$	IAEA (1985)	$1.0\ 10^{3}$	Bishop et al (1989)
Ru	$3.0\ 10^{2}$ [b]	IAEA (1985)	$1.0\ 10^{3}$	IAEA (1985)
Sb	$1.0\ 10^{3}$	IAEA (1985)	$5.0\ 10^{2}$	IAEA (1985)
Te	$1.0\ 10^{3}$	IAEA (1985)	$1.0\ 10^{3}$	IAEA (1985)
I	$2.0\ 10^{1}$	IAEA (1985)	$2.0\ 10^{2}$	IAEA (1985)
Cs	$3\ 10^{3}$ [b]	IAEA (1985)	$2.0\ 10^{3}$	IAEA (1985)
Ba	$5.0\ 10^{3}$	IAEA (1985)	$1.0\ 10^{4}$	IAEA (1985)
La	$2.0\ 10^{6}$ [c]		$1.0\ 10^{8}$ [c]	
Ce	$2.0\ 10^{6}$	IAEA (1985)	$1.0\ 10^{8}$	IAEA (1985)
Pm	$2.0\ 10^{6}$	IAEA (1985)	$1.0\ 10^{6}$	IAEA (1985)
Eu	$5.0\ 10^{5}$	IAEA (1985)	$4.0\ 10^{6}$	IAEA (1985)
Pb	$5.0\ 10^{3}$	Mc Donald et al (1992)	$3.0\ 10^{7}$	IAEA (1985)
Po	$1.0\ 10^{4}$	Mc Donald et al (1992)	$2.0\ 10^{7}$	IAEA (1985)
Ra	$5.0\ 10^{3}$	IAEA (1985)	$3.0\ 10^{4}$	IAEA (1985)
Th	$2.0\ 10^{6}$	IAEA (1985)	$5.0\ 10^{6}$	IAEA (1985)
U	$1.0\ 10^{3}$	IAEA (1985)	$5.0\ 10^{2}$	IAEA (1985)
Np	$1.0\ 10^{3}$	Bishop et al (1989)	$1.0\ 10^{3}$	IAEA (2004)
Pu	$1.0\ 10^{5}$	IAEA (1985)	$1.0\ 10^{5}$	IAEA (1985)
Am	$2.0\ 10^{6}$ [b]	IAEA (1985)	$2.0\ 10^{6}$	IAEA (1985)
Cm	$2.0\ 10^{6}$	IAEA (1985)	$2.0\ 10^{6}$	IAEA (1985)

Notes:

a- Assume coastal K_d the same as deep K_d

b- Values for coastal K_d for Irish sea: ^{60}Co – 2.5 10^{3}; ^{106}Ru – 7.1 10^{2} ; ^{137}Cs – 2.3 10^{2} (Titley, 2001; Mc Donald et al, 1992); ^{241}Am - 1 10^{5} (Mitchell et al, 1999).

c- Use value for cerium as chemically analogous (Lanthanide)

Table 4.10 - Parameter values used in the calculation of activity concentrations in seaspray

Parameter	Cs	Pu	Am
TDV (m y^{-1})	3.15 10^6	9.46 10^6	9.46 10^6
A (m y^{-1})	0.00555	0.103	0.057
*B**	0.95	2.95	1.77
α (km^{-1})	0.028	0.025	0.065
β (km^{-1})	0.37	0.616	0.44

Notes:

*: Values apply for distances of less or equal to 2 km. For greater distances *B* = 0

Table 4.11 - Default sediment model parameter values common to all compartments (Simmonds et al, 2002)

Parameter	Default value	Description
Lt	0.1 m	Thickness of top sediment layer
Lm	1.9 m	Thickness of middle sediment layer
P	2.6 t m^{-3}	Sediment mineral density
RW	5 10^{-3} m y^{-1}	Sediment reworking rate for shallow seas up to 200 m, also local compartment.
	5 10^{-4} m y^{-1}	Sediment reworking rate deep for deep seas greater than 200 m.
RT	1 y^{-1}	Pore water turn over rate for shallow seas up to 200 m, also local compartment.
	0.1 y^{-1}	Pore water turn over rate for deep seas greater than 200 m.
D	3.15 10^{-2} m^{-2} y^{-1}	Sediment diffusion rate for local compartments (see Table 4.9 for regional compartments)
ε	0.75	Sediment porosity for local compartment (see Table 4.9 for regional compartments)

Table 4.12 – Marine concentration factors Bq t^{-1} per Bq m^{-3}

Element	Fish	Ref	Crustacea	Ref	Mollusc	Ref	Seaweed	Ref
H	1.0 10^{0}	IAEA (1985)	1.0 10^{0}	IAEA (1985)	1.0 10^{0}	IAEA (1985)	1.0 10^{0}	IAEA (1985)
C	2.0 10^{4}	IAEA (1985)	2.0 10^{4}	IAEA (1985)	2.0 10^{4}	IAEA (1985)	1.0 10^{4}	IAEA (1985)
P	3.0 10^{4}	IAEA (2001)	2.0 10^{4}[a]		2.0 10^{4}	IAEA (2001)	2.0 10^{4}[a]	
S	2.0 10^{0}	IAEA (1985)	1.0 10^{0}	IAEA (1985)	4.0 10^{0}	IAEA (1985)	4.0 10^{0}	IAEA (1985)
Cr	2.0 10^{2}	IAEA (1985)	5.0 10^{2}	IAEA (1985)	8.0 10^{2}	IAEA (1985)	2.0 10^{3}	IAEA (1985)
Mn	4.0 10^{2}	IAEA (1985)	5.0 10^{2}	IAEA (1985)	5.0 10^{4}	IAEA (2004)	6.0 10^{3}	IAEA (1985)
Fe	5.0 10^{2}	Pentreath (1977)	5.0 10^{3}	IAEA (1985)	3.0 10^{4}	IAEA (1985)	3.0 10^{4}	IAEA (1985)
Co	1.0 10^{3}	IAEA (1985)	1.0 10^{4}	IAEA (1985)	5.0 10^{3}	IAEA (1985)	1.0 10^{4}	IAEA (1985)
Ni	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	2.0 10^{3}	IAEA (1985)	2.0 10^{3}	IAEA (1985)
Zn	1.0 10^{3}	IAEA (1985)	5.0 10^{4}	IAEA (1985)	3.0 10^{4}	IAEA (1985)	2.0 10^{3}	IAEA (2004)
Se	6.0 10^{3}	IAEA (1985)	5.0 10^{3}	IAEA (1985)	6.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)
Sr	2.0 10^{0}	IAEA (1985)	2.0 10^{0}	IAEA (1985)	1.0 10^{0}	IAEA (1985)	5.0 10^{0}	IAEA (1985)
Zr	2.0 10^{1}	IAEA (1985)	2.0 10^{2}	IAEA (1985)	5.0 10^{3}	IAEA (1985)	3.0 10^{3}	IAEA (1985)
Nb	3.0 10^{1}	IAEA (1985)	2.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	3.0 10^{3}	IAEA (1985)
Tc	3.0 10^{1}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	3.0 10^{4}	IAEA (2004)
Ru	2.0 10^{0}	IAEA (1985)	1.0 10^{2}	IAEA (1985)	2.0 10^{3}	IAEA (1985)	2.0 10^{3}	IAEA (1985)
Sb	4.0 10^{2}	IAEA (1985)	2.5 10^{1}	Swift and Kershaw (1999)	2.0 10^{1}	Swift and Kershaw (1999)	2.0 10^{1}	IAEA (2004)
Te	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{4}	IAEA (1985)
I	1.0 10^{1}	IAEA (1985)	1.0 10^{1}	IAEA (1985)	1.0 10^{1}	IAEA (1985)	1.0 10^{3}	IAEA (1985)
Cs	1.0 10^{2}	IAEA (1985)	3.0 10^{1}	IAEA (1985)	3.0 10^{1}	IAEA (1985)	5.0 10^{1}	IAEA (1985)
Ba	1.0 10^{1}	IAEA (1985)	1.0 10^{0}	IAEA (1985)	2.0 10^{1}	IAEA (1985)	1.0 10^{2}	IAEA (1985)
La	1.0 10^{2}	NCRP (1996)	1.0 10^{3}[b]		1.0 10^{3}	NCRP (1996)	5.0 10^{3}[b]	
Ce	5.0 10^{1}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	2.0 10^{3}	IAEA (2004)	5.0 10^{3}	IAEA (1985)
Pm	5.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	5.0 10^{3}	IAEA (1985)	3.0 10^{3}	IAEA (1985)
Eu	3.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	7.0 10^{3}	IAEA (1985)	3.0 10^{3}	IAEA (1985)
Pb	2.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)
Po	2.0 10^{4}	Swift and Kershaw	5.0 10^{4}	IAEA (1985)	1.0 10^{4}	IAEA (1985)	1.0 10^{3}	IAEA (1985)

Element	Fish	Ref	Crustacea	Ref	Mollusc	Ref	Seaweed	Ref
		(1999)						
Ra	5.0 10^{2}	IAEA (1985)	1.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{2}	IAEA (1985)
Th	6.0 10^{2}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	1.0 10^{3}	IAEA (1985)	2.0 10^{2}	IAEA (1985)
U	1.0 10^{0}	IAEA (1985)	1.0 10^{1}	IAEA (1985)	3.0 10^{1}	IAEA (1985)	1.0 10^{2}	IAEA (1985)
Np	1.0 10^{1}	IAEA (1985)	1.0 10^{2}	IAEA (1985)	4.0 10^{2}	IAEA (1985)	5.0 10^{1}	IAEA (1985)
Pu	1.0 10^{2}	Swift and Kershaw (1999)	2.0 10^{2}	Gomez et al (1985)	3.0 10^{3}	IAEA (1985)	2.0 10^{3}	IAEA (1985)
Am	1.0 10^{2}	Coughtrey et al (1984)	5.0 10^{2}	IAEA (1985)	2.0 10^{4}	IAEA (1985)	8.0 10^{3}	IAEA (1985)
Cm	5.0 10^{1}	IAEA (1985)	5.0 10^{2}	IAEA (1985)	3.0 10^{4}	IAEA (1985)	8.0 10^{3}	IAEA (1985)

Notes:

a- Assume same value as for mollusc

b - Use value for cerium as chemically analogous (Lanthanide) (IAEA, 1985)

Table 4.13 Annual marine fish consumption by model compartment[a]

Nation	Annual marine fish consumption t y^{-1}										
	Atlantic North NE (surface 0-1000 m)	Baie de la Seine	Baltic Sea E (bottom 53-163 m)	Baltic Sea E (surface 0-53 m)	Baltic Sea W (bottom 49-159 m)	Baltic Sea W (surface 0-49 m)	Bay of Biscay	Belt Sea (bottom 14-44 m)	Belt Sea (surface 0-14 m)	Bothnian Bay	Bothnian Sea
Austria	8.0 10^{2}	0.0 10^{0}					0.0 10^{0}				
Belgium	4.5 10^{3}	3.3 10^{2}	7.8 10^{1}	3.6 10^{2}	4.9 10^{1}	2.4 10^{2}	2.5 10^{1}	9.4 10^{1}	1.0 10^{2}	1.1 10^{0}	4.0 10^{0}
Denmark	2.5 10^{4}	6.9 10^{2}	9.9 10^{2}	4.5 10^{3}	2.8 10^{3}	1.4 10^{4}	0.0 10^{0}	4.8 10^{3}	5.2 10^{3}	5.0 10^{1}	1.8 10^{2}
Finland	6.5 10^{2}	5.0 10^{1}	1.4 10^{3}	6.2 10^{3}	7.0 10^{2}	3.4 10^{3}	0.0 10^{0}	3.4 10^{1}	3.6 10^{1}	3.8 10^{3}	1.7 10^{4}
France	3.9 10^{4}	1.0 10^{1}	5.7 10^{2}	2.6 10^{3}	2.8 10^{2}	1.4 10^{3}	7.0 10^{2}	5.6 10^{2}	5.9 10^{2}	2.0 10^{0}	7.0 10^{0}
Germany	4.7 10^{4}	0.0 10^{0}	1.9 10^{3}	8.6 10^{3}	6.5 10^{2}	3.2 10^{3}	1.0 10^{2}	1.4 10^{3}	1.5 10^{3}	6.0 10^{0}	2.0 10^{1}
Greece	4.5 10^{3}	2.5 10^{1}	1.1 10^{1}	4.9 10^{1}	9.3 10^{0}	4.6 10^{1}	0.0 10^{0}	1.9 10^{1}	2.0 10^{1}	0.0 10^{0}	0.0 10^{0}
Ireland	1.5 10^{4}	2.0 10^{2}	3.5 10^{0}	1.6 10^{1}	1.7 10^{1}	8.3 10^{1}	0.0 10^{0}	1.4 10^{1}	1.4 10^{1}	7.0 10^{-1}	2.2 10^{0}
Italy	7.5 10^{3}	3.4 10^{2}	1.6 10^{2}	7.4 10^{2}	1.3 10^{2}	6.2 10^{2}	1.0 10^{2}	2.7 10^{2}	2.8 10^{2}	2.0 10^{-1}	7.0 10^{-1}
Netherlands	2.4 10^{4}	3.0 10^{1}	2.5 10^{2}	1.1 10^{3}	2.4 10^{2}	1.2 10^{3}	0.0 10^{0}	3.6 10^{2}	3.9 10^{2}	7.5 10^{0}	2.6 10^{1}
Norway	7.5 10^{3}	3.5 10^{2}					0.0 10^{0}				
Portugal	1.5 10^{4}	2.4 10^{2}	5.0 10^{1}	2.3 10^{2}	4.6 10^{1}	2.3 10^{2}	5.0 10^{1}	7.7 10^{1}	8.3 10^{1}	8.0 10^{-1}	2.4 10^{0}
Spain	3.0 10^{4}	2.6 10^{3}	1.2 10^{2}	5.3 10^{2}	8.2 10^{1}	4.0 10^{2}	1.1 10^{3}	1.7 10^{2}	1.8 10^{2}	6.5 10^{-1}	2.6 10^{0}
Sweden	1.1 10^{3}	1.5 10^{1}	1.1 10^{3}	4.9 10^{3}	7.6 10^{3}	3.7 10^{4}	0.0 10^{0}	3.1 10^{3}	3.4 10^{3}	4.4 10^{2}	1.3 10^{3}
United Kingdom	8.5 10^{4}	1.0 10^{1}	1.5 10^{2}	7.0 10^{2}	1.0 10^{2}	5.0 10^{2}	2.5 10^{1}	2.1 10^{2}	2.2 10^{2}	6.0 10^{-1}	1.9 10^{0}
World	8.2 10^{5}	6.4 10^{3}	2.3 10^{4}	1.1 10^{5}	1.4 10^{4}	6.8 10^{4}	3.0 10^{3}	1.2 10^{4}	1.2 10^{4}	4.5 10^{3}	1.9 10^{4}

Table 4.13 continued Annual marine fish consumption by model compartment [a]

Nation	Annual marine fish consumption t y^{-1}										
	Bristol Channel	Cantabrian Sea	Cap de la Hague	Celtic Sea	Central Channel NE	Central Channel SE	Channel Islands	Cumbrian Waters	English Channel W	French Continental Shelf	Gulf of Cadiz
Austria	$0.0\ 10^{0}$	$2.5\ 10^{1}$	$1.5\ 10^{1}$	$1.5\ 10^{2}$	$2.0\ 10^{1}$	$2.5\ 10^{1}$	$2.0\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.0\ 10^{2}$	$1.5\ 10^{2}$
Belgium	$7.0\ 10^{2}$	$5.0\ 10^{1}$	$1.5\ 10^{2}$	$1.2\ 10^{3}$	$5.0\ 10^{2}$	$6.8\ 10^{2}$	$1.7\ 10^{2}$	$8.0\ 10^{0}$	$1.3\ 10^{3}$	$9.0\ 10^{2}$	$1.5\ 10^{2}$
Denmark	$2.5\ 10^{1}$	$5.0\ 10^{1}$	$1.3\ 10^{2}$	$5.0\ 10^{2}$	$3.5\ 10^{1}$	$5.0\ 10^{1}$	$1.5\ 10^{2}$	$5.0\ 10^{-1}$	$8.3\ 10^{3}$	$1.0\ 10^{2}$	$5.0\ 10^{1}$
Finland	$1.7\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.5\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
France	$2.2\ 10^{3}$	$2.1\ 10^{3}$	$1.8\ 10^{3}$	$1.5\ 10^{4}$	$3.7\ 10^{3}$	$4.9\ 10^{3}$	$2.1\ 10^{3}$	$2.0\ 10^{1}$	$3.5\ 10^{1}$	$2.5\ 10^{4}$	$1.6\ 10^{3}$
Germany	$1.0\ 10^{2}$	$3.0\ 10^{2}$	$5.0\ 10^{2}$	$4.5\ 10^{3}$	$4.7\ 10^{2}$	$6.4\ 10^{2}$	$5.7\ 10^{2}$	$5.0\ 10^{0}$	$5.0\ 10^{0}$	$1.8\ 10^{3}$	$2.5\ 10^{2}$
Greece	$0.0\ 10^{0}$	$1.5\ 10^{2}$	$1.0\ 10^{1}$	$1.0\ 10^{2}$	$1.0\ 10^{1}$	$1.5\ 10^{1}$	$1.5\ 10^{1}$	$0.0\ 10^{0}$	$3.3\ 10^{2}$	$5.0\ 10^{1}$	$1.5\ 10^{2}$
Ireland	$2.5\ 10^{1}$	$0.0\ 10^{0}$	$7.5\ 10^{1}$	$2.4\ 10^{3}$	$1.0\ 10^{1}$	$1.5\ 10^{1}$	$8.5\ 10^{1}$	$1.5\ 10^{1}$	$6.0\ 10^{2}$	$5.0\ 10^{1}$	$0.0\ 10^{0}$
Italy	$1.0\ 10^{2}$	$4.1\ 10^{3}$	$2.3\ 10^{2}$	$1.9\ 10^{3}$	$2.9\ 10^{2}$	$4.0\ 10^{2}$	$2.6\ 10^{2}$	$2.0\ 10^{0}$	$1.1\ 10^{3}$	$2.0\ 10^{3}$	$2.1\ 10^{3}$
Netherlands	$1.0\ 10^{2}$	$5.0\ 10^{1}$	$3.7\ 10^{2}$	$3.5\ 10^{3}$	$4.8\ 10^{2}$	$6.5\ 10^{2}$	$4.5\ 10^{2}$	$3.5\ 10^{0}$	$1.0\ 10^{2}$	$1.2\ 10^{3}$	$1.0\ 10^{2}$
Norway	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.8\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Portugal	$2.5\ 10^{1}$	$2.8\ 10^{3}$	$5.0\ 10^{1}$	$2.5\ 10^{2}$	$4.5\ 10^{1}$	$6.0\ 10^{1}$	$5.0\ 10^{1}$	$5.0\ 10^{-1}$	$8.5\ 10^{2}$	$5.0\ 10^{2}$	$3.7\ 10^{4}$
Spain	$2.5\ 10^{2}$	$4.8\ 10^{4}$	$3.3\ 10^{2}$	$3.3\ 10^{3}$	$3.5\ 10^{2}$	$4.6\ 10^{2}$	$3.8\ 10^{2}$	$1.0\ 10^{1}$	$4.7\ 10^{3}$	$9.0\ 10^{3}$	$2.0\ 10^{4}$
Sweden	$2.5\ 10^{1}$	$2.5\ 10^{1}$	$5.5\ 10^{1}$	$4.0\ 10^{2}$	$5.0\ 10^{1}$	$8.0\ 10^{1}$	$6.0\ 10^{1}$	$5.0\ 10^{-1}$	$5.0\ 10^{1}$	$3.5\ 10^{2}$	$5.0\ 10^{1}$
United Kingdom	$1.7\ 10^{3}$	$4.0\ 10^{2}$	$3.2\ 10^{3}$	$5.0\ 10^{3}$	$1.0\ 10^{3}$	$1.3\ 10^{3}$	$3.6\ 10^{3}$	$7.5\ 10^{1}$	$2.0\ 10^{2}$	$5.0\ 10^{2}$	$5.0\ 10^{2}$
World	$6.0\ 10^{3}$	$7.1\ 10^{4}$	$8.8\ 10^{3}$	$5.7\ 10^{4}$	$9.1\ 10^{3}$	$1.2\ 10^{4}$	$1.0\ 10^{4}$	$1.9\ 10^{2}$	$2.3\ 10^{4}$	$5.2\ 10^{4}$	$6.9\ 10^{4}$

Table 4.13 continued Annual marine fish consumption by model compartment [a]

Nation	Annual marine fish consumption t y^{-1}										
	Gulf of Finland	Gulf of Riga	Irish Sea N	Irish Sea NE	Irish Sea NW	Irish Sea S	Irish Sea SE	Irish Sea W	Isle of Wight	Kara and Barents sea	Kattegat (bottom 20-120 m)
Austria			0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	4.0 10^{0}	5.0 10^{-1}	4.5 10^{0}	1.0 10^{1}	0.0 10^{0}	5.0 10^{1}
Belgium	9.5 10^{-1}	3.0 10^{-1}	2.0 10^{1}	1.5 10^{1}	1.0 10^{2}	3.5 10^{2}	4.5 10^{1}	2.1 10^{2}	1.7 10^{2}	1.0 10^{1}	1.0 10^{2}
Denmark	2.1 10^{2}	1.3 10^{2}	1.5 10^{0}	1.0 10^{0}	1.1 10^{3}	2.5 10^{1}	3.0 10^{0}	1.0 10^{1}	1.0 10^{1}	3.0 10^{3}	1.7 10^{3}
Finland	3.8 10^{3}	2.9 10^{0}	0.0 10^{0}	0.0 10^{0}	1.0 10^{1}	1.0 10^{0}	0.0 10^{0}	5.0 10^{-1}	0.0 10^{0}	1.0 10^{2}	5.0 10^{1}
France	6.5 10^{0}	1.6 10^{0}	5.0 10^{1}	2.5 10^{1}	3.0 10^{0}	7.6 10^{2}	9.0 10^{1}	4.4 10^{2}	1.1 10^{3}	4.0 10^{2}	7.0 10^{2}
Germany	7.0 10^{1}	1.5 10^{1}	2.0 10^{1}	1.0 10^{1}	0.0 10^{0}	3.0 10^{2}	3.5 10^{1}	1.8 10^{2}	1.5 10^{2}	2.1 10^{3}	8.5 10^{2}
Greece	1.5 10^{-1}	0.0 10^{0}	5.0 10^{-1}	0.0 10^{0}	1.0 10^{1}	1.0 10^{1}	1.0 10^{0}	5.0 10^{0}	5.0 10^{0}	5.5 10^{1}	2.5 10^{1}
Ireland	0.0 10^{0}	0.0 10^{0}	4.0 10^{1}	2.5 10^{1}	3.0 10^{1}	6.4 10^{2}	8.0 10^{1}	3.7 10^{2}	5.0 10^{0}	1.0 10^{1}	2.5 10^{1}
Italy	9.5 10^{-1}	3.1 10^{1}	5.0 10^{0}	2.5 10^{0}	5.0 10^{1}	9.0 10^{1}	1.0 10^{1}	5.5 10^{1}	8.5 10^{1}	7.5 10^{1}	2.5 10^{2}
Netherlands	6.0 10^{1}	1.0 10^{1}	1.0 10^{1}	5.0 10^{0}	5.0 10^{0}	1.5 10^{2}	2.0 10^{1}	8.5 10^{1}	1.5 10^{2}	4.5 10^{2}	2.0 10^{2}
Norway			0.0 10^{0}	0.0 10^{0}	1.2 10^{2}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	2.2 10^{3}	1.0 10^{2}
Portugal	2.8 10^{1}	1.2 10^{1}	1.5 10^{0}	5.0 10^{-1}	1.1 10^{2}	2.0 10^{1}	2.5 10^{0}	1.5 10^{1}	1.0 10^{1}	5.0 10^{2}	1.0 10^{2}
Spain	4.3 10^{1}	5.0 10^{1}	2.0 10^{1}	1.0 10^{1}	2.6 10^{2}	3.3 10^{2}	4.0 10^{1}	1.9 10^{2}	1.0 10^{2}	1.5 10^{2}	2.5 10^{2}
Sweden	1.1 10^{2}	9.0 10^{1}	1.5 10^{0}	1.0 10^{0}	2.0 10^{0}	2.5 10^{1}	3.0 10^{0}	1.5 10^{1}	2.0 10^{1}	0.0 10^{0}	5.5 10^{3}
United Kingdom	6.0 10^{1}	4.1 10^{0}	2.1 10^{2}	1.1 10^{2}	2.2 10^{2}	3.1 10^{3}	3.8 10^{2}	1.8 10^{3}	3.1 10^{2}	1.5 10^{3}	1.5 10^{2}
World	1.6 10^{4}	1.1 10^{4}	5.0 10^{2}	2.7 10^{2}	2.6 10^{3}	7.6 10^{3}	9.3 10^{2}	4.5 10^{3}	2.9 10^{3}	1.1 10^{5}	3.0 10^{4}

Table 4.13 continued Annual marine fish consumption by model compartment [a]

Nation	Annual marine fish consumption t y^{-1}										
	Kattegat (surface 0-20 m)	Liverpool and Morecambe Bays	Lyme Bay	Mediterranean	North Sea Central	North Sea E	North Sea N	North Sea SE	North Sea SW	Norwegian Waters	Portuguese Continental Shelf
Austria	5.0 10^{1}	0.0 10^{0}	5.0 10^{0}		8.0 10^{2}	1.0 10^{2}	3.5 10^{2}	7.5 10^{1}	7.5 10^{1}	2.0 10^{2}	1.0 10^{2}
Belgium	1.0 10^{2}	1.5 10^{1}	3.0 10^{1}	1.5 10^{3}	6.5 10^{3}	6.5 10^{2}	1.4 10^{3}	2.0 10^{3}	2.0 10^{3}	6.5 10^{2}	1.0 10^{2}
Denmark	1.7 10^{3}	1.5 10^{0}	2.5 10^{1}	6.4 10^{2}	1.9 10^{4}	1.9 10^{3}	1.4 10^{4}	3.0 10^{2}	3.0 10^{2}	3.1 10^{4}	5.0 10^{1}
Finland	5.0 10^{1}	0.0 10^{0}	0.0 10^{0}		2.0 10^{2}	5.0 10^{1}	2.5 10^{2}	0.0 10^{0}	0.0 10^{0}	8.0 10^{2}	0.0 10^{0}
France	7.0 10^{2}	3.0 10^{1}	3.8 10^{2}	1.7 10^{4}	1.6 10^{4}	1.6 10^{3}	1.7 10^{4}	2.8 10^{3}	2.8 10^{3}	7.0 10^{3}	1.6 10^{3}
Germany	8.5 10^{2}	1.5 10^{1}	1.1 10^{2}	6.1 10^{3}	2.4 10^{4}	2.5 10^{3}	2.1 10^{4}	1.4 10^{3}	1.4 10^{3}	2.5 10^{4}	2.5 10^{2}
Greece	2.5 10^{1}	5.0 10^{-1}	5.0 10^{0}	3.5 10^{4}	5.0 10^{2}	5.0 10^{1}	2.5 10^{2}	5.0 10^{1}	5.0 10^{1}	2.5 10^{2}	1.0 10^{2}
Ireland	2.5 10^{1}	2.5 10^{1}	1.5 10^{1}	7.0 10^{0}	4.0 10^{2}	5.0 10^{1}	1.3 10^{3}	2.5 10^{1}	2.5 10^{1}	1.1 10^{3}	0.0 10^{0}
Italy	2.5 10^{2}	5.0 10^{0}	5.0 10^{1}	1.2 10^{5}	7.0 10^{3}	7.0 10^{2}	3.0 10^{3}	1.0 10^{3}	1.0 10^{3}	1.6 10^{3}	2.2 10^{3}
Netherlands	2.0 10^{2}	5.0 10^{0}	6.5 10^{1}	3.9 10^{3}	1.2 10^{4}	1.2 10^{3}	4.9 10^{3}	2.9 10^{3}	2.9 10^{3}	6.0 10^{3}	5.0 10^{1}
Norway	1.0 10^{2}	0.0 10^{0}	0.0 10^{0}		3.1 10^{3}	3.5 10^{2}	7.5 10^{3}	1.0 10^{2}	1.0 10^{2}	2.2 10^{4}	0.0 10^{0}
Portugal	1.0 10^{2}	1.5 10^{0}	1.0 10^{1}	3.6 10^{3}	2.5 10^{3}	2.5 10^{2}	2.4 10^{3}	1.5 10^{2}	1.5 10^{2}	7.0 10^{3}	3.9 10^{4}
Spain	2.5 10^{2}	1.5 10^{1}	7.0 10^{1}	4.9 10^{4}	6.5 10^{3}	6.5 10^{2}	5.0 10^{3}	9.0 10^{2}	9.0 10^{2}	3.8 10^{4}	2.1 10^{4}
Sweden	5.5 10^{3}	1.5 10^{0}	1.0 10^{1}		5.0 10^{3}	5.0 10^{2}	2.2 10^{3}	1.0 10^{2}	1.0 10^{2}	3.3 10^{3}	5.0 10^{1}
United Kingdom	1.5 10^{2}	1.3 10^{2}	6.8 10^{2}	2.5 10^{3}	3.2 10^{4}	3.1 10^{3}	7.5 10^{4}	1.8 10^{3}	1.8 10^{3}	2.1 10^{4}	5.0 10^{1}
World	3.0 10^{4}	3.3 10^{2}	1.9 10^{3}	2.4 10^{5}	2.5 10^{5}	2.5 10^{4}	2.9 10^{5}	2.3 10^{4}	2.3 10^{4}	5.6 10^{5}	7.2 10^{4}

Table 4.13 continued Annual marine fish consumption by model compartment [a]

Nation	Annual marine fish consumption t y^{-1}				
	Sam's Beach	Scottish Waters E	Scottish Waters W	Skagerrak	Spitzbergen
Austria	$5.0\ 10^{0}$	$5.0\ 10^{1}$	$3.5\ 10^{2}$	$1.5\ 10^{2}$	$5.0\ 10^{1}$
Belgium	$1.8\ 10^{2}$	$2.0\ 10^{2}$	$1.8\ 10^{3}$	$3.0\ 10^{2}$	$1.0\ 10^{2}$
Denmark	$1.0\ 10^{1}$	$1.8\ 10^{3}$	$3.5\ 10^{3}$	$5.5\ 10^{3}$	$1.0\ 10^{3}$
Finland	$0.0\ 10^{0}$	$5.0\ 10^{1}$	$1.0\ 10^{2}$	$2.0\ 10^{2}$	$5.0\ 10^{1}$
France	$1.2\ 10^{3}$	$2.1\ 10^{3}$	$2.0\ 10^{4}$	$2.3\ 10^{3}$	$1.0\ 10^{3}$
Germany	$1.6\ 10^{2}$	$2.5\ 10^{2}$	$1.5\ 10^{4}$	$2.6\ 10^{3}$	$2.3\ 10^{3}$
Greece	$5.0\ 10^{0}$	$5.0\ 10^{1}$	$3.0\ 10^{2}$	$5.0\ 10^{1}$	$5.0\ 10^{1}$
Ireland	$5.0\ 10^{0}$	$1.5\ 10^{2}$	$1.2\ 10^{4}$	$5.0\ 10^{1}$	$1.0\ 10^{2}$
Italy	$1.0\ 10^{2}$	$4.0\ 10^{2}$	$3.2\ 10^{3}$	$8.0\ 10^{2}$	$4.5\ 10^{2}$
Netherlands	$1.7\ 10^{2}$	$6.5\ 10^{2}$	$6.5\ 10^{3}$	$7.0\ 10^{2}$	$3.0\ 10^{2}$
Norway	$0.0\ 10^{0}$	$1.0\ 10^{3}$	$1.7\ 10^{3}$	$3.0\ 10^{2}$	$6.0\ 10^{2}$
Portugal	$1.5\ 10^{1}$	$3.5\ 10^{2}$	$1.0\ 10^{3}$	$3.5\ 10^{2}$	$1.3\ 10^{3}$
Spain	$1.2\ 10^{2}$	$6.5\ 10^{2}$	$8.0\ 10^{3}$	$7.5\ 10^{2}$	$3.7\ 10^{3}$
Sweden	$2.0\ 10^{1}$	$3.0\ 10^{2}$	$6.5\ 10^{2}$	$1.6\ 10^{4}$	$5.0\ 10^{1}$
United Kingdom	$3.0\ 10^{2}$	$9.0\ 10^{3}$	$6.8\ 10^{4}$	$5.0\ 10^{2}$	$3.0\ 10^{3}$
World	$3.1\ 10^{3}$	$3.4\ 10^{4}$	$2.1\ 10^{5}$	$3.7\ 10^{4}$	$4.5\ 10^{4}$

Zero catch for Atlantic NE middle and deep; Other Atlantic, Arctic Ocean; Arctic South, Other Oceans.(Atlantic NE includes Arctic South data)

Notes:

a - Hagel (2002); ICES (2001); Nielson (2000).

Table 4.14 Annual marine crustacean consumption by model compartment[a]

Nation	Annual marine crustacean consumption t y^{-1}										
	Atlantic North NE (surface 0-1000 m)	Baie de la Seine	Bay of Biscay	Belt Sea (surface 0-14 m)	Bristol Channel	Cantabrian Sea	Cap de la Hague	Celtic Sea	Central Channel NE	Central Channel SE	Channel Islands
Austria	0.0 10^{0}	0.0 10^{0}	2.3 10^{-2}		5.0 10^{-2}	7.5 10^{-2}	3.3 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	3.3 10^{-1}
Belgium	6.0 10^{1}	2.7 10^{0}	0.0 10^{0}	2.7 10^{0}	3.3 10^{0}	0.0 10^{0}	6.7 10^{0}	1.7 10^{1}	1.7 10^{0}	2.3 10^{0}	1.3 10^{1}
Denmark	3.0 10^{3}	6.7 10^{-1}	4.6 10^{-2}	5.6 10^{0}	0.0 10^{0}	0.0 10^{0}	3.3 10^{0}	0.0 10^{0}	6.7 10^{-1}	1.0 10^{0}	3.3 10^{0}
Finland	3.3 10^{1}	0.0 10^{0}	5.3 10^{-3}	2.1 10^{-1}	3.4 10^{-2}	8.8 10^{-2}	0.0 10^{0}	8.6 10^{-2}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
France	1.8 10^{3}	5.0 10^{1}	5.3 10^{1}	1.8 10^{1}	1.0 10^{2}	6.7 10^{0}	4.0 10^{2}	7.7 10^{2}	4.3 10^{1}	5.7 10^{1}	7.3 10^{2}
Germany	4.7 10^{2}	3.3 10^{0}	0.0 10^{0}	3.5 10^{1}	3.3 10^{0}	0.0 10^{0}	6.7 10^{0}	1.3 10^{1}	2.0 10^{0}	2.7 10^{0}	1.3 10^{1}
Greece	6.7 10^{0}	3.3 10^{-1}	4.2 10^{-2}	6.3 10^{-1}	0.0 10^{0}	0.0 10^{0}	1.0 10^{0}	0.0 10^{0}	3.3 10^{-1}	3.3 10^{-1}	2.0 10^{0}
Ireland	1.2 10^{3}	2.3 10^{0}	1.0 10^{-1}	2.8 10^{-1}	3.3 10^{0}	5.9 10^{0}	6.7 10^{0}	2.0 10^{2}	2.0 10^{0}	2.3 10^{0}	1.0 10^{1}
Italy	5.0 10^{2}	6.7 10^{0}	0.0 10^{0}	9.1 10^{0}	1.7 10^{1}	6.7 10^{0}	3.3 10^{1}	6.7 10^{1}	6.7 10^{0}	6.7 10^{0}	3.7 10^{1}
Netherlands	3.3 10^{2}	1.7 10^{0}	0.0 10^{0}	1.1 10^{1}	3.3 10^{0}	0.0 10^{0}	5.0 10^{0}	3.3 10^{0}	1.3 10^{0}	2.0 10^{0}	1.0 10^{1}
Norway	4.7 10^{2}	0.0 10^{0}	0.0 10^{0}		0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
Portugal	6.7 10^{1}	2.3 10^{0}	0.0 10^{0}	2.6 10^{0}	3.3 10^{0}	6.7 10^{0}	1.0 10^{1}	1.7 10^{1}	2.0 10^{0}	2.3 10^{0}	2.0 10^{1}
Spain	1.0 10^{3}	1.7 10^{1}	1.7 10^{0}	6.3 10^{0}	3.3 10^{1}	3.3 10^{2}	5.0 10^{1}	6.7 10^{1}	1.3 10^{1}	2.0 10^{1}	8.7 10^{1}
Sweden	4.3 10^{2}	0.0 10^{0}	1.7 10^{-2}	1.3 10^{1}	0.0 10^{0}	0.0 10^{0}	3.3 10^{-1}	1.7 10^{1}	0.0 10^{0}	3.3 10^{-1}	1.0 10^{0}
United Kingdom	5.3 10^{3}	5.3 10^{1}	6.7 10^{0}	6.3 10^{0}	1.0 10^{2}	0.0 10^{0}	1.4 10^{2}	6.7 10^{1}	4.7 10^{1}	6.0 10^{1}	2.3 10^{2}
World	3.1 10^{4}	1.9 10^{2}	6.7 10^{1}	1.1 10^{2}	3.3 10^{2}	3.7 10^{2}	8.2 10^{2}	1.3 10^{3}	1.7 10^{2}	2.2 10^{2}	1.4 10^{3}

Table 4.14 continued. Annual marine crustacean consumption by model compartment [a]

Nation	Annual marine crustacean consumption t y^{-1}										
	Cumbrian Waters	English Channel W	French Continental Shelf	Gulf of Cadiz	Irish Sea N	Irish Sea NE	Irish Sea NW	Irish Sea S	Irish Sea SE	Irish Sea W	Isle of Wight
Austria	0.0 10^{0}	6.7 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	3.3 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
Belgium	1.0 10^{0}	2.3 10^{1}	3.3 10^{1}	0.0 10^{0}	1.0 10^{0}	1.0 10^{0}	3.3 10^{0}	1.2 10^{1}	3.3 10^{0}	6.7 10^{0}	1.3 10^{0}
Denmark	3.3 10^{-1}	5.0 10^{0}	0.0 10^{0}	0.0 10^{0}	6.7 10^{-1}	6.7 10^{-1}	1.3 10^{0}	5.0 10^{0}	2.3 10^{0}	3.3 10^{0}	3.3 10^{-1}
Finland	0.0 10^{0}	3.3 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	3.3 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
France	1.3 10^{1}	1.1 10^{3}	2.6 10^{3}	1.7 10^{1}	2.0 10^{1}	2.0 10^{1}	5.0 10^{1}	2.0 10^{2}	6.7 10^{1}	1.1 10^{2}	2.7 10^{1}
Germany	1.0 10^{0}	2.0 10^{1}	1.7 10^{1}	0.0 10^{0}	1.3 10^{0}	1.3 10^{0}	3.3 10^{0}	1.3 10^{1}	3.3 10^{0}	6.7 10^{0}	1.3 10^{0}
Greece	0.0 10^{0}	3.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	6.7 10^{-1}	2.0 10^{0}	1.0 10^{0}	1.0 10^{0}	0.0 10^{0}
Ireland	1.7 10^{1}	1.3 10^{1}	0.0 10^{0}	0.0 10^{0}	2.0 10^{1}	2.3 10^{1}	5.7 10^{1}	2.2 10^{2}	6.7 10^{1}	1.3 10^{2}	1.3 10^{0}
Italy	6.7 10^{0}	5.3 10^{1}	3.3 10^{1}	1.7 10^{1}	1.0 10^{1}	1.0 10^{1}	2.3 10^{1}	9.7 10^{1}	3.3 10^{1}	5.7 10^{1}	3.3 10^{0}
Netherlands	6.7 10^{-1}	1.3 10^{1}	1.7 10^{1}	0.0 10^{0}	1.0 10^{0}	1.0 10^{0}	3.3 10^{0}	1.0 10^{1}	3.3 10^{0}	5.7 10^{0}	1.3 10^{0}
Norway	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
Portugal	1.0 10^{0}	2.7 10^{1}	3.3 10^{1}	2.3 10^{2}	1.3 10^{0}	1.3 10^{0}	3.3 10^{0}	1.3 10^{1}	3.3 10^{0}	6.7 10^{0}	1.0 10^{0}
Spain	1.0 10^{1}	1.3 10^{2}	6.7 10^{1}	1.1 10^{3}	1.3 10^{1}	1.7 10^{1}	3.7 10^{1}	1.4 10^{2}	4.3 10^{1}	8.0 10^{1}	1.0 10^{1}
Sweden	1.0 10^{0}	1.0 10^{0}	0.0 10^{0}	0.0 10^{0}	1.3 10^{0}	1.3 10^{0}	4.7 10^{0}	1.4 10^{1}	3.3 10^{0}	1.0 10^{1}	0.0 10^{0}
United Kingdom	2.7 10^{1}	3.6 10^{2}	3.3 10^{1}	0.0 10^{0}	3.3 10^{1}	3.7 10^{1}	9.0 10^{1}	3.4 10^{2}	1.0 10^{2}	2.0 10^{2}	2.7 10^{1}
World	1.0 10^{2}	2.2 10^{3}	3.0 10^{3}	1.4 10^{3}	1.3 10^{2}	1.4 10^{2}	3.5 10^{2}	1.4 10^{3}	4.3 10^{2}	8.0 10^{2}	1.0 10^{2}

Table 4.14 continued. Annual marine crustacean consumption by model compartment [a]

Nation	Annual marine crustacean consumption t y^{-1}										
	Kara and Barents sea	Kattegat (bottom 20-120 m)	Kattegat (surface 0-20 m)	Liverpool and Morecambe Bays	Lyme Bay	Mediterranean	North Sea Central	North Sea E	North Sea N	North Sea SE	North Sea SW
Austria	0.0 10^{0}	0.0 10^{0}	6.7 10^{0}	0.0 10^{0}	0.0 10^{0}		3.3 10^{1}	3.3 10^{0}	1.3 10^{1}	3.3 10^{0}	3.3 10^{0}
Belgium	0.0 10^{0}	0.0 10^{0}	6.7 10^{0}	2.0 10^{0}	3.3 10^{0}	1.7 10^{1}	2.7 10^{2}	3.3 10^{1}	3.3 10^{1}	2.0 10^{2}	2.0 10^{2}
Denmark	1.7 10^{2}	0.0 10^{0}	1.7 10^{2}	6.7 10^{-1}	2.3 10^{0}	1.1 10^{0}	1.3 10^{2}	1.7 10^{1}	1.3 10^{2}	3.3 10^{1}	3.3 10^{1}
Finland	1.0 10^{2}	0.0 10^{0}	5.0 10^{1}	0.0 10^{0}	0.0 10^{0}		1.7 10^{1}	0.0 10^{0}	3.3 10^{1}	0.0 10^{0}	0.0 10^{0}
France	1.7 10^{1}	0.0 10^{0}	1.3 10^{2}	3.0 10^{1}	2.2 10^{2}	2.7 10^{2}	6.0 10^{2}	6.7 10^{1}	4.3 10^{2}	2.7 10^{2}	2.7 10^{2}
Germany	3.3 10^{1}	0.0 10^{0}	6.7 10^{1}	2.0 10^{0}	3.3 10^{0}	4.1 10^{1}	3.3 10^{3}	3.7 10^{2}	1.0 10^{2}	1.3 10^{2}	1.3 10^{2}
Greece	0.0 10^{0}	0.0 10^{0}	3.3 10^{0}	3.3 10^{-1}	6.7 10^{-1}	1.1 10^{3}	1.7 10^{1}	0.0 10^{0}	2.0 10^{1}	3.3 10^{0}	3.3 10^{0}
Ireland	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	3.7 10^{1}	3.3 10^{0}	0.0 10^{0}	3.3 10^{1}	3.3 10^{0}	3.3 10^{1}	1.0 10^{1}	1.0 10^{1}
Italy	0.0 10^{0}	0.0 10^{0}	1.3 10^{2}	1.7 10^{1}	1.0 10^{1}	1.1 10^{4}	2.0 10^{2}	3.3 10^{1}	2.0 10^{2}	6.7 10^{1}	6.7 10^{1}
Netherlands	1.7 10^{1}	0.0 10^{0}	6.7 10^{1}	1.3 10^{0}	3.3 10^{0}	1.3 10^{1}	7.0 10^{2}	6.7 10^{1}	6.7 10^{1}	2.8 10^{2}	2.8 10^{2}
Norway	1.9 10^{3}	0.0 10^{0}	8.0 10^{2}	0.0 10^{0}	0.0 10^{0}		0.0 10^{0}	0.0 10^{0}	6.7 10^{2}	0.0 10^{0}	0.0 10^{0}
Portugal	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	1.7 10^{0}	6.7 10^{0}	9.1 10^{1}	3.3 10^{1}	3.3 10^{0}	3.3 10^{1}	3.3 10^{1}	3.3 10^{1}
Spain	0.0 10^{0}	0.0 10^{0}	3.3 10^{1}	2.3 10^{1}	2.7 10^{1}	2.8 10^{3}	3.0 10^{2}	3.3 10^{1}	3.7 10^{2}	1.0 10^{2}	1.0 10^{2}
Sweden	2.0 10^{2}	0.0 10^{0}	9.7 10^{2}	2.0 10^{0}	3.3 10^{-1}		1.0 10^{2}	6.7 10^{0}	2.3 10^{2}	0.0 10^{0}	0.0 10^{0}
United Kingdom	3.7 10^{2}	0.0 10^{0}	2.7 10^{2}	5.7 10^{1}	7.0 10^{1}	2.8 10^{1}	8.7 10^{2}	1.0 10^{2}	1.3 10^{3}	1.7 10^{2}	1.7 10^{2}
World	4.3 10^{3}	0.0 10^{0}	3.7 10^{3}	2.2 10^{2}	4.3 10^{2}	1.5 10^{4}	9.0 10^{3}	1.0 10^{3}	5.3 10^{3}	1.9 10^{3}	1.9 10^{3}

Table 4.14 continued. Annual marine crustacean consumption by model compartment [a]

Nation	Annual marine crustacean consumption t y^{-1}						
	Norwegian Waters	Portuguese Continental Shelf	Sam's Beach	Scottish Waters E	Scottish Waters W	Skagerrak	Spitzbergen
Austria	2.1 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	6.7 10^{0}	3.1 10^{-2}
Belgium	0.0 10^{0}	0.0 10^{0}	1.3 10^{0}	3.3 10^{0}	6.7 10^{1}	6.7 10^{0}	3.3 10^{0}
Denmark	6.7 10^{1}	0.0 10^{0}	3.3 10^{-1}	3.3 10^{1}	3.3 10^{1}	1.7 10^{2}	4.3 10^{2}
Finland	6.7 10^{1}	0.0 10^{0}	0.0 10^{0}	3.3 10^{0}	0.0 10^{0}	5.0 10^{1}	2.3 10^{2}
France	3.3 10^{0}	1.7 10^{1}	3.0 10^{1}	6.7 10^{1}	6.3 10^{2}	1.3 10^{2}	2.7 10^{1}
Germany	2.7 10^{1}	0.0 10^{0}	1.3 10^{0}	1.7 10^{1}	6.7 10^{1}	6.7 10^{1}	6.7 10^{1}
Greece	3.3 10^{-2}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	1.7 10^{1}	3.3 10^{0}	0.0 10^{0}
Ireland	5.2 10^{-3}	0.0 10^{0}	1.3 10^{0}	3.3 10^{0}	4.7 10^{2}	0.0 10^{0}	0.0 10^{0}
Italy	1.4 10^{-1}	1.7 10^{1}	3.3 10^{0}	3.3 10^{1}	3.0 10^{2}	1.3 10^{2}	3.3 10^{0}
Netherlands	3.5 10^{0}	0.0 10^{0}	1.0 10^{0}	3.3 10^{0}	3.3 10^{1}	6.7 10^{1}	3.3 10^{1}
Norway	1.0 10^{3}	0.0 10^{0}	0.0 10^{0}	1.0 10^{2}	0.0 10^{0}	8.0 10^{2}	1.0 10^{3}
Portugal	3.6 10^{-3}	2.7 10^{2}	1.3 10^{0}	3.3 10^{0}	5.0 10^{1}	0.0 10^{0}	3.3 10^{1}
Spain	0.0 10^{0}	1.1 10^{3}	1.0 10^{1}	6.7 10^{1}	5.0 10^{2}	3.3 10^{1}	1.3 10^{2}
Sweden	1.3 10^{2}	0.0 10^{0}	0.0 10^{0}	3.3 10^{1}	3.3 10^{1}	9.7 10^{2}	5.0 10^{2}
United Kingdom	1.7 10^{2}	0.0 10^{0}	3.0 10^{1}	2.0 10^{2}	1.4 10^{3}	2.7 10^{2}	1.0 10^{3}
World	1.6 10^{3}	1.5 10^{3}	1.1 10^{2}	8.0 10^{2}	5.0 10^{3}	3.7 10^{3}	7.3 10^{3}

Zero catch for Atlantic NE middle and deep; Other Atlantic, Arctic Ocean; Arctic South, Baltic Sea W,E, surface and bottom) Belt sea bottom, Bothonian Bay, Bothonian Sea, Gulf of Finland, Gulf of Riga, Kattegat Bottom, Other Oceans, (Atlantic NE includes Arctic South data).

Notes:

a - Hagel (2002); ICES (2001); Nielson (2000).

Table 4.15 Annual marine mollusc consumption by model compartment [a]

Nation	Annual marine mollusc consumption t y^{-1}										
	Atlantic North N.E. (surface 0-1000 m)	Baie de la Seine	Bay of Biscay	Belt Sea (surface 0-14 m)	Bristol Channel	Cantabrian Sea	Celtic Sea	Central Channel SE	Channel Islands	Cumbrian Waters	English Channel W
Austria	4.3 10^{-1}	5.0 10^{-1}	5.2 10^{-3}		0.0 10^{0}	5.0 10^{-2}	0.0 10^{0}	0.0 10^{0}	1.2 10^{0}	0.0 10^{0}	8.3 10^{-1}
Belgium	1.9 10^{1}	3.8 10^{1}	0.0 10^{0}	2.3 10^{0}	2.7 10^{1}	1.7 10^{0}	1.0 10^{1}	5.0 10^{0}	6.3 10^{1}	3.3 10^{0}	4.2 10^{1}
Denmark	1.1 10^{2}	1.3 10^{0}	0.0 10^{0}	9.2 10^{2}	1.7 10^{0}	0.0 10^{0}	0.0 10^{0}	1.7 10^{-1}	3.3 10^{0}	3.3 10^{-1}	3.3 10^{0}
Finland	2.3 10^{-1}	1.7 10^{-1}	2.2 10^{-3}	1.8 10^{-1}	0.0 10^{0}	4.1 10^{-2}	0.0 10^{0}	0.0 10^{0}	5.0 10^{-1}	0.0 10^{0}	3.3 10^{-1}
France	8.2 10^{2}	5.8 10^{2}	5.0 10^{0}	1.3 10^{1}	1.8 10^{2}	2.7 10^{1}	2.8 10^{1}	7.2 10^{1}	1.4 10^{3}	5.5 10^{1}	9.0 10^{2}
Germany	2.8 10^{1}	1.5 10^{1}	0.0 10^{0}	2.9 10^{1}	3.3 10^{0}	5.0 10^{0}	6.7 10^{0}	1.7 10^{0}	3.5 10^{1}	1.7 10^{0}	2.3 10^{1}
Greece	3.5 10^{0}	1.7 10^{0}	7.2 10^{-2}	4.8 10^{-1}	1.7 10^{0}	3.3 10^{0}	1.7 10^{0}	1.7 10^{-1}	3.3 10^{0}	3.3 10^{-1}	3.3 10^{0}
Ireland	4.0 10^{2}	8.3 10^{-1}	8.4 10^{-4}	2.3 10^{-1}	3.3 10^{0}	7.0 10^{0}	5.0 10^{1}	1.7 10^{-1}	5.0 10^{0}	2.8 10^{1}	3.3 10^{0}
Italy	9.7 10^{1}	4.2 10^{1}	1.7 10^{0}	6.9 10^{0}	4.8 10^{1}	5.8 10^{1}	3.3 10^{1}	5.0 10^{0}	1.2 10^{2}	1.0 10^{1}	8.3 10^{1}
Netherlands	3.4 10^{1}	1.0 10^{1}	0.0 10^{0}	8.6 10^{0}	2.5 10^{1}	0.0 10^{0}	6.7 10^{0}	8.3 10^{-1}	3.8 10^{1}	3.3 10^{0}	2.5 10^{1}
Norway	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}		0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}
Portugal	2.8 10^{1}	3.3 10^{0}	0.0 10^{0}	2.0 10^{0}	3.3 10^{0}	3.3 10^{1}	3.3 10^{0}	3.3 10^{-1}	8.3 10^{0}	6.7 10^{-1}	6.7 10^{0}
Spain	2.4 10^{2}	1.2 10^{2}	1.0 10^{1}	4.8 10^{0}	8.3 10^{1}	5.2 10^{2}	9.3 10^{1}	1.5 10^{1}	3.2 10^{2}	1.5 10^{1}	2.0 10^{2}
Sweden	2.1 10^{0}	5.0 10^{-1}	5.6 10^{-3}	5.9 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	1.7 10^{0}	3.3 10^{-1}	5.0 10^{-1}
United Kingdom	2.6 10^{2}	1.5 10^{2}	3.2 10^{-2}	4.8 10^{0}	6.5 10^{2}	1.7 10^{0}	1.0 10^{2}	1.8 10^{1}	7.3 10^{2}	7.2 10^{1}	4.8 10^{2}
World	5.8 10^{3}	1.0 10^{3}	1.8 10^{1}	1.0 10^{3}	1.0 10^{3}	7.0 10^{2}	6.3 10^{2}	1.2 10^{2}	2.8 10^{3}	2.1 10^{2}	1.8 10^{3}

Table 4.15 continued - Annual marine mollusc consumption by model compartment [a]

Nation	Annual marine mollusc consumption $t\ y^{-1}$										
	French Continental Shelf	Gulf of Cadiz	Irish Sea N	Irish Sea NE	Irish Sea NW	Irish Sea S	Irish Sea SE	Irish Sea W	Isle of Wight	Kara and Barents sea	Kattegat (surface 0-20 m)
Austria	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.7\ 10^{-1}$	$3.3\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$8.3\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Belgium	$5.0\ 10^{1}$	$5.0\ 10^{0}$	$5.0\ 10^{0}$	$6.7\ 10^{0}$	$8.3\ 10^{0}$	$1.8\ 10^{1}$	$3.3\ 10^{0}$	$6.7\ 10^{0}$	$4.7\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Denmark	$1.7\ 10^{0}$	$1.7\ 10^{0}$	$5.0\ 10^{-1}$	$6.7\ 10^{-1}$	$6.7\ 10^{-1}$	$1.2\ 10^{0}$	$3.3\ 10^{-1}$	$6.7\ 10^{-1}$	$1.7\ 10^{0}$	$5.0\ 10^{1}$	$6.7\ 10^{0}$
Finland	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.7\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.3\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
France	$1.2\ 10^{3}$	$8.3\ 10^{1}$	$8.7\ 10^{1}$	$1.6\ 10^{2}$	$2.0\ 10^{2}$	$4.1\ 10^{2}$	$9.0\ 10^{1}$	$1.5\ 10^{2}$	$7.1\ 10^{2}$	$1.7\ 10^{1}$	$0.0\ 10^{0}$
Germany	$3.3\ 10^{1}$	$1.7\ 10^{1}$	$1.7\ 10^{0}$	$3.3\ 10^{0}$	$5.0\ 10^{0}$	$1.0\ 10^{1}$	$1.7\ 10^{0}$	$3.3\ 10^{0}$	$1.7\ 10^{1}$	$1.7\ 10^{1}$	$3.3\ 10^{0}$
Greece	$3.3\ 10^{0}$	$1.7\ 10^{1}$	$3.3\ 10^{-1}$	$6.7\ 10^{-1}$	$6.7\ 10^{-1}$	$6.7\ 10^{-1}$	$3.3\ 10^{-1}$	$6.7\ 10^{-1}$	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Ireland	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$4.5\ 10^{1}$	$8.5\ 10^{1}$	$1.1\ 10^{2}$	$2.1\ 10^{2}$	$4.7\ 10^{1}$	$7.7\ 10^{1}$	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Italy	$8.3\ 10^{1}$	$1.7\ 10^{2}$	$1.5\ 10^{1}$	$2.8\ 10^{1}$	$3.7\ 10^{1}$	$7.3\ 10^{1}$	$1.7\ 10^{1}$	$2.7\ 10^{1}$	$5.2\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Netherlands	$8.3\ 10^{0}$	$1.7\ 10^{0}$	$5.0\ 10^{0}$	$8.3\ 10^{0}$	$1.0\ 10^{1}$	$2.0\ 10^{1}$	$5.0\ 10^{0}$	$6.7\ 10^{0}$	$1.2\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Norway	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$4.2\ 10^{2}$	$0.0\ 10^{0}$
Portugal	$1.7\ 10^{1}$	$1.8\ 10^{3}$	$6.7\ 10^{-1}$	$1.3\ 10^{0}$	$1.3\ 10^{0}$	$3.3\ 10^{0}$	$8.3\ 10^{-1}$	$1.3\ 10^{0}$	$3.3\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Spain	$3.2\ 10^{2}$	$1.8\ 10^{3}$	$2.2\ 10^{1}$	$4.2\ 10^{1}$	$5.3\ 10^{1}$	$1.1\ 10^{2}$	$2.3\ 10^{1}$	$3.8\ 10^{1}$	$1.5\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Sweden	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$1.7\ 10^{-1}$	$6.7\ 10^{-1}$	$5.0\ 10^{-1}$	$1.0\ 10^{0}$	$1.7\ 10^{-1}$	$6.7\ 10^{-1}$	$5.0\ 10^{-1}$	$0.0\ 10^{0}$	$3.3\ 10^{0}$
United Kingdom	$1.7\ 10^{1}$	$8.3\ 10^{0}$	$1.2\ 10^{2}$	$2.2\ 10^{2}$	$2.7\ 10^{2}$	$5.4\ 10^{2}$	$1.2\ 10^{2}$	$2.0\ 10^{2}$	$1.8\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
World	$1.8\ 10^{3}$	$4.2\ 10^{3}$	$3.2\ 10^{2}$	$6.0\ 10^{2}$	$7.5\ 10^{2}$	$1.5\ 10^{3}$	$3.3\ 10^{2}$	$5.5\ 10^{2}$	$1.2\ 10^{3}$	$1.5\ 10^{4}$	$1.7\ 10^{1}$

Table 4.15 continued - Annual marine mollusc consumption by model compartment [a]

Nation	Annual marine mollusc consumption t y^{-1}										
	Liverpool and Morecambe Bays	Lyme Bay	Mediterranean	North Sea Central	North Sea E	North Sea N	North Sea SE	North Sea SW	Norwegian Waters	Portuguese Continental Shelf	Sam's Beach
Austria	$1.7\ 10^{-1}$	$6.7\ 10^{-1}$		$6.7\ 10^{1}$	$5.0\ 10^{1}$	$0.0\ 10^{0}$	$1.0\ 10^{1}$	$1.0\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$5.0\ 10^{-1}$
Belgium	$1.0\ 10^{1}$	$3.5\ 10^{1}$	$3.9\ 10^{1}$	$5.0\ 10^{1}$	$5.0\ 10^{1}$	$1.7\ 10^{1}$	$1.5\ 10^{3}$	$1.5\ 10^{3}$	$6.3\ 10^{-2}$	$5.0\ 10^{0}$	$3.5\ 10^{1}$
Denmark	$5.0\ 10^{-1}$	$1.7\ 10^{0}$	$1.4\ 10^{0}$	$4.7\ 10^{3}$	$3.7\ 10^{3}$	$3.3\ 10^{0}$	$1.7\ 10^{1}$	$1.7\ 10^{1}$	$1.0\ 10^{1}$	$1.7\ 10^{0}$	$1.3\ 10^{0}$
Finland	$1.7\ 10^{-1}$	$3.3\ 10^{-1}$		$3.3\ 10^{1}$	$2.3\ 10^{1}$	$0.0\ 10^{0}$	$1.7\ 10^{0}$	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.7\ 10^{-1}$
France	$2.4\ 10^{2}$	$7.8\ 10^{2}$	$3.1\ 10^{3}$	$9.2\ 10^{2}$	$7.3\ 10^{2}$	$1.3\ 10^{2}$	$1.6\ 10^{3}$	$1.6\ 10^{3}$	$5.0\ 10^{0}$	$8.3\ 10^{1}$	$5.3\ 10^{2}$
Germany	$5.0\ 10^{0}$	$1.8\ 10^{1}$	$2.3\ 10^{2}$	$6.3\ 10^{3}$	$5.0\ 10^{3}$	$3.3\ 10^{0}$	$2.2\ 10^{2}$	$2.2\ 10^{2}$	$1.7\ 10^{0}$	$1.7\ 10^{1}$	$1.3\ 10^{1}$
Greece	$5.0\ 10^{-1}$	$1.7\ 10^{0}$	$1.4\ 10^{3}$	$1.7\ 10^{1}$	$1.0\ 10^{1}$	$0.0\ 10^{0}$	$1.0\ 10^{1}$	$1.0\ 10^{1}$	$0.0\ 10^{0}$	$1.7\ 10^{1}$	$1.7\ 10^{0}$
Ireland	$1.3\ 10^{2}$	$1.7\ 10^{0}$	$3.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.3\ 10^{0}$	$6.7\ 10^{0}$	$6.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$8.3\ 10^{-1}$
Italy	$4.3\ 10^{1}$	$6.7\ 10^{1}$	$2.8\ 10^{4}$	$2.0\ 10^{2}$	$1.7\ 10^{2}$	$3.3\ 10^{1}$	$5.8\ 10^{2}$	$5.8\ 10^{2}$	$2.1\ 10^{-2}$	$1.7\ 10^{2}$	$3.8\ 10^{1}$
Netherlands	$1.3\ 10^{1}$	$2.0\ 10^{1}$	$2.8\ 10^{1}$	$1.1\ 10^{3}$	$8.3\ 10^{2}$	$1.7\ 10^{1}$	$1.6\ 10^{3}$	$1.6\ 10^{3}$	$2.1\ 10^{-2}$	$1.7\ 10^{0}$	$8.3\ 10^{0}$
Norway	$0.0\ 10^{0}$	$0.0\ 10^{0}$		$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.3\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.0\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Portugal	$1.7\ 10^{0}$	$5.0\ 10^{0}$	$8.0\ 10^{1}$	$3.3\ 10^{1}$	$3.3\ 10^{1}$	$3.3\ 10^{0}$	$1.3\ 10^{2}$	$1.3\ 10^{2}$	$0.0\ 10^{0}$	$1.8\ 10^{3}$	$3.3\ 10^{0}$
Spain	$6.2\ 10^{1}$	$1.8\ 10^{2}$	$3.5\ 10^{3}$	$1.0\ 10^{2}$	$8.3\ 10^{1}$	$6.7\ 10^{1}$	$6.7\ 10^{2}$	$6.7\ 10^{2}$	$0.0\ 10^{0}$	$1.8\ 10^{3}$	$1.1\ 10^{2}$
Sweden	$3.3\ 10^{-1}$	$6.7\ 10^{-1}$		$6.7\ 10^{1}$	$5.0\ 10^{1}$	$0.0\ 10^{0}$	$5.0\ 10^{0}$	$5.0\ 10^{0}$	$1.1\ 10^{0}$	$0.0\ 10^{0}$	$5.0\ 10^{-1}$
United Kingdom	$3.2\ 10^{2}$	$4.0\ 10^{2}$	$1.3\ 10^{1}$	$3.0\ 10^{2}$	$2.3\ 10^{2}$	$5.7\ 10^{2}$	$1.2\ 10^{3}$	$1.2\ 10^{3}$	$2.1\ 10^{-2}$	$8.3\ 10^{0}$	$1.3\ 10^{2}$
World	$8.8\ 10^{2}$	$1.6\ 10^{3}$	$3.6\ 10^{4}$	$1.6\ 10^{4}$	$1.3\ 10^{4}$	$8.5\ 10^{2}$	$8.2\ 10^{3}$	$8.2\ 10^{3}$	$1.2\ 10^{2}$	$4.2\ 10^{3}$	$9.0\ 10^{2}$

Table 4.15 continued - Annual marine mollusc consumption by model compartment [a]

Nation	Annual marine mollusc consumption t y^{-1}			
	Scottish Waters E	Scottish Waters W	Skagerrak	Spitzbergen
Austria	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Belgium	$8.3\ 10^{0}$	$1.7\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Denmark	$1.7\ 10^{0}$	$1.7\ 10^{0}$	$6.7\ 10^{0}$	$2.8\ 10^{1}$
Finland	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
France	$5.0\ 10^{1}$	$3.3\ 10^{2}$	$1.7\ 10^{0}$	$1.3\ 10^{1}$
Germany	$1.7\ 10^{0}$	$8.3\ 10^{0}$	$1.7\ 10^{0}$	$5.0\ 10^{0}$
Greece	$0.0\ 10^{0}$	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Ireland	$1.7\ 10^{0}$	$8.3\ 10^{1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Italy	$1.7\ 10^{1}$	$8.3\ 10^{1}$	$0.0\ 10^{0}$	$6.1\ 10^{-2}$
Netherlands	$1.7\ 10^{1}$	$3.3\ 10^{1}$	$0.0\ 10^{0}$	$6.1\ 10^{-2}$
Norway	$1.7\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$2.8\ 10^{2}$
Portugal	$1.7\ 10^{0}$	$5.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Spain	$3.3\ 10^{1}$	$1.3\ 10^{2}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Sweden	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.3\ 10^{0}$	$3.3\ 10^{0}$
United Kingdom	$2.5\ 10^{2}$	$9.2\ 10^{2}$	$0.0\ 10^{0}$	$6.1\ 10^{-2}$
World	$3.8\ 10^{2}$	$1.7\ 10^{3}$	$1.7\ 10^{1}$	$3.3\ 10^{2}$

Zero catch for Atlantic NE middle and deep; Other Atlantic, Arctic Ocean; Arctic South, Baltic Sea W,E, surface and bottom) Belt sea bottom, Bothonian Bay, Bothonian Sea, Cap de la Hague, Central Channel NE, Gulf of Finland, Gulf of Riga, Kattegat Bottom, Other Oceans, (Atlantic NE includes Arctic South data)

Notes:

a - Hagel (2002); ICES (2001); Nielson (2000).

Table 4.16 Edible fraction of seafood [a]

Food	Fraction edible
Fish	0.5
Crustaceans	0.35
Mollusc	0.15
Seaweed	0.2

Notes:

a - Simmonds et al (1995) (Seaweed assumed 2 x value in this reference).

Table 4.17 Individual occupancy of beaches [a]

Region /critical group	Occupancy h y^{-1}
All age groups and regions	2000

Notes:

a - Jones (2002).

Table 4.18 Collective occupancy of beaches [a]

Region /critical group	Occupancy man h y^{-1} m^{-1}
Arctic Regions	5
Scandinavia (Norway Iceland etc)	10
Northern Europe	50
Mediterranean	75

Notes:

a - Simmonds et al (1995)

Table 4.19 – Coastline lengths for Northern Europe [a]

Nation	Coastline length (m)										
	Arctic Ocean	Arctic south	Baie de la Seine	Baltic Sea E (surface 0-53 m)	Baltic Sea W (surface 0-49 m)	Belt Sea (surface 0-14 m)	Bothnian Bay	Bothnian Sea	Bristol Channel	Cantabrian Sea	Cap de la Hague
Austria	-	-	-	-	-	-	-	-	-	-	-
Belgium	-	-	-	-	-	-	-	-	-	-	-
Denmark	-	-	-	8.700 10^{4}	8.700 10^{4}	6.350 10^{5}	-	-	-	-	-
Finland	-	-	-	2.500 10^{5}	-	-	5.300 10^{5}	1.400 10^{6}	-	-	-
France	-	-	3.200 10^{5}	-	-	-	-	-	-	-	2.000 10^{4}
Germany	-	-	-	2.500 10^{5}	-	3.700 10^{5}	-	-	-	-	-
Greece	-	-	-	-	-	-	-	-	-	-	-
Ireland	-	5.700 10^{5}	-	-	-	-	-	-	-	-	-
Italy	-	-	-	-	-	-	-	-	-	-	-
Netherlands	-	-	-	-	-	-	-	-	-	-	-
Norway	-	-	-	-	-	-	-	-	-	-	-
Portugal	-	-	-	-	-	-	-	-	-	-	-
Spain	-	-	-	-	-	-	-	-	-	8.500 10^{5}	-
Sweden	-	-	-	-	1.259 10^{6}	2.500 10^{5}	4.900 10^{5}	7.700 10^{5}	-	-	-
United Kingdom	-	-	-	-	-	-	-	-	6.272 10^{5}	-	-
World	1.600 10^{6}	1.380 10^{7}	3.200 10^{5}	2.066 10^{6}	1.346 10^{6}	1.260 10^{6}	1.020 10^{6}	2.170 10^{6}	6.272 10^{5}	8.500 10^{5}	2.000 10^{4}

Table 4.19 continued – Coastline lengths for Northern Europe [a]

Nation	Coastline length (m)										
	Celtic Sea	Central Channel SE	Channel Islands	Cumbrian Waters	English Channel W	French Continental Shelf	Gulf of Cadiz	Gulf of Finland	Gulf of Riga	Irish Sea N	Irish Sea NE
Austria	-	-	-	-	-	-	-	-	-	-	-
Belgium	-	-	-	-	-	-	-	-	-	-	-
Denmark	-	-	-	-	-	-	-	-	-	-	-
Finland	-	-	-	-	-	-	-	$3.900\ 10^5$	-	-	-
France	$1.500\ 10^5$	$3.000\ 10^4$	$3.800\ 10^5$	-	$1.000\ 10^5$	$1.020\ 10^6$	-	-	-	-	-
Germany	-	-	-	-	-	-	-	-	-	-	-
Greece	-	-	-	-	-	-	-	-	-	-	-
Ireland	$5.100\ 10^5$	-	-	-	-	-	-	-	-	-	-
Italy	-	-	-	-	-	-	-	-	-	-	-
Netherlands	-	-	-	-	-	-	-	-	-	-	-
Norway	-	-	-	-	-	-	-	-	-	-	-
Portugal	-	-	-	-	-	-	$2.000\ 10^5$	-	-	-	-
Spain	-	-	-	-	-	$1.500\ 10^5$	$2.800\ 10^5$	-	-	-	-
Sweden	-	-	-	-	-	-	-	-	-	-	-
United Kingdom	$1.110\ 10^5$	-	$5.000\ 10^4$	$7.650\ 10^4$	$1.600\ 10^5$	-	-	-	-	$1.000\ 10^5$	$2.250\ 10^5$
World	$7.710\ 10^5$	$3.000\ 10^4$	$4.300\ 10^5$	$7.650\ 10^4$	$2.600\ 10^5$	$1.170\ 10^6$	$4.800\ 10^5$	$1.108\ 10^6$	$7.110\ 10^5$	$1.000\ 10^5$	$2.250\ 10^5$

Table 4.19 continued – Coastline lengths for Northern Europe [a]

Nation	Coastline length (m)										
	Irish Sea NW	Irish Sea S	Irish Sea SE	Irish Sea W	Isle of Wight	Kara and Barents sea	Kattegat (surface 0-20 m)	Liverpool and Morecambe Bays	Lyme Bay	Mediterranean Sea	North Sea Central
Austria	-	-	-	-	-	-	-	-	-	-	-
Belgium	-	-	-	-	-	-	-	-	-	-	-
Denmark	-	-	-	-	-	-	3.500 10^{5}	-	-	-	-
Finland	-	-	-	-	-	-	-	-	-	-	-
France	-	-	-	-	-	-	-	-	-	5.100 10^{5}	-
Germany	-	-	-	-	-	-	-	-	-	-	-
Greece	-	-	-	-	-	-	-	-	-	3.050 10^{6}	-
Ireland	-	2.130 10^{5}	-	1.650 10^{5}	-	-	-	-	-	-	-
Italy	-	-	-	-	-	-	-	-	-	4.540 10^{6}	-
Netherlands	-	-	-	-	-	-	-	-	-	-	-
Norway	-	-	-	-	-	1.660 10^{6}	-	-	-	-	-
Portugal	-	-	-	-	-	-	-	-	-	-	-
Spain	-	-	-	-	-	-	-	-	-	1.890 10^{6}	-
Sweden	-	-	-	-	-	-	2.500 10^{5}	-	-	-	-
United Kingdom	2.350 10^{5}	2.210 10^{5}	1.200 10^{5}	1.340 10^{5}	3.500 10^{5}	-	-	2.100 10^{5}	2.500 10^{5}	-	8.000 10^{5}
World	2.350 10^{5}	4.340 10^{5}	1.200 10^{5}	2.990 10^{5}	3.500 10^{5}	1.240 10^{7}	6.000 10^{5}	2.100 10^{5}	2.500 10^{5}	1.858 10^{7}	8.000 10^{5}

Table 4.19 continued – Coastline lengths for Northern Europe [a]

Nation	Coastline length (m)										
	North Sea E	North Sea N	North Sea SE	North Sea SW	Norwegian Waters	Other Atlantic	Other Oceans	Portuguese Continental Shelf	Sam's Beach	Scottish Waters E	Scottish Waters W
Austria	-	-	-	-	-	-	-	-	-	-	-
Belgium	-	-	6.300 10^{4}	-	-	-	-	-	-	-	-
Denmark	3.000 10^{5}	-	-	-	-	-	-	-	-	-	-
Finland	-	-	-	-	-	-	-	-	-	-	-
France	-	-	4.000 10^{4}	-	-	-	-	-	2.200 10^{5}	-	-
Germany	3.700 10^{5}	-	-	-	-	-	-	-	-	-	-
Greece	-	-	-	-	-	-	-	-	-	-	-
Ireland	-	-	-	-	-	-	-	-	-	-	3.700 10^{5}
Italy	-	-	-	-	-	-	-	-	-	-	-
Netherlands	-	-	3.700 10^{5}	-	-	-	-	-	-	-	-
Norway	-	6.100 10^{5}	-	-	1.800 10^{6}	-	-	-	-	-	-
Portugal	-	-	-	-	-	-	-	1.170 10^{6}	-	-	-
Spain	-	-	-	-	-	-	-	2.200 10^{5}	-	-	-
Sweden	-	-	-	-	-	-	-	-	-	-	-
United Kingdom	-	2.500 10^{5}	7.650 10^{4}	4.600 10^{5}	-	-	-	-	-	4.200 10^{5}	1.390 10^{6}
World	6.700 10^{5}	8.600 10^{5}	5.500 10^{5}	4.600 10^{5}	1.800 10^{6}	1.900 10^{7}	2.377 10^{8}	1.390 10^{6}	2.200 10^{5}	4.200 10^{5}	1.760 10^{6}

Table 4.19 continued – Coastline lengths for Northern Europe [a]

	Coastline length (m)	
Nation	Skagerrak	Spitzbergen
Austria	-	-
Belgium	-	-
Denmark	1.700 10^{5}	-
Finland	-	-
France	-	-
Germany	-	-
Greece	-	-
Ireland	-	-
Italy	-	-
Netherlands	-	-
Norway	3.050 10^{5}	1.350 10^{6}
Portugal	-	-
Spain	-	-
Sweden	1.300 10^{5}	-
United Kingdom	-	-
World	6.050 10^{5}	1.350 10^{6}

No coastline for Atlantic NE surface, middle and deep; Baltic Sea W,E, surface, Belt sea bottom, Central Channel NE, Kattegat Bottom, Bay of Biscay

Notes:

a - Simmonds et al (2002)

4.8 Figures

Figure 4.1 – Transport and transformation of pollutants in rivers by physical, chemical and biological pathways

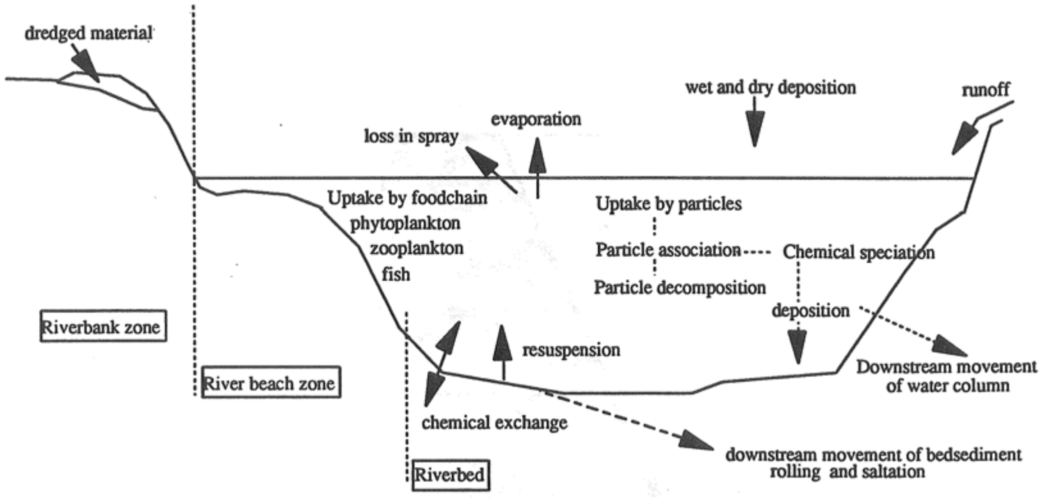


Figure 4.2 Interface between the local marine compartment and the marine model

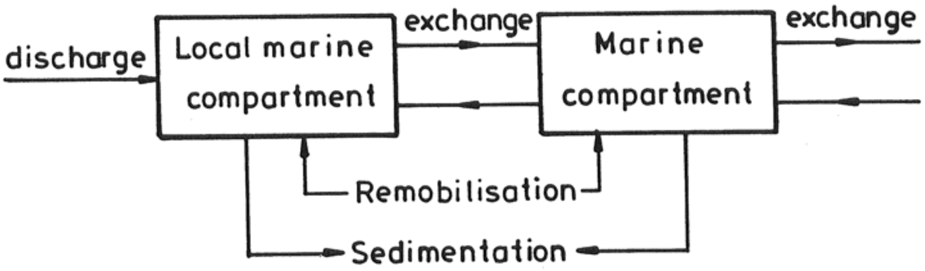
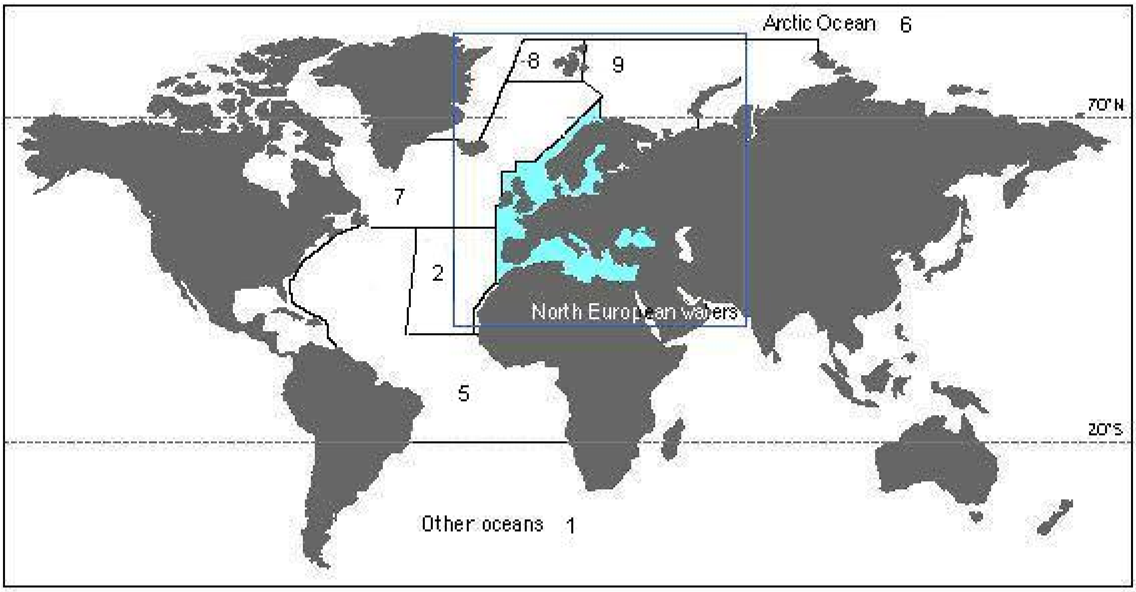
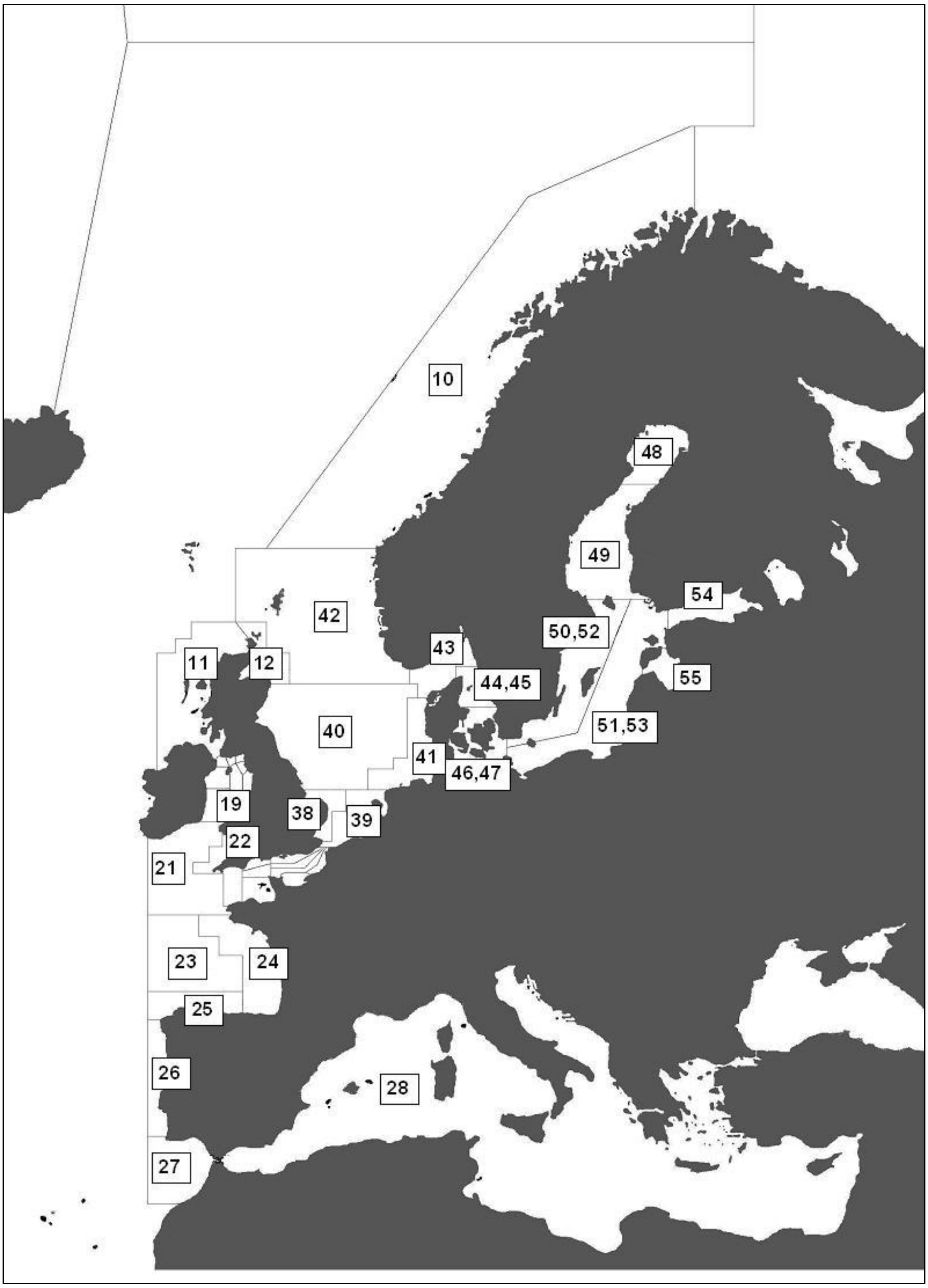


Figure 4.3 World marine compartments included in the marine dispersion model as modified for PC-CREAM 08



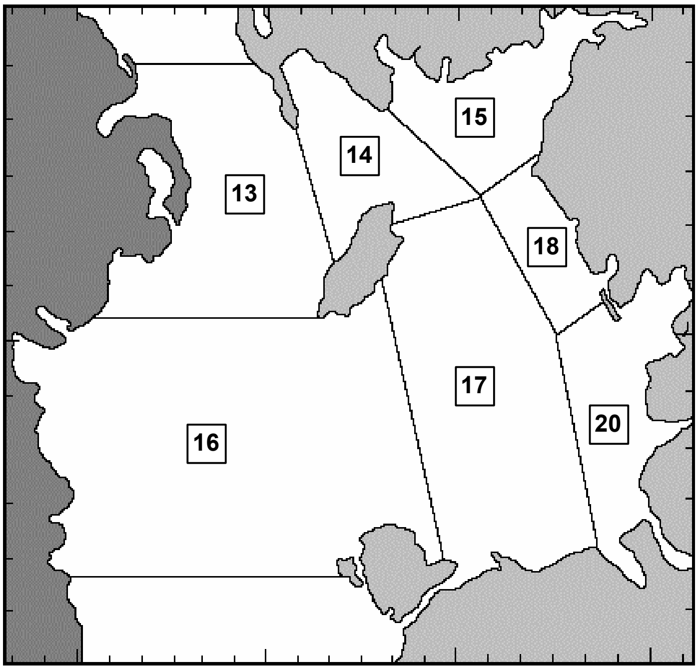
For details see Table 4.7

Figure 4.4 – Northern European regional compartments included in the marine dispersion model as modified for PC-CREAM 08



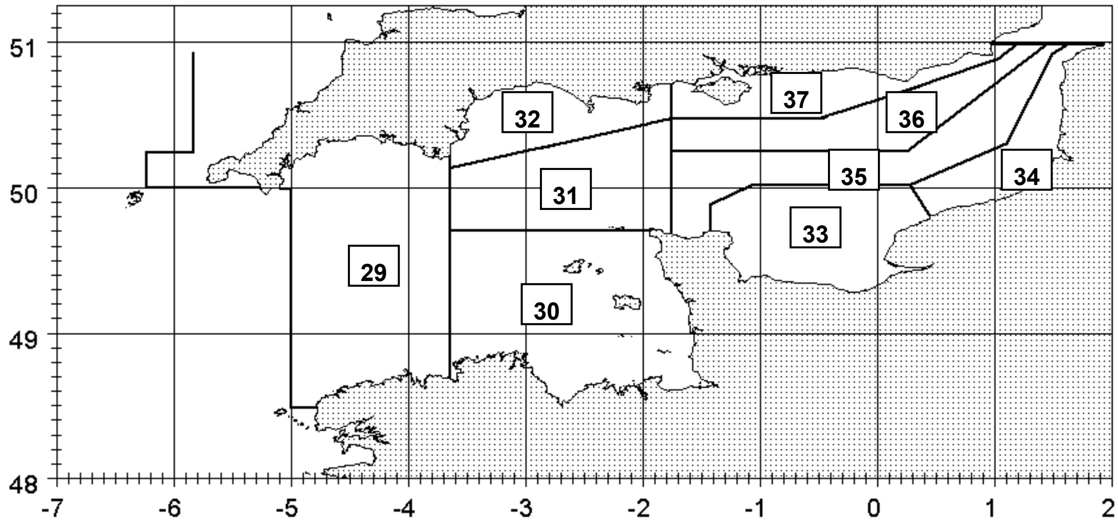
For details see Table 4.7

Figure 4.5 Irish Sea regional compartments in the marine dispersion model in PC-CREAM 08



For details see Table 4.7

Figure 4.6 English Channel compartments the marine dispersion model in PC-CREAM 08



For details see Table 4.7

Figure 4.7 – Annual average concentration in filtered sea water in the Sellafield local compartment, following a hypothetical release of 1 TBq in 1 year of caesium-137 and plutonium-239

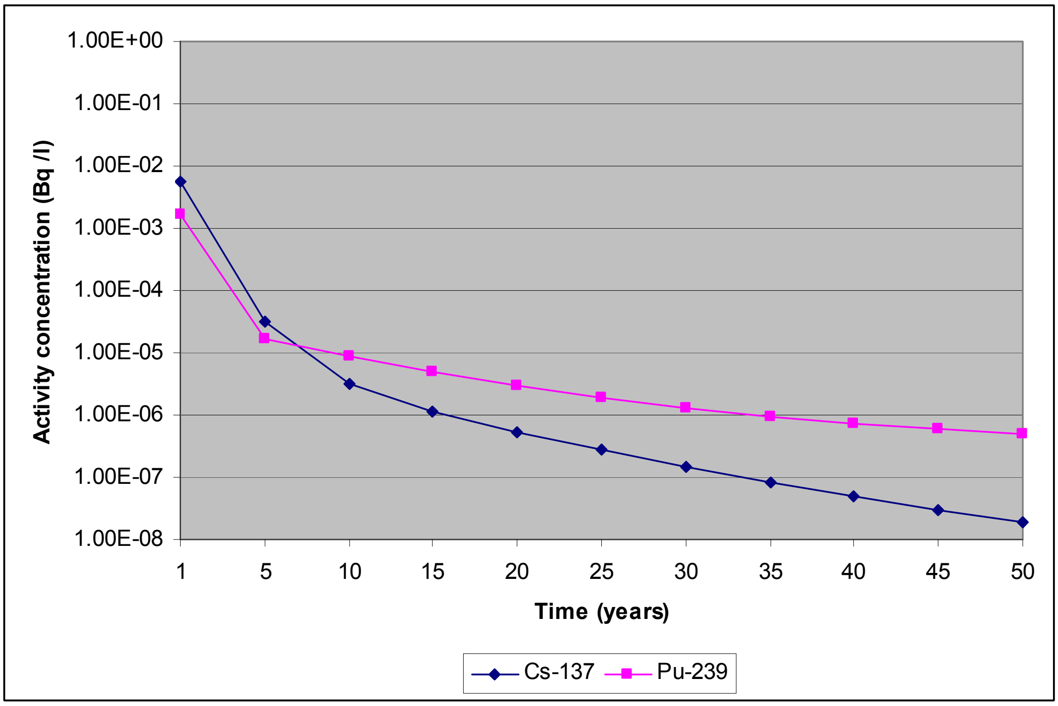


Figure 4.8 – Annual average concentration in filtered sea water in the Vandellos local compartment, following a hypothetical release of 1 TBq in 1 year of caesium-137 and plutonium-239

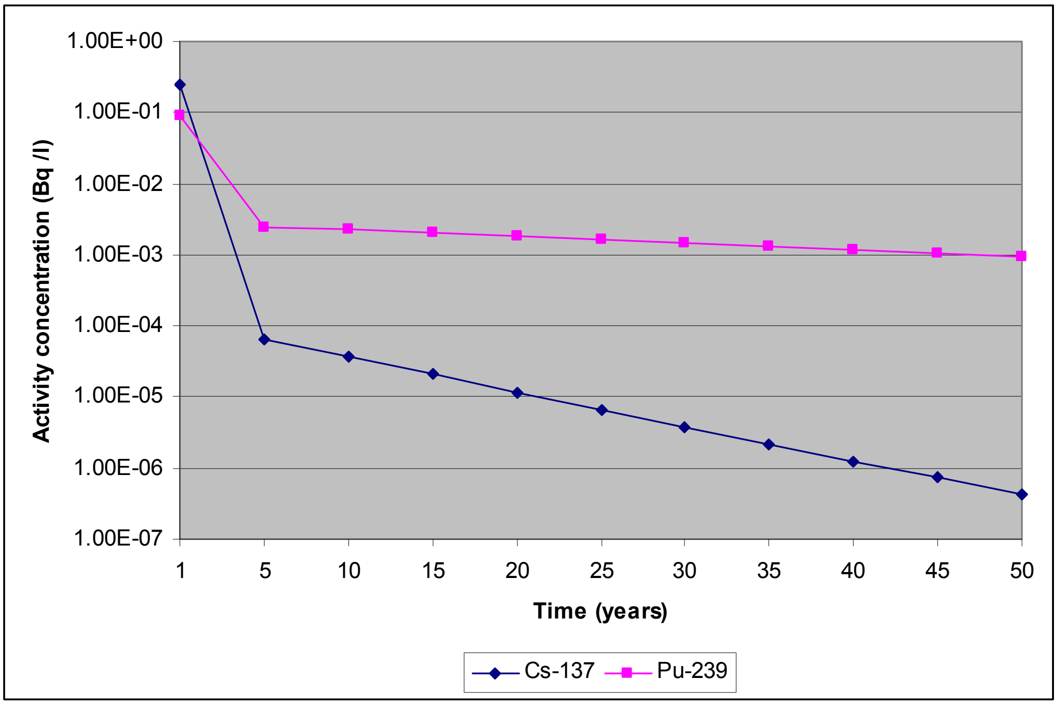


Figure 4.9 – Annual average concentration in the top sea bed sediment in the Sellafield local compartment, following a hypothetical release of 1 TBq in 1 year of caesium-137 and plutonium-239

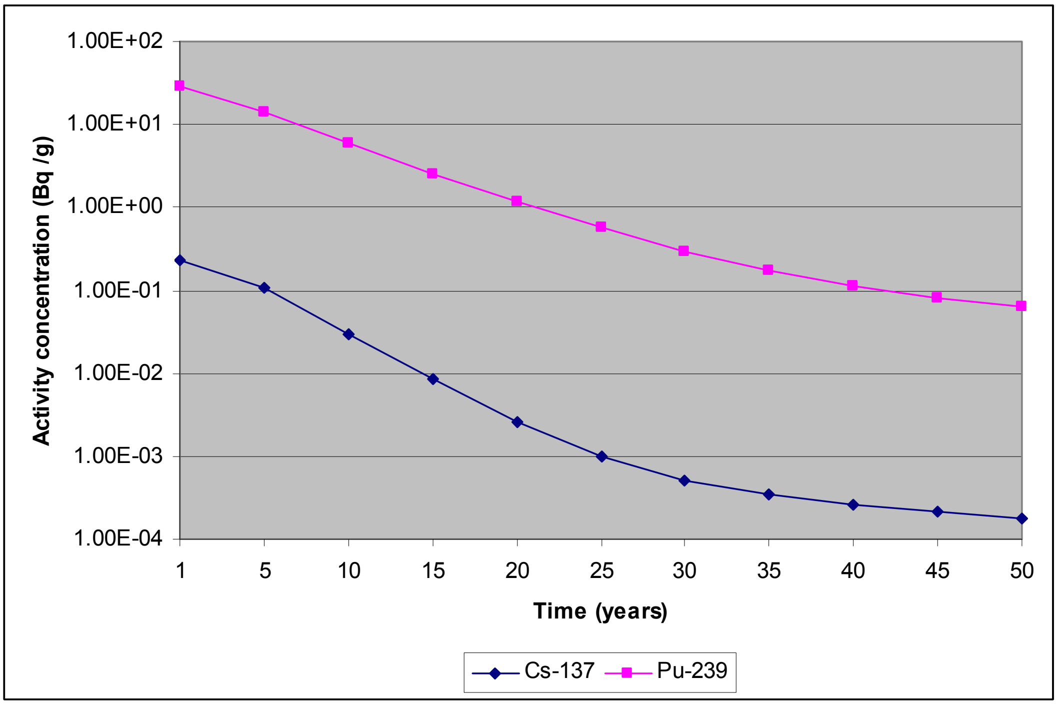


Figure 4.10 – Annual average concentration in the top sea bed sediment in the Vandellos local compartment, following a hypothetical release of 1 TBq in 1 year of caesium-137 and plutonium-239

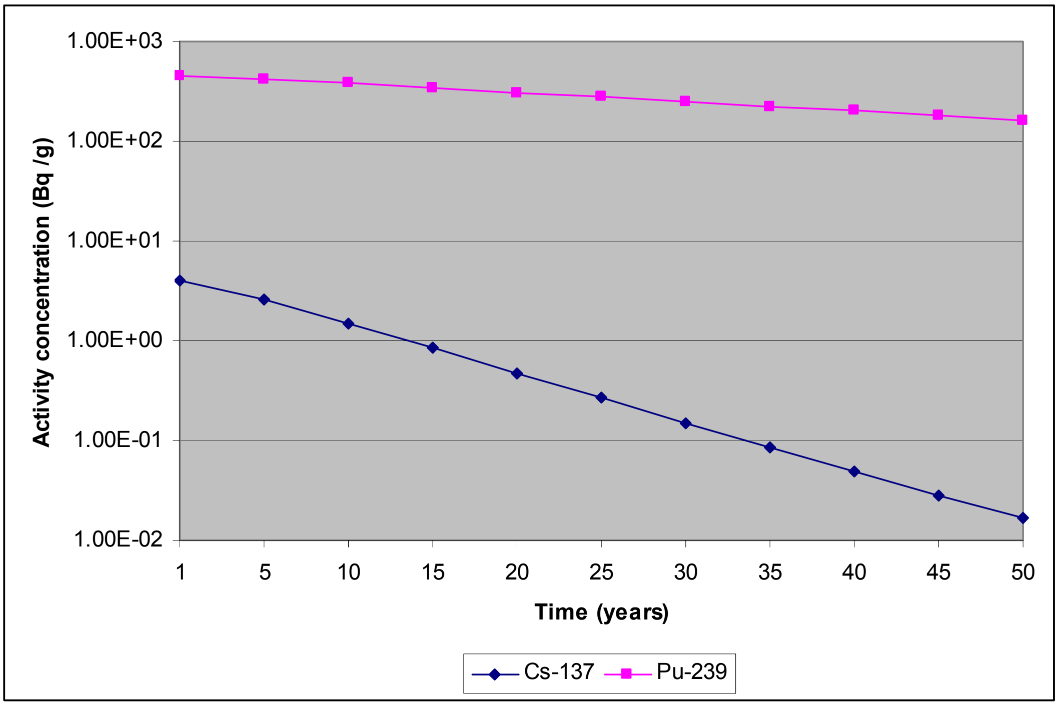


Figure 4.11 – Results of PC-CREAM 08 showing the annual collective dose following a hypothetical release of 1 TBq in 1 year of caesium-137 to the local Sellafield compartment for the EU12 population integrated to 50 years

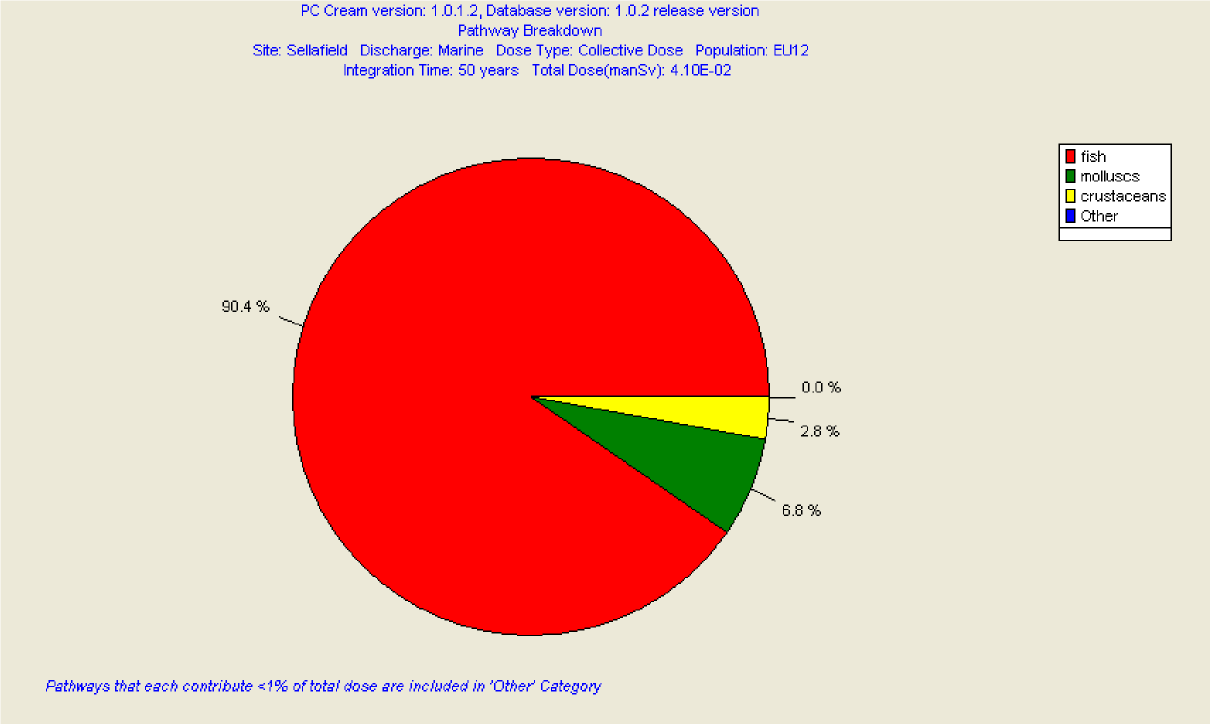


Figure 4.12 – Results of PC-CREAM 08 showing the annual collective dose following a hypothetical release of 1 TBq in 1 year of plutonium-239 to the local Sellafield compartment for the EU12 population integrated to 50 years

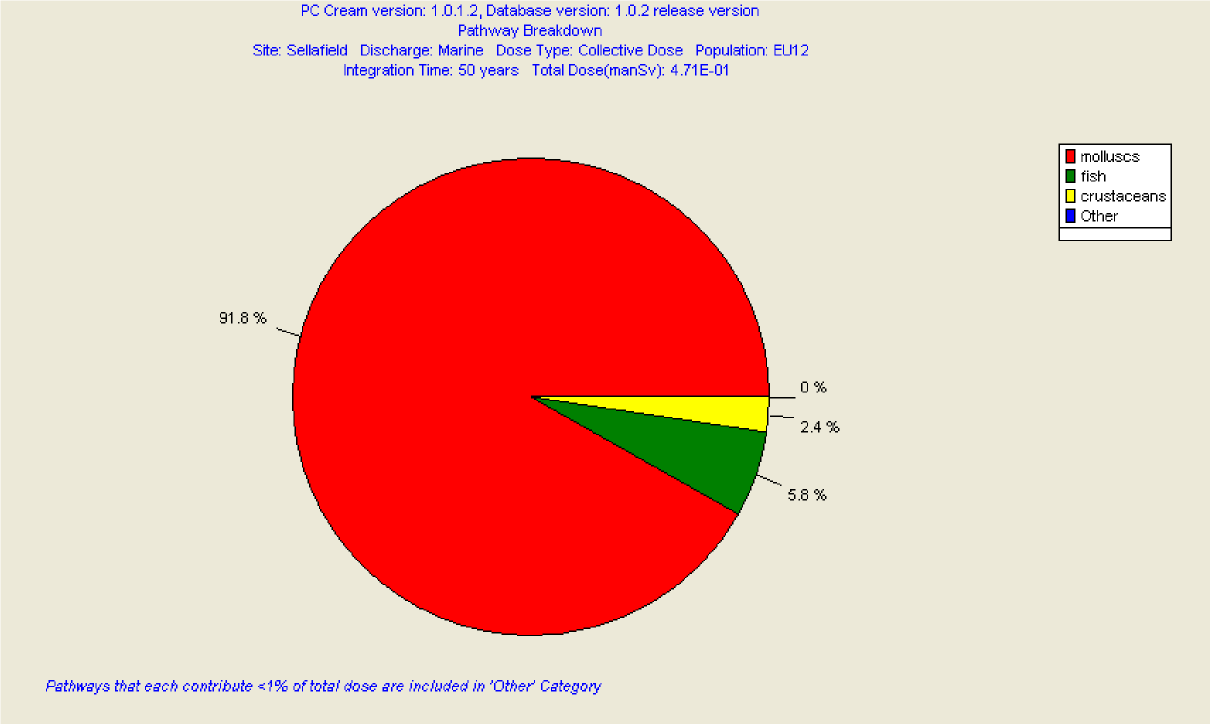


Figure 4.13- Comparison of $^{239/240}$Pu discharges from Sellafield with MARINA II model predictions and environmental measurements in Irish Sea west for filtered water (Bexon et al, 2003)

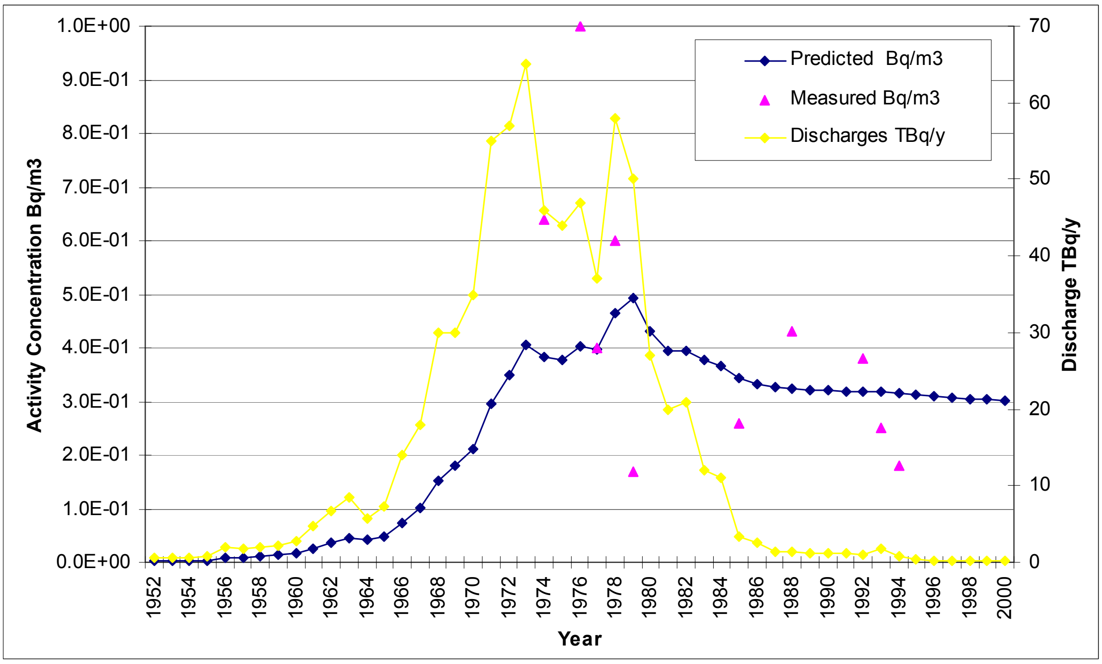


Figure 4.14 - Comparison of activity concentrations of $^{239/240}$Pu in filtered water in Irish Sea West for a historic discharge from Sellafield (Bexon et al, 2003)

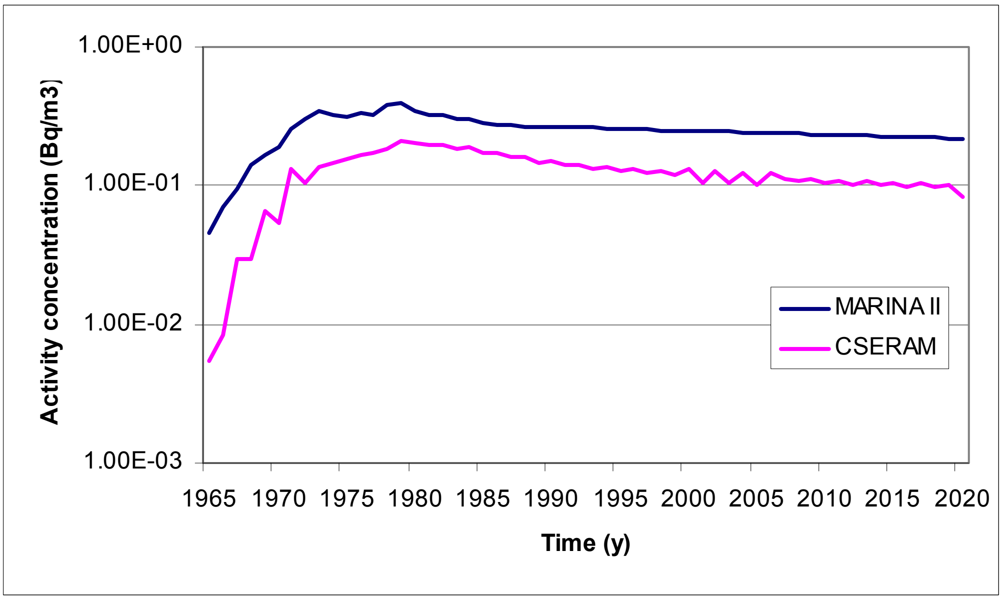
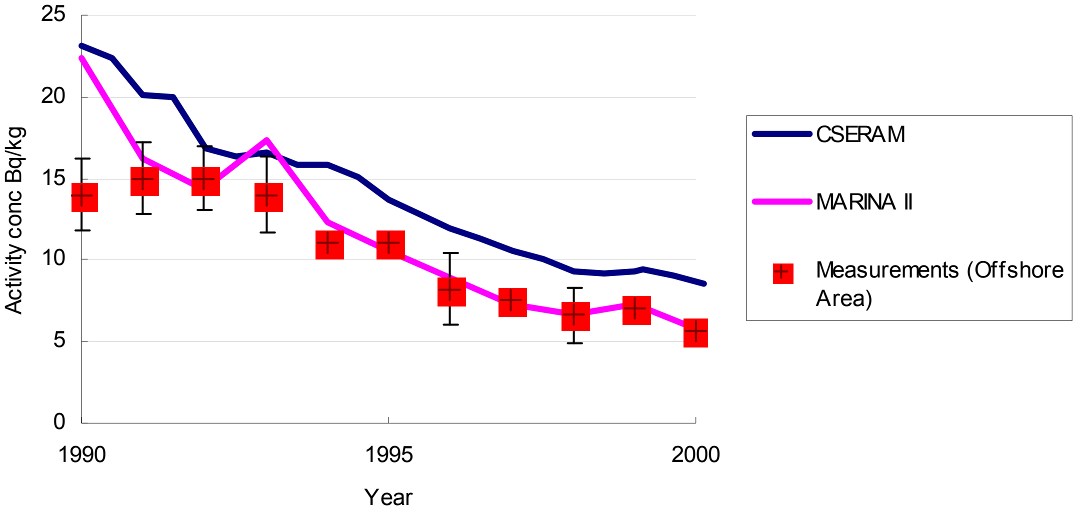


Figure 4.15 - Estimated and measured activity concentrations of caesium-137 in fish for the Sellafield coastal area (Jones et al, 2003)



5 GLOBAL CIRCULATION OF RADIONUCLIDES

5.1 Introduction

Some radionuclides, owing to the magnitude of their radioactive half-lives and their behaviour in the environment, may become globally dispersed and act as a long term source of irradiation of both regional and world populations; this is in addition to the irradiation of the population exposed during the initial dispersion of these radionuclides from their points of discharge. The radionuclides which are important in this context are krypton-85, tritium, iodine-129 and carbon-14 with half-lives of 10.72, 12.35, 1.57 10^7 and 5730 years, respectively.

Once these radionuclides become globally distributed, essentially the whole of the population of the EC and the world will be irradiated at the same level. Krypton-85 is not readily incorporated into body tissues and it is significant only in respect of external irradiation of the population. The nature of the radiation emitted by the other three radionuclides is such that external irradiation is not important compared with that from the incorporation of these radionuclides in the body. Doses to individuals arising from the global circulation of these radionuclides are generally small and the models included in PC-CREAM 08 and described in this chapter are principally used to determine collective doses.

5.2 Global circulation models

Global circulation models are compartmental models in which a compartment may represent the whole of a particular environmental medium, for example, seabed sediment or soil, on a global basis. Each compartment in a global circulation model may represent a large part of an environmental medium, in which the radionuclide is assumed to be homogeneously mixed. Thus, the short term, local dispersion of the radionuclide cannot be adequately modelled using a global circulation model. Local dispersion models, such as those described in Chapters 3 and 4, are used to predict environmental concentrations close to the release point at relatively short times. Such local dispersion models are specific to the actual point of release and the nature of the release, for example, to atmosphere, to a river or to the sea. Maximum doses to individuals and collective doses to the population near the release point are calculated by these local models.

5.2.1 Krypton-85

The compartmental model for krypton-85 is unchanged from that described previously in Simmonds, Lawson and Mayall (1995), being a simplified version of a model devised by Kelly et al (1975). The model is shown in Figure 5.1. The discharged krypton-85 is assumed to be dispersed uniformly, and instantaneously, throughout the troposphere of the northern hemisphere which is assumed to have a height of 10 km and a mass of 1.9 10^{18} kg. Exchange takes place between the troposphere of the two hemispheres with a half time of about 2 years. Within a few years the krypton-85 becomes uniformly mixed throughout the whole troposphere and the only loss from the system is by radioactive decay.

Krypton-85 in the atmosphere results in the exposure of man by external irradiation from both photons and electrons. The dose to the skin is dominated by beta irradiation, the gamma component being about 1% of the skin dose (ICRP, 1979; Jacob et al, 1990). About half of the effective dose arises from gamma irradiation, the other half arising from the beta irradiation of the skin (ICRP, 1979; Zankl et al, 1992). The dose rate for immersion in a semi-infinite cloud of krypton-85, appropriate for use with the global circulation model, is given in Table 5.1. The dose rate presented in Table 5.1 assumes that the irradiated persons are not shielded in any way (ICRP, 1996). As described in Chapter 3 the gamma component of the dose rates can be multiplied by a factor of 0.2 to obtain the dose rates to people inside buildings typical of those in the EC. The effectiveness of clothing in shielding the skin from the beta irradiation is variable and is not generally taken into account. In PC-CREAM 08 only effective doses are calculated.

5.2.2 Tritium

The compartmental model for the global circulation of tritium is based upon a model of the global water cycle developed by the National Council on Radiation Protection and Measurements (NCRP, 1979). The model is shown in Figure 5.2 and the parameter values are given in Table 5.2. The model is used to calculate the specific activity of tritium in the water in each compartment as a function of time following release.

The average intakes of water by adults are given in Table 5.3. The total intake is the average of those for men and women (ICRP, 1975). Data on water intake rates given in ICRP publication 89 (ICRP, 2002) differ slightly to the values given in Table 5.3 but do not justify a change to the model. The intakes by inhalation and absorption through the skin are based upon the water content of air (Simmonds, Lawson and Mayall, 1995) of 8.1 g m^{-3}; the average adult inhalation rate is taken to be 19 m^3 day^{-1} (NEA/OECD, 1983) and the skin is taken to absorb the water in 14.4 m^3 of air per day (NCRP, 1979). Ingestion of water by drinking fluids, excluding milk, is taken to be 600 kg y^{-1} (NRPB, 1987) and water intake by ingestion of seafood is taken to be 7.3 kg y^{-1} (NCRP, 1979). The intake of water by ingestion of foods, other than seafood, has been obtained by subtraction of the total intake by other routes from the total intake by all routes (ICRP, 1975).

In calculating the intake of tritium by man the intake rates in Table 5.3 are multiplied by the specific activities of tritium in water in several compartments of the model (NCRP, 1979). Any foodstuffs obtained from a compartment are assumed to have the same specific activity of tritium in water as that in the compartment itself. Intake by inhalation and absorption through the skin is assumed to take place from the atmosphere compartment. Intake by drinking fluids, other than milk, is assumed to be 80% from the surface streams and fresh water lakes compartment and 20% from the deep ground water compartment. Intake by seafood is assumed to be entirely from the ocean surface compartment. Intake by other foods is assumed to be 50% from the atmosphere compartment and 50% from the surface soil water compartment.

5.2.3 Iodine-129

The compartmental model for the global circulation of iodine is shown in Figure 5.3. This represents a revision (Titley et al, 1995) of a preceding model devised by Kocher (1979) and modified by Smith and White (1983). The major sections of the environment in which iodine circulates are included and the transfers of iodine assumed to occur between them represent known mechanisms. The inventories of stable iodine in the compartments and the fluxes between them were determined from environmental measurements and from the requirement for mass balance. Iodine-129 released into any compartment is assumed to be transported with stable iodine and hence the specific activity of iodine-129 in the total iodine in each compartment is determined.

Iodine intakes by man from each compartment, and hence intakes of iodine-129, are estimated from average food consumption and inhalation rates. The average adult inhalation rate is taken to be 19 m^3 day^{-1} (NEA/OECD, 1983). The assumed ingestion rates are given in Table 5.4 (Titley et al, 1995; Kocher, 1979). For estimating collective doses it is considered adequate to use adult values for intakes and doses per unit intake and not take account of the different values for children (see Chapter 2). The exposure pathways considered, the model compartments used, and the parameter values used to obtain iodine concentrations in foodstuffs are given in Table 5.5. Using the assumed concentrations of stable iodine in air and in foodstuffs the daily intake of stable iodine can be estimated (see Table 5.6.).

5.2.4 Carbon-14

The compartmental model for the global circulation of carbon-14 is based upon a model for the global carbon cycle shown in Figure 5.4. The stable carbon inventories and fluxes between compartments are given in Table 5.7. The oceanic compartments of the model have been derived from those in the MINIBOX model (Mobbs et al, 1988). The oceanic compartment volumes and transfer fluxes were converted to masses of dissolved carbon dioxide using an average oceanic concentration of 2.25 10^{-3} molar (Bainbridge, 1981). The exchange of carbon dioxide between the atmosphere and the ocean was determined using a total flux of 9 10^{13} kg carbon y^{-1}, apportioned by ocean surface area. Exchange between the bottom layers of the oceans and the single sediment compartment has also been apportioned according to ocean area, using a total sedimentation flux of 2 10^{11} kg carbon y^{-1} and an identical resuspension flux (Siegenthaler, 1989). The terrestrial compartments and fluxes in the model are from Emanuel et al (1981) and further details of the model and parameter values are given in Titley et al (1995). Changes to the global carbon cycle caused by burning fossil fuels and changes in land use are likely to have a significant effect on exposures that will occur hundreds or thousands of years in the future. However, as future trends for these processes and other factors that affect collective dose are unknown they have not been taken into account in the current version of the model used in PC-CREAM 08.

The intake of carbon by man is assumed to arise entirely from ingestion, because most carbon dioxide inhaled is subsequently exhaled. It is assumed that the specific activity of carbon-14 in the stable carbon ingested is the same as that in the most relevant compartments for food intakes. 99% of food intakes are from terrestrial biota compartments and 1% from marine foods in the surface marine compartments. The ingestion rate of carbon by an average adult is taken to be 0.255 kg day^{-1} (ICRP, 1975).

5.3 Application of the models

The models described in section 5.2 have been applied to evaluate the time variation of the concentration and integrated concentration of each radionuclide in the relevant compartments, following the discharge of 1 TBq of the radionuclide over 1 year. Consideration has been given to the discharge of each radionuclide to the atmospheric and to the aquatic environment, except for krypton-85 which is discharged only to the atmosphere. In each model, for the compartments most appropriate to the estimation of doses to man, the time variation of the concentration and time-integral of concentration of each radionuclide are given in Tables 5.8 to 5.11. The concentrations and integrated concentrations are expressed in terms of the activity of the radionuclide in unit mass of air for krypton-85, of water for tritium, of stable iodine for iodine-129 and of stable carbon for carbon-14. When a concentration is required at an intermediate time from those listed in Tables 5.8 to 5.11 it is calculated using linear interpolation.

The predicted concentrations and integrated concentrations in the environment are combined with pathway and dosimetric data to obtain collective doses. The following explanation is given in terms of the calculation of collective effective dose from integrated concentrations; the calculation of collective dose rates from the predicted concentrations is similar. In the case of krypton-85 the collective dose commitment in the exposed population is evaluated directly from the predicted integrated concentration in the air:

$$D\ (t) = I\ (t)\ F\ P \qquad (5.1)$$

where D (t) = collective effective dose truncated at time t (man Sv),

I (t) = integrated concentration in the appropriate hemisphere at time t (Bq y kg^{-1}) (see Table 5.8),

F = effective dose rate per unit concentration in air (Sv y^{-1} per Bq kg^{-1}) (see Table 5.1),

P = exposed population

The models do not take into account future population growth because of the uncertainties and use instead static populations. In PC-CREAM 08 the world population is set at 10^{10} and the population used for each European country is for 2003 (Table 5.12). If the collective dose to the world population is being considered then both hemispheres must be taken into account with about 85% of the population currently being in the northern hemisphere. However, this is likely to change during the next few decades as the main regions of population growth are expected to be in the southern hemisphere (UN, 2007).

In the case of tritium and carbon-14 a specific activity approach is used to obtain intakes by man and hence doses, as explained in sections 5.2.2 and 5.2.4. The collective effective dose commitment is obtained as

$$D(t) = \sum_c \sum_p I_c(t)\, f_{p,c}\, R_p\, U_p\, P \tag{5.2}$$

where $D(t)$ = collective effective dose truncated at time t (man Sv),

$I_c(t)$ = integrated specific activity in compartment c at time t (Bq y kg^{-1}),

$f_{p,c}$ = fraction of an individual's intake of water or stable carbon for pathway p from compartment c,

R_p = total intake by an individual of water or stable carbon from pathway p (kg y^{-1}),

U_p = effective dose per unit intake from pathway p (Sv Bq^{-1}),

P = exposed population.

The summations are performed over the relevant compartments and pathways, as described in sections 5.2.2 and 5.2.4.

A similar approach is used to calculate collective doses from iodine-129 except that the intake by an individual is not calculated from the intake of stable iodine but is obtained using transfer factors to relate activity concentrations in food and air to the activity concentrations in the various compartments; see section 5.2.3.

5.4 Interface between the marine dispersion models and global circulation models

When tritium, carbon-14 or iodine-129 are discharged to the aquatic environment there will be a considerable delay before they become uniformly distributed throughout the oceans of the northern hemisphere. In the tritium and iodine-129 global circulation models this dispersion is assumed to be instantaneous; in the case of carbon-14 the structure in the representation of the oceans enables the model to simulate more accurately the dispersion. In all cases, however, the assumption in the global models that the activity is dispersed instantaneously into the compartment receiving the discharge means that the short-term results of these models should be regarded with caution.

The marine models described in Chapter 4 also take some account of the global dispersion of radionuclides in the oceans. However, the transfer of the radionuclides to man calculated by the marine models is essentially that arising only from uptake into seafood, whereas the global circulation models include the transfer of the radionuclides to the atmosphere and the terrestrial environment. Thus the marine models will underestimate the transfer of the radionuclides to man in the long term. It is recommended, therefore, that for tritium and iodine-129, in which ingestion of seafood is considered explicitly, this pathway is ignored in the global model until the release has become substantially dispersed into a large ocean, such as the Atlantic Ocean. Up to this time the intake from seafood should be estimated by the marine model. After this time the results of the marine model can be ignored and the results of the global models used. In the case of carbon-14, the intake from seafood is small in comparison with that from terrestrial vegetation; the collective doses calculated by the marine model can be summed with those calculated by the global model at all times.

5.5 References

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5.6 Tables

TABLE 5.1 Dose rates from external irradiation per unit concentration in air of krypton-85*

Organ	Dose rate (Sv y^{-1} / Bq kg^{-1})
Effective	8.0 10^{-9}

* The density of air has been taken to be 1.225 kg m^{-3}

TABLE 5.2 Parameter values for the global circulation model for tritium (see also Figure 5.2)

Compartment number	Compartment name	Volume m^3	Flux to compartment number	Flux m^3 y^{-1}
1	Atmosphere	1.30 10^{13}	2	9.93 10^{13}
			3	6.00 10^{11}
			4	1.00 10^{11}
			5	3.20 10^{14}
2	Surface soil water	6.70 10^{13}	1	6.85 10^{13}
			3	2.94 10^{13}
			6	2.55 10^{13}
3	Surface streams and fresh water lakes	1.26 10^{14}	1	1.00 10^{12}
			5	3.00 10^{13}
4	Saline lakes and inland seas	1.04 10^{14}	1	5.00 10^{11}
5	Ocean surface	2.70 10^{16}	1	3.50 10^{14}
			7	1.60 10^{15}
6	Deep ground water	8.35 10^{15}	2	2.40 10^{13}
			3	1.00 10^{12}
			4	4.00 10^{11}
7	Deep ocean	1.29 10^{18}	5	1.60 10^{15}

TABLE 5.3 Average adult water intake rates for the tritium model

Source	Intake rate (kg y^{-1})
Inhalation	56
Absorption through skin	43
Drinking fluids (excluding milk)	600
Food (including milk, excluding seafood)	220
Seafood	7.3
Total	930

TABLE 5.4 Average adult consumption rates for the iodine-129 model

Food	World (kg year^{-1})	EC (kg year^{-1})
Milk	67	180
Meat	21	41
Leafy vegetables	42	62
Other vegetables, fruits and nuts	135	197
Cereals	135	137
Freshwater fish	1.4	2.1
Marine fish and shellfish	8.0	12
Other foods	25	53
Fluids excluding milk	600	600

TABLE 5.5 Pathways and parameter values for the iodine-129 model

Pathway	Compartment	Parameter values
Inhalation	Land Atmosphere	Inhalation Rate: 19 m^3 day^{-1}
Foliar deposition	Land Atmosphere	Concentration in food / concentration in land atmosphere: milk 1.94 10^2 m^3 kg^{-1} meat 7.28 10^2 m^3 kg^{-1} leafy vegetables 1.87 10^2 m^3 kg^{-1} other vegetables and fruit 2.07 10^2 m^3 kg^{-1} cereals 7.12 10^2 m^3 kg^{-1}
Ingestion of land surface water	Surface Soil Water	^{129}I concentration factor for freshwater fish: 20 l kg^{-1}; ^{129}I transfer factor for meat: 2 10^{-3} d kg^{-1}; ^{129}I transfer factor for milk: 5 10^{-3} d kg^{-1}; Water intake rate by beef cattle: 55 l d^{-1}; Water intake rate by dairy cattle: 55 l d^{-1}.
Ingestion of marine fish and shellfish	Ocean mixed layer	^{129}I concentration factor for seafood: 10 l kg^{-1};
Root uptake	Surface Soil Water	Forage consumption rate for cattle: 60 kg d^{-1}.

TABLE 5.6 Assumed daily intake of stable iodine in the iodine-129 model

Pathway	World (μ g d^{-1})	EC (μ g d^{-1})
Inhalation	0.29	0.29
Foliar deposition	6.60	8.81
Ingestion of land surface water	5.34	5.72
Ingestion of marine fish and shellfish	11.0	16.2
Root uptake	197	278
Total	221	309

TABLE 5.7 Parameter values for the global circulation model for carbon-14 (see also Figure 5.4)

Compartment Number	Compartment Name	Total carbon in the compartment (kg x 10^{13})	Flux to Compartment Number	Flux of carbon (kg y^{-1} x 10^{10})
1	Atmosphere	75.0	2	7700
			4	3600
			7	1981
			8	175
			9	5812
			10	48
			11	220
			12	165
2	Non-Woody Tree Parts	3.7	1	2500
			3	3100
			5	2100
3	Woody Tree Parts	45.2	1	1400
			5	1500
			6	200
4	Ground Vegetation	6.9	1	1800
			5	1200
			6	600
5	Detritus & Decomposers	8.1	1	4500
			6	300
6	Active Soil Carbon	112	1	1020
			7	54
			8	8.4
			9	17.5
7	Atlantic Ocean 0-500 m	105	1	203.4
			9	385
			10	4.5
			14	3655
8	Arctic Ocean 0-200 m	9.6	1	192
			14	50.2
			15	1051
9	Pacific Ocean 0-500 m	308	1	5820
			7	560
			10	451
			16	2726
10	Antarctic Ocean 0-500 m	34.3	1	648
		~~34.3~~	7	266
			9	14.5
			11	420
11	Antarctic Ocean 500-2200 m	168	1	220
			10	21240
			12	1515
			14	1177

Compartment Number	Compartment Name	Total carbon in the compartment (kg x 10^{13})	Flux to Compartment Number	Flux of carbon (kg y^{-1} x 10^{10})
			16	379
12	Antarctic Ocean 2200-3200 m	148	1	165
			11	1253
			13	1446
			17	98.1
			18	559
13	Antarctic Ocean >3200 m	180	12	1184
			18	1075
			19	439
			21	2.5
14	Atlantic Ocean 500-2200 m	357	7	3219
			8	836
			11	391
			15	55.8
			16	422
			17	893
15	Arctic Ocean 200-1200 m	47.9	8	265
			14	231
			17	441
			19	178
			21	0.96
16	Pacific Ocean 500-2200 m	1050	9	3338
			11	1252
			14	160
			18	809
17	Atlantic Ocean 2200-3200 m	210	12	535
			14	544
			15	4.0
			18	115
			19	836
18	Pacific Ocean >2200 m	1230	12	122
			13	290
			16	2032
			17	115
			19	397
			21	12.3
19	Atlantic Ocean >3200 m	168	13	963
			15	3.0
			17	487
			18	397
			21	4.2
21	Sediments	7200000	13	2.5
			15	0.96

Compartment Number	Compartment Name	Total carbon in the compartment (kg x 10^{13})	Flux to Compartment Number	Flux of carbon (kg y^{-1} x 10^{10})
			18	12.3
			19	4.2

TABLE 5.8 Model results for the global circulation of 1 TBq of krypton-85 released over 1 year to the atmosphere of the northern hemisphere

Time after release commences (years)	Concentration* and integrated concentration in the atmosphere Bq kg^{-1}, Bq y kg^{-1}			
	Northern hemisphere		Southern hemisphere	
1	$2.5\ 10^{-7}$	$1.4\ 10^{-7}$	$5.7\ 10^{-8}$	$2.1\ 10^{-8}$
2	$1.8\ 10^{-7}$	$3.5\ 10^{-7}$	$1.1\ 10^{-7}$	$1.1\ 10^{-7}$
5	$1.2\ 10^{-7}$	$7.7\ 10^{-7}$	$1.2\ 10^{-7}$	$4.8\ 10^{-7}$
10	$8.7\ 10^{-8}$	$1.3\ 10^{-6}$	$8.7\ 10^{-8}$	$9.9\ 10^{-7}$
20	$4.5\ 10^{-8}$	$1.9\ 10^{-6}$	$4.5\ 10^{-8}$	$1.6\ 10^{-6}$
50	$6.5\ 10^{-9}$	$2.5\ 10^{-6}$	$6.5\ 10^{-9}$	$2.2\ 10^{-6}$
100	$2.6\ 10^{-10}$	$2.6\ 10^{-6}$	$2.6\ 10^{-10}$	$2.3\ 10^{-6}$
200	$4.0\ 10^{-13}$	$2.6\ 10^{-6}$	$4.0\ 10^{-13}$	$2.3\ 10^{-6}$
500	$1.5\ 10^{-21}$	$2.6\ 10^{-6}$	$1.5\ 10^{-21}$	$2.3\ 10^{-6}$
700	$1.0\ 10^{-26}$	$2.6\ 10^{-6}$	$1.0\ 10^{-26}$	$2.3\ 10^{-6}$
1000	$6.8\ 10^{-30}$	$2.6\ 10^{-6}$	$6.8\ 10^{-30}$	$2.3\ 10^{-6}$
10000	$1.7\ 10^{-35}$	$2.6\ 10^{-6}$	$1.7\ 10^{-35}$	$2.3\ 10^{-6}$
1000000	$3.0\ 10^{-52}$	$2.6\ 10^{-6}$	$3.0\ 10^{-52}$	$2.3\ 10^{-6}$
100000000	$2.2\ 10^{-82}$	$2.6\ 10^{-6}$	$2.2\ 10^{-82}$	$2.3\ 10^{-6}$

* The concentration is expressed as Bq of krypton-85 per kg of air in the atmosphere.

TABLE 5.9a Specific activity and integrated specific activity of tritium in each compartment following a release of 1 TBq in one year to the ocean surface

Time after release commences (years)	Specific activity and integrated specific activity (Bq kg^{-1}, Bq y kg^{-1})													
	Atmosphere		Surface soil water		Surface streams and freshwater lakes		Saline lakes and inland seas		Deep ground water		Ocean surface		Deep ocean	
1	3.0 10^{-8}	1.5 10^{-8}	1.3 10^{-8}	4.7 10^{-9}	1.1 10^{-9}	2.9 10^{-10}	1.4 10^{-11}	4.5 10^{-12}	1.4 10^{-11}	3.7 10^{-12}	3.5 10^{-8}	1.8 10^{-8}	2.2 10^{-11}	7.3 10^{-12}
2	3.0 10^{-8}	4.5 10^{-8}	2.2 10^{-8}	2.4 10^{-8}	4.8 10^{-9}	3.2 10^{-9}	4.2 10^{-11}	3.3 10^{-11}	7.0 10^{-11}	4.4 10^{-11}	3.1 10^{-8}	5.1 10^{-8}	6.0 10^{-11}	4.9 10^{-11}
5	2.1 10^{-8}	1.2 10^{-7}	1.8 10^{-8}	8.6 10^{-8}	1.2 10^{-8}	3.0 10^{-8}	1.0 10^{-10}	2.6 10^{-10}	2.3 10^{-10}	5.2 10^{-10}	2.2 10^{-8}	1.3 10^{-7}	1.4 10^{-10}	3.6 10^{-10}
10	1.2 10^{-8}	2.0 10^{-7}	1.0 10^{-8}	1.5 10^{-7}	1.0 10^{-8}	8.8 10^{-8}	1.5 10^{-10}	9.1 10^{-10}	3.5 10^{-10}	2.0 10^{-9}	1.2 10^{-8}	2.1 10^{-7}	1.9 10^{-10}	1.2 10^{-9}
20	3.8 10^{-9}	2.7 10^{-7}	3.2 10^{-9}	2.1 10^{-7}	4.0 10^{-9}	1.6 10^{-7}	1.4 10^{-10}	2.4 10^{-9}	3.2 10^{-10}	5.6 10^{-9}	3.9 10^{-9}	2.9 10^{-7}	1.8 10^{-10}	3.1 10^{-9}
50	1.6 10^{-10}	3.1 10^{-7}	1.4 10^{-10}	2.4 10^{-7}	1.7 10^{-10}	1.9 10^{-7}	4.0 10^{-11}	4.9 10^{-9}	8.2 10^{-11}	1.1 10^{-8}	1.6 10^{-10}	3.2 10^{-7}	4.4 10^{-11}	6.2 10^{-9}
100	3.2 10^{-12}	3.1 10^{-7}	3.6 10^{-12}	2.4 10^{-7}	3.7 10^{-12}	1.9 10^{-7}	2.9 10^{-12}	5.6 10^{-9}	4.9 10^{-12}	1.3 10^{-8}	3.2 10^{-12}	3.2 10^{-7}	2.8 10^{-12}	6.9 10^{-9}
200	1.0 10^{-14}	3.1 10^{-7}	1.2 10^{-14}	2.4 10^{-7}	1.2 10^{-14}	1.9 10^{-7}	1.3 10^{-14}	5.7 10^{-9}	1.7 10^{-14}	1.3 10^{-8}	1.0 10^{-14}	3.2 10^{-7}	1.0 10^{-14}	7.0 10^{-9}
500	5.1 10^{-22}	3.1 10^{-7}	5.4 10^{-22}	2.4 10^{-7}	5.4 10^{-22}	1.9 10^{-7}	6.5 10^{-22}	5.7 10^{-9}	6.6 10^{-22}	1.3 10^{-8}	5.0 10^{-22}	3.2 10^{-7}	5.0 10^{-22}	7.0 10^{-9}
700	6.9 10^{-27}	3.1 10^{-7}	7.1 10^{-27}	2.4 10^{-7}	7.2 10^{-27}	1.9 10^{-7}	8.4 10^{-27}	5.7 10^{-9}	8.2 10^{-27}	1.3 10^{-8}	6.8 10^{-27}	3.2 10^{-7}	6.8 10^{-27}	7.0 10^{-9}
1000	3.7 10^{-30}	3.1 10^{-7}	3.8 10^{-30}	2.4 10^{-7}	3.9 10^{-30}	1.9 10^{-7}	4.6 10^{-30}	5.7 10^{-9}	4.4 10^{-30}	1.3 10^{-8}	3.7 10^{-30}	3.2 10^{-7}	3.7 10^{-30}	7.0 10^{-9}
10000	1.2 10^{-35}	3.1 10^{-7}	1.3 10^{-35}	2.4 10^{-7}	1.3 10^{-35}	1.9 10^{-7}	1.5 10^{-35}	5.7 10^{-9}	1.4 10^{-35}	1.3 10^{-8}	1.2 10^{-35}	3.2 10^{-7}	1.2 10^{-35}	7.0 10^{-9}
1000000	4.1 10^{-52}	3.1 10^{-7}	4.2 10^{-52}	2.4 10^{-7}	4.2 10^{-52}	1.9 10^{-7}	4.8 10^{-52}	5.7 10^{-9}	4.7 10^{-52}	1.3 10^{-8}	4.0 10^{-52}	3.2 10^{-7}	4.0 10^{-52}	7.0 10^{-9}
100000000	1.6 10^{-81}	3.1 10^{-7}	1.6 10^{-81}	2.4 10^{-7}	1.6 10^{-81}	1.9 10^{-7}	1.7 10^{-81}	5.7 10^{-9}	1.8 10^{-81}	1.3 10^{-8}	1.6 10^{-81}	3.2 10^{-7}	1.6 10^{-81}	7.0 10^{-9}

TABLE 5.9b Specific activity and integrated specific activity of tritium in each compartment following a release of 1 TBq in one year to the surface streams and freshwater lakes

Time after release commences (years)	Specific activity and integrated specific activity (Bq kg^{-1}, Bq y kg^{-1})													
	Atmosphere		Surface soil water		Surface streams and freshwater lakes		Saline lakes and inland seas		Deep ground water		Ocean surface		Deep ocean	
1	$2.0\ 10^{-8}$	$9.6\ 10^{-9}$	$8.4\ 10^{-9}$	$3.0\ 10^{-9}$	$6.9\ 10^{-6}$	$3.6\ 10^{-6}$	$9.0\ 10^{-12}$	$2.9\ 10^{-12}$	$9.1\ 10^{-12}$	$2.4\ 10^{-12}$	$3.9\ 10^{-9}$	$1.4\ 10^{-9}$	$1.7\ 10^{-12}$	$4.2\ 10^{-13}$
2	$2.3\ 10^{-8}$	$3.2\ 10^{-8}$	$1.6\ 10^{-8}$	$1.6\ 10^{-8}$	$5.1\ 10^{-6}$	$9.5\ 10^{-6}$	$2.9\ 10^{-11}$	$2.2\ 10^{-11}$	$4.8\ 10^{-11}$	$2.9\ 10^{-11}$	$9.8\ 10^{-9}$	$8.5\ 10^{-9}$	$1.0\ 10^{-11}$	$5.8\ 10^{-12}$
5	$2.1\ 10^{-8}$	$9.9\ 10^{-8}$	$1.7\ 10^{-8}$	$6.9\ 10^{-8}$	$2.1\ 10^{-6}$	$2.0\ 10^{-5}$	$8.5\ 10^{-11}$	$2.0\ 10^{-10}$	$1.9\ 10^{-10}$	$3.9\ 10^{-10}$	$1.6\ 10^{-8}$	$5.1\ 10^{-8}$	$5.7\ 10^{-11}$	$1.0\ 10^{-10}$
10	$1.4\ 10^{-8}$	$1.9\ 10^{-7}$	$1.2\ 10^{-8}$	$1.4\ 10^{-7}$	$4.6\ 10^{-7}$	$2.5\ 10^{-5}$	$1.4\ 10^{-10}$	$7.8\ 10^{-10}$	$3.3\ 10^{-10}$	$1.7\ 10^{-9}$	$1.3\ 10^{-8}$	$1.3\ 10^{-7}$	$1.3\ 10^{-10}$	$5.8\ 10^{-10}$
20	$4.9\ 10^{-9}$	$2.8\ 10^{-7}$	$4.2\ 10^{-9}$	$2.2\ 10^{-7}$	$2.7\ 10^{-8}$	$2.6\ 10^{-5}$	$1.5\ 10^{-10}$	$2.3\ 10^{-9}$	$3.5\ 10^{-10}$	$5.3\ 10^{-9}$	$5.0\ 10^{-9}$	$2.2\ 10^{-7}$	$1.5\ 10^{-10}$	$2.1\ 10^{-9}$
50	$1.9\ 10^{-10}$	$3.2\ 10^{-7}$	$1.8\ 10^{-10}$	$2.5\ 10^{-7}$	$2.1\ 10^{-10}$	$2.7\ 10^{-5}$	$4.5\ 10^{-11}$	$5.1\ 10^{-9}$	$9.3\ 10^{-11}$	$1.2\ 10^{-8}$	$1.9\ 10^{-10}$	$2.6\ 10^{-7}$	$4.4\ 10^{-11}$	$4.9\ 10^{-9}$
100	$3.4\ 10^{-12}$	$3.2\ 10^{-7}$	$3.8\ 10^{-12}$	$2.5\ 10^{-7}$	$4.0\ 10^{-12}$	$2.7\ 10^{-5}$	$3.3\ 10^{-12}$	$5.9\ 10^{-9}$	$5.6\ 10^{-12}$	$1.3\ 10^{-8}$	$3.3\ 10^{-12}$	$2.6\ 10^{-7}$	$2.8\ 10^{-12}$	$5.6\ 10^{-9}$
200	$1.1\ 10^{-14}$	$3.2\ 10^{-7}$	$1.2\ 10^{-14}$	$2.5\ 10^{-7}$	$1.2\ 10^{-14}$	$2.7\ 10^{-5}$	$1.4\ 10^{-14}$	$5.9\ 10^{-9}$	$1.9\ 10^{-14}$	$1.3\ 10^{-8}$	$1.0\ 10^{-14}$	$2.6\ 10^{-7}$	$1.0\ 10^{-14}$	$5.7\ 10^{-9}$
500	$5.1\ 10^{-22}$	$3.2\ 10^{-7}$	$5.5\ 10^{-22}$	$2.5\ 10^{-7}$	$5.6\ 10^{-22}$	$2.7\ 10^{-5}$	$7.1\ 10^{-22}$	$5.9\ 10^{-9}$	$7.0\ 10^{-22}$	$1.3\ 10^{-8}$	$5.0\ 10^{-22}$	$2.6\ 10^{-7}$	$5.0\ 10^{-22}$	$5.7\ 10^{-9}$
700	$6.9\ 10^{-27}$	$3.2\ 10^{-7}$	$7.2\ 10^{-27}$	$2.5\ 10^{-7}$	$7.3\ 10^{-27}$	$2.7\ 10^{-5}$	$9.0\ 10^{-27}$	$5.9\ 10^{-9}$	$8.6\ 10^{-27}$	$1.3\ 10^{-8}$	$6.8\ 10^{-27}$	$2.6\ 10^{-7}$	$6.8\ 10^{-27}$	$5.7\ 10^{-9}$
1000	$3.7\ 10^{-30}$	$3.2\ 10^{-7}$	$3.9\ 10^{-30}$	$2.5\ 10^{-7}$	$3.9\ 10^{-30}$	$2.7\ 10^{-5}$	$4.9\ 10^{-30}$	$5.9\ 10^{-9}$	$4.6\ 10^{-30}$	$1.3\ 10^{-8}$	$3.7\ 10^{-30}$	$2.6\ 10^{-7}$	$3.7\ 10^{-30}$	$5.7\ 10^{-9}$
10000	$1.2\ 10^{-35}$	$3.2\ 10^{-7}$	$1.3\ 10^{-35}$	$2.5\ 10^{-7}$	$1.3\ 10^{-35}$	$2.7\ 10^{-5}$	$1.6\ 10^{-35}$	$5.9\ 10^{-9}$	$1.5\ 10^{-35}$	$1.3\ 10^{-8}$	$1.2\ 10^{-35}$	$2.6\ 10^{-7}$	$1.2\ 10^{-35}$	$5.7\ 10^{-9}$
1000000	$4.1\ 10^{-52}$	$3.2\ 10^{-7}$	$4.2\ 10^{-52}$	$2.5\ 10^{-7}$	$4.2\ 10^{-52}$	$2.7\ 10^{-5}$	$5.1\ 10^{-52}$	$5.9\ 10^{-9}$	$4.9\ 10^{-52}$	$1.3\ 10^{-8}$	$4.0\ 10^{-52}$	$2.6\ 10^{-7}$	$4.0\ 10^{-52}$	$5.7\ 10^{-9}$
100000000	$1.5\ 10^{-81}$	$3.2\ 10^{-7}$	$1.6\ 10^{-81}$	$2.5\ 10^{-7}$	$1.6\ 10^{-81}$	$2.7\ 10^{-5}$	$1.9\ 10^{-81}$	$5.9\ 10^{-9}$	$1.8\ 10^{-81}$	$1.3\ 10^{-8}$	$1.5\ 10^{-81}$	$2.6\ 10^{-7}$	$1.6\ 10^{-81}$	$5.7\ 10^{-9}$

TABLE 5.9c Specific activity and integrated specific activity of tritium in each compartment following a release of 1 TBq in one year to the atmosphere

Time after release commences (years)	Specific activity and integrated specific activity (Bq kg^{-1}, Bq y kg^{-1})													
	Atmosphere		Surface soil water		Surface streams and freshwater lakes		Saline lakes and inland seas		Deep ground water		Ocean surface		Deep ocean	
1	$2.7\ 10^{-6}$	$2.5\ 10^{-6}$	$1.7\ 10^{-6}$	$1.0\ 10^{-6}$	$2.3\ 10^{-7}$	$8.6\ 10^{-8}$	$2.3\ 10^{-9}$	$1.1\ 10^{-9}$	$3.1\ 10^{-9}$	$1.1\ 10^{-9}$	$2.8\ 10^{-8}$	$1.3\ 10^{-8}$	$1.6\ 10^{-11}$	$5.3\ 10^{-12}$
2	$8.7\ 10^{-8}$	$2.7\ 10^{-6}$	$3.7\ 10^{-7}$	$1.9\ 10^{-6}$	$3.4\ 10^{-7}$	$4.0\ 10^{-7}$	$2.4\ 10^{-9}$	$3.5\ 10^{-9}$	$5.6\ 10^{-9}$	$5.8\ 10^{-9}$	$2.7\ 10^{-8}$	$4.2\ 10^{-8}$	$4.9\ 10^{-11}$	$3.8\ 10^{-11}$
5	$2.1\ 10^{-8}$	$2.8\ 10^{-6}$	$2.0\ 10^{-8}$	$2.2\ 10^{-6}$	$1.7\ 10^{-7}$	$1.2\ 10^{-6}$	$2.2\ 10^{-9}$	$1.1\ 10^{-8}$	$5.4\ 10^{-9}$	$2.3\ 10^{-8}$	$2.0\ 10^{-8}$	$1.1\ 10^{-7}$	$1.2\ 10^{-10}$	$3.1\ 10^{-10}$
10	$1.2\ 10^{-8}$	$2.9\ 10^{-6}$	$1.1\ 10^{-8}$	$2.3\ 10^{-6}$	$4.6\ 10^{-8}$	$1.7\ 10^{-6}$	$1.8\ 10^{-9}$	$2.0\ 10^{-8}$	$4.2\ 10^{-9}$	$4.7\ 10^{-8}$	$1.2\ 10^{-8}$	$1.9\ 10^{-7}$	$1.8\ 10^{-10}$	$1.1\ 10^{-9}$
20	$3.8\ 10^{-9}$	$3.0\ 10^{-6}$	$3.6\ 10^{-9}$	$2.3\ 10^{-6}$	$6.1\ 10^{-9}$	$1.8\ 10^{-6}$	$1.1\ 10^{-9}$	$3.4\ 10^{-8}$	$2.5\ 10^{-9}$	$8.0\ 10^{-8}$	$3.8\ 10^{-9}$	$2.6\ 10^{-7}$	$1.6\ 10^{-10}$	$2.9\ 10^{-9}$
50	$1.7\ 10^{-10}$	$3.0\ 10^{-6}$	$2.3\ 10^{-10}$	$2.4\ 10^{-6}$	$2.6\ 10^{-10}$	$1.9\ 10^{-6}$	$2.3\ 10^{-10}$	$5.1\ 10^{-8}$	$4.5\ 10^{-10}$	$1.2\ 10^{-7}$	$1.6\ 10^{-10}$	$3.0\ 10^{-7}$	$4.2\ 10^{-11}$	$5.7\ 10^{-9}$
100	$4.1\ 10^{-12}$	$3.0\ 10^{-6}$	$8.1\ 10^{-12}$	$2.4\ 10^{-6}$	$8.8\ 10^{-12}$	$1.9\ 10^{-6}$	$1.6\ 10^{-11}$	$5.5\ 10^{-8}$	$2.5\ 10^{-11}$	$1.2\ 10^{-7}$	$3.3\ 10^{-12}$	$3.0\ 10^{-7}$	$2.7\ 10^{-12}$	$6.4\ 10^{-9}$
200	$1.3\ 10^{-14}$	$3.0\ 10^{-6}$	$2.5\ 10^{-14}$	$2.4\ 10^{-6}$	$2.7\ 10^{-14}$	$1.9\ 10^{-6}$	$6.2\ 10^{-14}$	$5.5\ 10^{-8}$	$7.4\ 10^{-14}$	$1.2\ 10^{-7}$	$1.1\ 10^{-14}$	$3.0\ 10^{-7}$	$9.9\ 10^{-15}$	$6.4\ 10^{-9}$
500	$5.7\ 10^{-22}$	$3.0\ 10^{-6}$	$8.6\ 10^{-22}$	$2.4\ 10^{-6}$	$9.0\ 10^{-22}$	$1.9\ 10^{-6}$	$2.4\ 10^{-21}$	$5.5\ 10^{-8}$	$2.1\ 10^{-21}$	$1.2\ 10^{-7}$	$5.1\ 10^{-22}$	$3.0\ 10^{-7}$	$4.9\ 10^{-22}$	$6.4\ 10^{-9}$
700	$7.5\ 10^{-27}$	$3.0\ 10^{-6}$	$9.9\ 10^{-27}$	$2.4\ 10^{-6}$	$1.0\ 10^{-26}$	$1.9\ 10^{-6}$	$2.5\ 10^{-26}$	$5.5\ 10^{-8}$	$2.0\ 10^{-26}$	$1.2\ 10^{-7}$	$6.9\ 10^{-27}$	$3.0\ 10^{-7}$	$6.8\ 10^{-27}$	$6.4\ 10^{-9}$
1000	$4.0\ 10^{-30}$	$3.0\ 10^{-6}$	$5.4\ 10^{-30}$	$2.4\ 10^{-6}$	$5.6\ 10^{-30}$	$1.9\ 10^{-6}$	$1.4\ 10^{-29}$	$5.5\ 10^{-8}$	$1.1\ 10^{-29}$	$1.2\ 10^{-7}$	$3.7\ 10^{-30}$	$3.0\ 10^{-7}$	$3.6\ 10^{-30}$	$6.4\ 10^{-9}$
10000	$1.3\ 10^{-35}$	$3.0\ 10^{-6}$	$1.7\ 10^{-35}$	$2.4\ 10^{-6}$	$1.8\ 10^{-35}$	$1.9\ 10^{-6}$	$4.3\ 10^{-35}$	$5.5\ 10^{-8}$	$3.5\ 10^{-35}$	$1.2\ 10^{-7}$	$1.2\ 10^{-35}$	$3.0\ 10^{-7}$	$1.2\ 10^{-35}$	$6.4\ 10^{-9}$
1000000	$4.3\ 10^{-52}$	$3.0\ 10^{-6}$	$5.5\ 10^{-52}$	$2.4\ 10^{-6}$	$5.7\ 10^{-52}$	$1.9\ 10^{-6}$	$1.3\ 10^{-51}$	$5.5\ 10^{-8}$	$1.0\ 10^{-51}$	$1.2\ 10^{-7}$	$4.1\ 10^{-52}$	$3.0\ 10^{-7}$	$4.0\ 10^{-52}$	$6.4\ 10^{-9}$
100000000	$1.6\ 10^{-81}$	$3.0\ 10^{-6}$	$1.9\ 10^{-81}$	$2.4\ 10^{-6}$	$2.0\ 10^{-81}$	$1.9\ 10^{-6}$	$3.7\ 10^{-81}$	$5.5\ 10^{-8}$	$3.2\ 10^{-81}$	$1.2\ 10^{-7}$	$1.6\ 10^{-81}$	$3.0\ 10^{-7}$	$1.6\ 10^{-81}$	$6.4\ 10^{-9}$

TABLE 5.10a Specific activity and integrated specific activity of iodine-129 in sections of the environment from which man derives his iodine intake following the release of 1 TBq in one year to the land atmosphere

Time after release commences (years)	Specific activity and integrated specific activity (Bq kg^{-1}, Bq y kg^{-1}) in the:			
	Land atmosphere	Surface oceans	Land surface waters	Soil water
1	$5.7\ 10^{3}$ 5060	$2.7\ 10^{-1}$ 0.115	$5.6\ 10^{2}$ 449	$8.2\ 10^{2}$ 661
2	$3.5\ 10^{0}$ 5700	$3.0\ 10^{-1}$ 0.425	$1.1\ 10^{0}$ 558	$1.5\ 10^{0}$ 820
5	$3.9\ 10^{-1}$ 5700	$1.8\ 10^{-1}$ 1.12	$4.2\ 10^{-1}$ 559	$4.7\ 10^{-1}$ 822
10	$2.4\ 10^{-1}$ 5700	$7.9\ 10^{-2}$ 1.73	$4.1\ 10^{-1}$ 561	$4.5\ 10^{-1}$ 824
20	$1.2\ 10^{-1}$ 5700	$1.9\ 10^{-2}$ 2.14	$3.9\ 10^{-1}$ 565	$4.3\ 10^{-1}$ 829
50	$2.9\ 10^{-2}$ 5700	$6.6\ 10^{-3}$ 2.41	$3.8\ 10^{-1}$ 577	$4.2\ 10^{-1}$ 841
100	$1.1\ 10^{-2}$ 5700	$6.6\ 10^{-3}$ 2.74	$3.8\ 10^{-1}$ 596	$4.1\ 10^{-1}$ 862
200	$9.1\ 10^{-3}$ 5700	$6.7\ 10^{-3}$ 3.41	$3.7\ 10^{-1}$ 633	$4.0\ 10^{-1}$ 903
500	$9.2\ 10^{-3}$ 5710	$7.0\ 10^{-3}$ 5.47	$3.4\ 10^{-1}$ 739	$3.8\ 10^{-1}$ 1020
700	$9.3\ 10^{-3}$ 5710	$7.2\ 10^{-3}$ 6.89	$3.2\ 10^{-1}$ 808	$3.7\ 10^{-1}$ 1100
1000	$9.5\ 10^{-3}$ 5710	$7.5\ 10^{-3}$ 9.08	$3.0\ 10^{-1}$ 897	$3.5\ 10^{-1}$ 1200
10000	$1.1\ 10^{-2}$ 5810	$1.1\ 10^{-2}$ 96.1	$5.4\ 10^{-2}$ 2160	$6.6\ 10^{-2}$ 2710
1000000	$2.0\ 10^{-3}$ 9600	$2.1\ 10^{-3}$ 3900	$8.8\ 10^{-4}$ 4040	$1.1\ 10^{-3}$ 5040
100000000	$8.0\ 10^{-7}$ 14100	$8.0\ 10^{-7}$ 8450	$8.0\ 10^{-7}$ 6730	$8.0\ 10^{-7}$ 8060

TABLE 5.10b Specific activity and integrated specific activity of iodine-129 in sections of the environment from which man derives his iodine intake following the release of 1 TBq in one year to the surface oceans

Time after release commences (years)	Specific activity and integrated specific activity (Bq kg^{-1}, Bq y kg^{-1}) in the:			
	Land atmosphere	Surface oceans	Land surface waters	Soil water
1	$5.4\ 10^{-1}$ 0.2276	$6.8\ 10^{-1}$ 0.3502	$4.8\ 10^{-2}$ 0.018286	$7.0\ 10^{-2}$ 0.02689
2	$5.9\ 10^{-1}$ 0.8471	$5.7\ 10^{-1}$ 0.9764	$5.9\ 10^{-2}$ 0.078074	$8.7\ 10^{-2}$ 0.1148
5	$3.6\ 10^{-1}$ 2.241	$3.5\ 10^{-1}$ 2.329	$3.6\ 10^{-2}$ 0.216809	$5.2\ 10^{-2}$ 0.3187
10	$1.6\ 10^{-1}$ 3.456	$1.5\ 10^{-1}$ 3.508	$1.6\ 10^{-2}$ 0.338381	$2.3\ 10^{-2}$ 0.4971
20	$3.8\ 10^{-2}$ 4.267	$3.7\ 10^{-2}$ 4.298	$4.0\ 10^{-3}$ 0.42143	$5.8\ 10^{-3}$ 0.6182
50	$1.2\ 10^{-2}$ 4.779	$1.2\ 10^{-2}$ 4.805	$1.5\ 10^{-3}$ 0.480959	$2.1\ 10^{-3}$ 0.7022
100	$1.2\ 10^{-2}$ 5.386	$1.2\ 10^{-2}$ 5.415	$1.5\ 10^{-3}$ 0.557292	$2.1\ 10^{-3}$ 0.808
200	$1.2\ 10^{-2}$ 6.6	$1.2\ 10^{-2}$ 6.63	$1.6\ 10^{-3}$ 0.71829	$2.2\ 10^{-3}$ 1.03
500	$1.2\ 10^{-2}$ 10.22	$1.2\ 10^{-2}$ 10.28	$1.8\ 10^{-3}$ 1.229764	$2.4\ 10^{-3}$ 1.724
700	$1.2\ 10^{-2}$ 12.6	$1.2\ 10^{-2}$ 12.7	$1.9\ 10^{-3}$ 1.607421	$2.6\ 10^{-3}$ 2.23
1000	$1.2\ 10^{-2}$ 16.3	$1.2\ 10^{-2}$ 16.3	$2.1\ 10^{-3}$ 2.21171	$2.8\ 10^{-3}$ 3.03
10000	$1.1\ 10^{-2}$ 121.9	$1.1\ 10^{-2}$ 122.4	$4.4\ 10^{-3}$ 34.5959	$5.5\ 10^{-3}$ 44.03
1000000	$2.0\ 10^{-3}$ 3890	$2.0\ 10^{-3}$ 3900	$8.7\ 10^{-4}$ 1637.68	$1.1\ 10^{-3}$ 2040
100000000	$8.0\ 10^{-7}$ 8428	$8.0\ 10^{-7}$ 8446	$8.0\ 10^{-7}$ 4331.52	$8.0\ 10^{-7}$ 5068

TABLE 5.11a Specific activity and integrated specific activity of carbon-14 in sections of the environment from which man derives his carbon intake following a release of 1 TBq in one year to the atmosphere

Time after release	Specific activity and integrated specific activity Bq kg^{-1} carbon, Bq y kg^{-1} carbon in:									
commences (years)	Ground Vegetation		Atlantic Ocean Surface		Pacific Ocean Surface		Arctic Ocean Surface		Antarctic Ocean Surface	
1	$2.7\ 10^{-4}$	$9.6\ 10^{-5}$	$1.1\ 10^{-5}$	$3.9\ 10^{-6}$	$1.2\ 10^{-5}$	$3.9\ 10^{-6}$	$1.1\ 10^{-5}$	$3.7\ 10^{-6}$	$9.7\ 10^{-6}$	$3.4\ 10^{-6}$
2	$5.9\ 10^{-4}$	$5.5\ 10^{-4}$	$3.0\ 10^{-5}$	$2.5\ 10^{-5}$	$3.1\ 10^{-5}$	$2.6\ 10^{-5}$	$2.7\ 10^{-5}$	$2.3\ 10^{-5}$	$2.1\ 10^{-5}$	$2.0\ 10^{-5}$
5	$6.8\ 10^{-4}$	$2.6\ 10^{-3}$	$6.5\ 10^{-5}$	$1.7\ 10^{-4}$	$6.8\ 10^{-5}$	$1.8\ 10^{-4}$	$5.2\ 10^{-5}$	$1.5\ 10^{-4}$	$3.0\ 10^{-5}$	$1.0\ 10^{-4}$
10	$4.2\ 10^{-4}$	$5.3\ 10^{-3}$	$8.8\ 10^{-5}$	$5.7\ 10^{-4}$	$9.8\ 10^{-5}$	$6.1\ 10^{-4}$	$5.7\ 10^{-5}$	$4.3\ 10^{-4}$	$3.3\ 10^{-5}$	$2.6\ 10^{-4}$
20	$2.1\ 10^{-4}$	$8.2\ 10^{-3}$	$9.1\ 10^{-5}$	$1.5\ 10^{-3}$	$1.1\ 10^{-4}$	$1.7\ 10^{-3}$	$4.6\ 10^{-5}$	$9.4\ 10^{-4}$	$3.7\ 10^{-5}$	$6.1\ 10^{-4}$
50	$1.1\ 10^{-4}$	$1.2\ 10^{-2}$	$7.1\ 10^{-5}$	$3.9\ 10^{-3}$	$9.5\ 10^{-5}$	$4.9\ 10^{-3}$	$4.0\ 10^{-5}$	$2.2\ 10^{-3}$	$4.0\ 10^{-5}$	$1.8\ 10^{-3}$
100	$7.0\ 10^{-5}$	$1.7\ 10^{-2}$	$5.6\ 10^{-5}$	$7.0\ 10^{-3}$	$6.5\ 10^{-5}$	$8.8\ 10^{-3}$	$4.4\ 10^{-5}$	$4.3\ 10^{-3}$	$3.9\ 10^{-5}$	$3.8\ 10^{-3}$
200	$4.6\ 10^{-5}$	$2.2\ 10^{-2}$	$4.4\ 10^{-5}$	$1.2\ 10^{-2}$	$4.2\ 10^{-5}$	$1.4\ 10^{-2}$	$4.2\ 10^{-5}$	$8.7\ 10^{-3}$	$3.3\ 10^{-5}$	$7.4\ 10^{-3}$
500	$2.9\ 10^{-5}$	$3.3\ 10^{-2}$	$3.0\ 10^{-5}$	$2.3\ 10^{-2}$	$2.7\ 10^{-5}$	$2.4\ 10^{-2}$	$3.0\ 10^{-5}$	$1.9\ 10^{-2}$	$2.6\ 10^{-5}$	$1.6\ 10^{-2}$
700	$2.5\ 10^{-5}$	$3.8\ 10^{-2}$	$2.6\ 10^{-5}$	$2.8\ 10^{-2}$	$2.4\ 10^{-5}$	$2.9\ 10^{-2}$	$2.6\ 10^{-5}$	$2.5\ 10^{-2}$	$2.4\ 10^{-5}$	$2.1\ 10^{-2}$
1000	$2.2\ 10^{-5}$	$4.5\ 10^{-2}$	$2.2\ 10^{-5}$	$3.5\ 10^{-2}$	$2.2\ 10^{-5}$	$3.6\ 10^{-2}$	$2.2\ 10^{-5}$	$3.2\ 10^{-2}$	$2.2\ 10^{-5}$	$2.8\ 10^{-2}$
10000	$7.0\ 10^{-6}$	$1.6\ 10^{-1}$	$7.0\ 10^{-6}$	$1.5\ 10^{-1}$	$7.0\ 10^{-6}$	$1.5\ 10^{-1}$	$6.9\ 10^{-6}$	$1.5\ 10^{-1}$	$7.0\ 10^{-6}$	$1.4\ 10^{-1}$
1000000	$0.0\ 10^{0}$	$2.2\ 10^{-1}$	$0.0\ 10^{0}$	$2.1\ 10^{-1}$	$0.0\ 10^{0}$	$2.1\ 10^{-1}$	$0.0\ 10^{0}$	$2.0\ 10^{-1}$	$0.0\ 10^{0}$	$2.0\ 10^{-1}$
100000000	$0.0\ 10^{0}$	$2.2\ 10^{-1}$	$0.0\ 10^{0}$	$2.1\ 10^{-1}$	$0.0\ 10^{0}$	$2.1\ 10^{-1}$	$0.0\ 10^{0}$	$2.0\ 10^{-1}$	$0.0\ 10^{0}$	$2.0\ 10^{-1}$

TABLE 5.11b Specific activity and integrated specific activity of carbon-14 in sections of the environment from which man derives his carbon intake following a release of 1 TBq in one year to the Atlantic surface

Time after release commences (years)	Specific activity and integrated specific activity Bq kg^{-1} carbon, Bq y kg^{-1} carbon in:									
	Ground Vegetation		Atlantic Ocean Surface		Pacific Ocean Surface		Arctic Ocean Surface		Antarctic Ocean Surface	
1	$1.8\ 10^{-6}$	$4.8\ 10^{-7}$	$9.3\ 10^{-4}$	$4.7\ 10^{-4}$	$6.5\ 10^{-7}$	$2.1\ 10^{-7}$	$2.0\ 10^{-7}$	$5.2\ 10^{-8}$	$1.2\ 10^{-7}$	$3.6\ 10^{-8}$
2	$1.0\ 10^{-5}$	$6.1\ 10^{-6}$	$8.7\ 10^{-4}$	$1.4\ 10^{-3}$	$2.2\ 10^{-6}$	$1.6\ 10^{-6}$	$1.3\ 10^{-6}$	$7.3\ 10^{-7}$	$5.0\ 10^{-7}$	$3.3\ 10^{-7}$
5	$4.6\ 10^{-5}$	$9.1\ 10^{-5}$	$7.4\ 10^{-4}$	$3.8\ 10^{-3}$	$7.6\ 10^{-6}$	$1.6\ 10^{-5}$	$9.0\ 10^{-6}$	$1.5\ 10^{-5}$	$2.1\ 10^{-6}$	$4.2\ 10^{-6}$
10	$8.1\ 10^{-5}$	$4.2\ 10^{-4}$	$5.7\ 10^{-4}$	$7.0\ 10^{-3}$	$1.7\ 10^{-5}$	$7.8\ 10^{-5}$	$2.7\ 10^{-5}$	$1.0\ 10^{-4}$	$4.9\ 10^{-6}$	$2.2\ 10^{-5}$
20	$9.2\ 10^{-5}$	$1.3\ 10^{-3}$	$3.5\ 10^{-4}$	$1.2\ 10^{-2}$	$3.2\ 10^{-5}$	$3.3\ 10^{-4}$	$6.0\ 10^{-5}$	$5.5\ 10^{-4}$	$1.0\ 10^{-5}$	$9.8\ 10^{-5}$
50	$6.9\ 10^{-5}$	$3.7\ 10^{-3}$	$1.4\ 10^{-4}$	$1.8\ 10^{-2}$	$4.7\ 10^{-5}$	$1.6\ 10^{-3}$	$9.1\ 10^{-5}$	$3.0\ 10^{-3}$	$2.1\ 10^{-5}$	$5.8\ 10^{-4}$
100	$5.2\ 10^{-5}$	$6.7\ 10^{-3}$	$7.7\ 10^{-5}$	$2.3\ 10^{-2}$	$4.2\ 10^{-5}$	$3.9\ 10^{-3}$	$8.2\ 10^{-5}$	$7.4\ 10^{-3}$	$2.8\ 10^{-5}$	$1.8\ 10^{-3}$
200	$3.9\ 10^{-5}$	$1.1\ 10^{-2}$	$5.0\ 10^{-5}$	$2.9\ 10^{-2}$	$3.4\ 10^{-5}$	$7.6\ 10^{-3}$	$5.7\ 10^{-5}$	$1.4\ 10^{-2}$	$2.9\ 10^{-5}$	$4.7\ 10^{-3}$
500	$2.7\ 10^{-5}$	$2.1\ 10^{-2}$	$2.9\ 10^{-5}$	$4.0\ 10^{-2}$	$2.5\ 10^{-5}$	$1.6\ 10^{-2}$	$3.0\ 10^{-5}$	$2.6\ 10^{-2}$	$2.5\ 10^{-5}$	$1.3\ 10^{-2}$
700	$2.4\ 10^{-5}$	$2.6\ 10^{-2}$	$2.5\ 10^{-5}$	$4.5\ 10^{-2}$	$2.3\ 10^{-5}$	$2.1\ 10^{-2}$	$2.5\ 10^{-5}$	$3.2\ 10^{-2}$	$2.3\ 10^{-5}$	$1.8\ 10^{-2}$
1000	$2.1\ 10^{-5}$	$3.2\ 10^{-2}$	$2.2\ 10^{-5}$	$5.2\ 10^{-2}$	$2.1\ 10^{-5}$	$2.8\ 10^{-2}$	$2.2\ 10^{-5}$	$3.9\ 10^{-2}$	$2.1\ 10^{-5}$	$2.5\ 10^{-2}$
10000	$7.0\ 10^{-6}$	$1.5\ 10^{-1}$	$7.0\ 10^{-6}$	$1.7\ 10^{-1}$	$7.0\ 10^{-6}$	$1.4\ 10^{-1}$	$6.9\ 10^{-6}$	$1.5\ 10^{-1}$	$7.0\ 10^{-6}$	$1.4\ 10^{-1}$
1000000	$1.1\ 10^{-19}$	$2.0\ 10^{-1}$	$1.1\ 10^{-19}$	$2.2\ 10^{-1}$	$1.2\ 10^{-19}$	$2.0\ 10^{-1}$	$1.1\ 10^{-19}$	$2.1\ 10^{-1}$	$1.1\ 10^{-19}$	$2.0\ 10^{-1}$
100000000	$8.4\ 10^{-32}$	$2.0\ 10^{-1}$	$8.4\ 10^{-32}$	$2.2\ 10^{-1}$	$8.4\ 10^{-32}$	$2.0\ 10^{-1}$	$8.3\ 10^{-32}$	$2.1\ 10^{-1}$	$8.4\ 10^{-32}$	$2.0\ 10^{-1}$

Table 5.12 Population data

Country	Population	Reference
Austria #	$8.12\ 10^6$	http://epp.eurostat.ec.europa.eu Data for 2003
Belgium/Luxembourg # +	$1.04\ 10^7$/$4.50\ 10^5$	"
Cyprus #	$7.23\ 10^5$	"
Czech Republic #	$1.02\ 10^7$	"
Denmark # +	$5.39\ 10^6$	"
Estonia #	$1.35\ 10^6$	"
Finland #	$5.21\ 10^6$	"
France # +	$6.00\ 10^7$	"
Germany # +	$8.25\ 10^7$	"
Greece # +	$1.10\ 10^7$	"
Hungary #	$1.01\ 10^7$	"
Ireland # +	$4.00\ 10^6$	"
Italy # +	$5.76\ 10^7$	"
Latvia #	$2.33\ 10^6$	"
Lithuania #	$3.45\ 10^6$	"
Malta #	$3.99\ 10^5$	"
Netherlands # +	$1.62\ 10^7$	"
Poland #	$3.82\ 10^7$	"
Portugal # +	$1.04\ 10^7$	"
Slovakia #	$5.38\ 10^6$	"
Slovenia #	$2.00\ 10^6$	"
Spain # +	$4.20\ 10^7$	"
Sweden #	$8.96\ 10^6$	"
United Kingdom # +	$5.96\ 10^7$	"
EU25	$4.56\ 10^8$	"
EU12	$3.60\ 10^8$	"
World	$1.00\ 10^{10}$	http://esa.un.org/unpp/ United Nations medium estimate for 2050 is $9\ 10^9$.

+ Countries included in EU12

Countries included in EU25

5.7 Figures

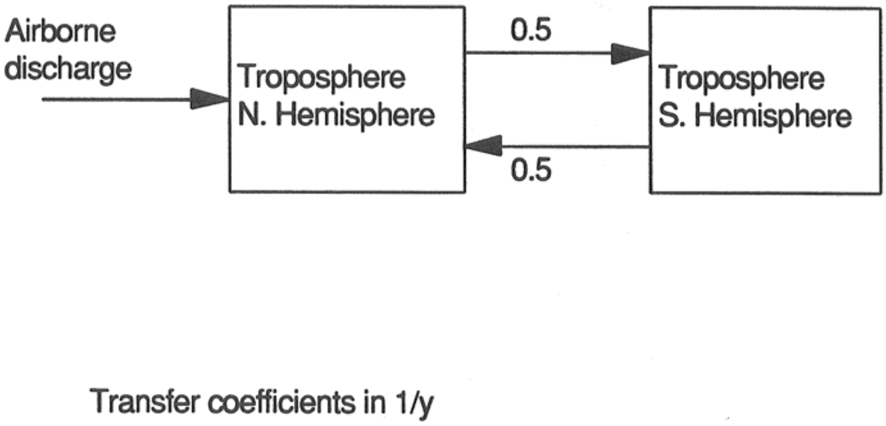

FIGURE 5.1 Model for global circulation of krypton-85

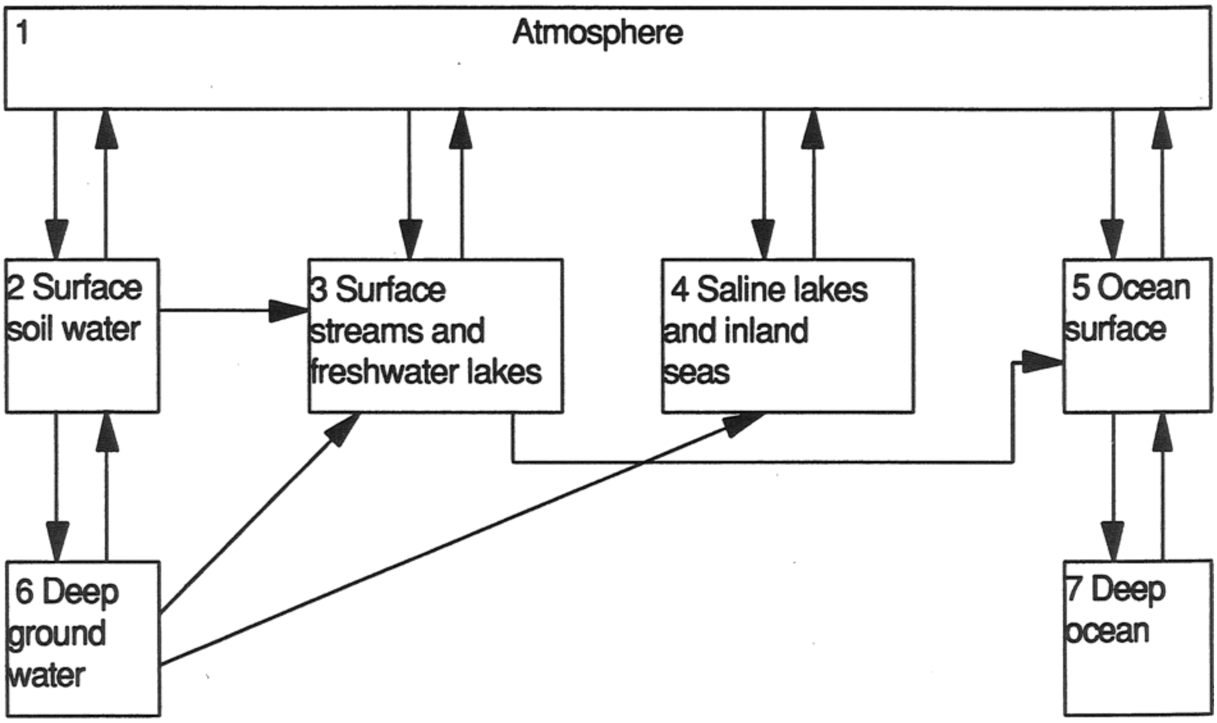

FIGURE 5.2 Model for global circulation of tritium (see also Table 5.2)

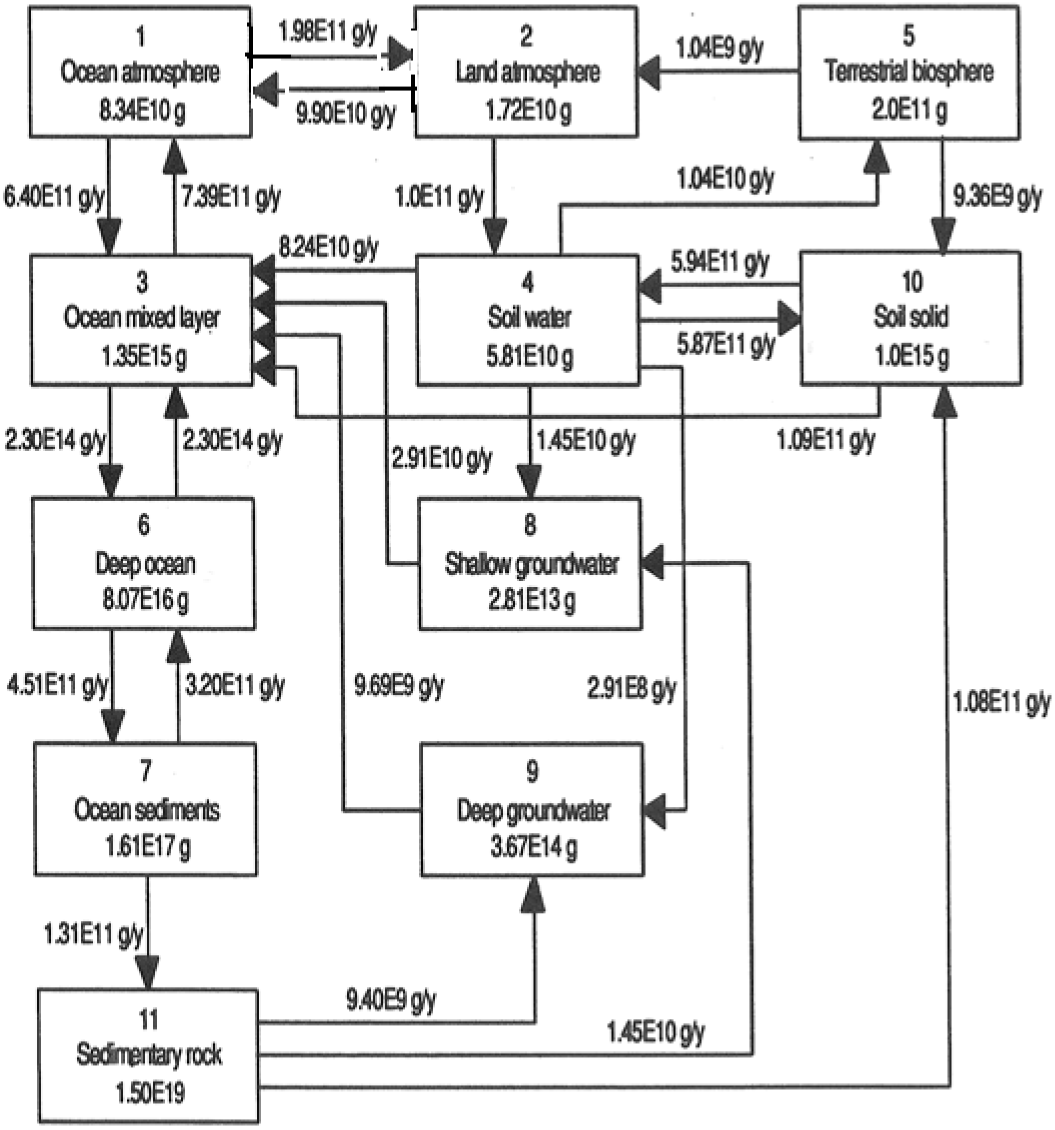

FIGURE 5.3 Model for global circulation of iodine-129

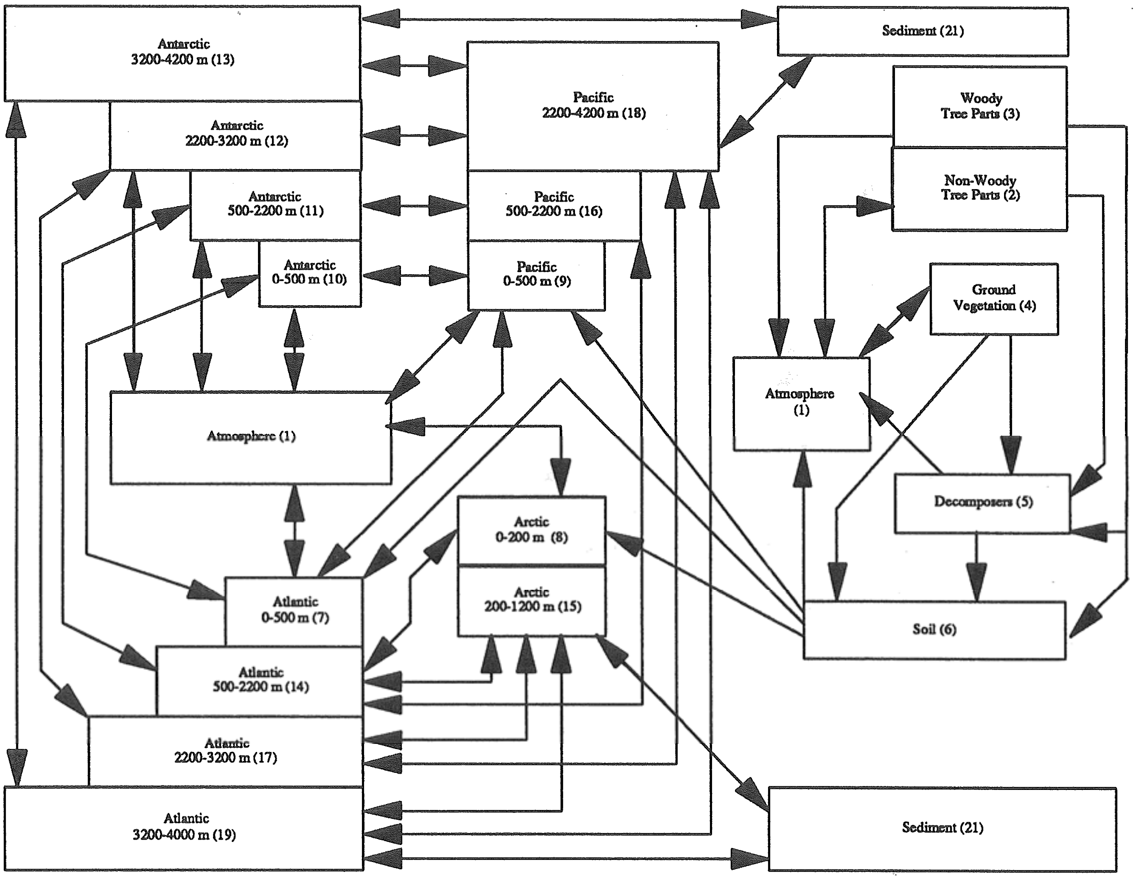

FIGURE 5.4 Model for global circulation of carbon-14 (see also Table 5.7)

6 MODELS FOR INTERNAL DOSIMETRY

In estimating exposure due to external irradiation from the discharge of radionuclides to the environment, the appropriate models are described in Chapters 3 and 4 for releases to atmosphere and to the aquatic environment, respectively. However, these chapters only considered the quantity of activity inhaled or ingested. The irradiation of the body and its various organs from incorporated radionuclides can be determined as the product of these intakes and the appropriate dose coefficients, representing the doses per unit intake from ingestion or inhalation. Dose coefficients for internal irradiation are calculated and published by the International Commission on Radiological Protection (ICRP, 1996). The models used to determine the effective dose per unit intake by inhalation or ingestion are outlined below, although reference should be made to the original ICRP publications for a detailed account of the models employed.

The dose coefficients currently included in PC-CREAM 08 are for effective dose as defined in ICRP Publication 60 (ICRP, 1991). They are the same as those used in the European Directive laying down the basic safety standards for radiological protection in the EC (European Commission, 1996). ICRP has recently published new recommendations (ICRP, 2007a) which have a slightly modified definition of effective dose. However, ICRP is yet to publish revised dose coefficients for internal irradiation. When such data are available they can be substituted in PC-CREAM 08.

6.1 Calculation of equivalent dose to organs of the body from unit intakes of radionuclides by ingestion or inhalation

6.1.1 Introduction

As radioactive material progresses through the body, organs are irradiated both from the radiations resulting from transformations occurring in the organs themselves and from those occurring in surrounding organs. Irradiated organs are referred to as target organs and those in which transformations occur are referred to as source organs.

There are four main steps necessary to estimate the doses delivered in any organ from a given radionuclide in a given period of time.

(i) The computation of the number of nuclear transformations of the inhaled (or ingested) radionuclide in each of the source organs, ie, those through which the radioactive material passes. A similar computation is necessary for any radioactive progeny formed following intake of the parent nuclide.
(ii) The preparation of matrices (one for each radionuclide involved) containing the absorbed doses received by the target organs per transformation in the source organs.
(iii) The conversion of the absorbed doses in the matrices evaluated in (ii) to equivalent doses; radiation weighting factors of 1 and 20 are used for low (β,γ) and high (α) LET radiations respectively (ICRP, 1991).
(iv) The combination of (i) and (iii) to obtain the equivalent dose in each target organ from the inhaled (or ingested) parent and any progeny radionuclides.

Approximately 30 organs are considered as source and/or target organs. In addition to the estimation of dose equivalents in organs the effective dose is also calculated according to the procedure described in ICRP Publication 60 (ICRP, 1991) and detailed in Chapter 2.

6.1.2 Dose coefficients for intakes of radionuclides

The International Commission on Radiological Protection (ICRP) provides a system of protection against the risks from exposure to ionising radiation (ICRP, 1991; 2007a). An important component of that system is the provision of biokinetic and dosimetric models for the assessment of doses from the intakes of radionuclides. A series of reports (ICRP Publications 56, 67, 69 and 71 (ICRP 1989, 1993b, 1995a, 1995b), culminating in a compilation report ICRP 72 (ICRP, 1996)) gave dose coefficients for the ingestion or inhalation of selected radioisotopes of 31 elements by 3 month old infants, children aged 1, 5, 10 and 15 years and adult members of the public. In each case, the values given are of committed effective dose per Bq ingested or inhaled by the infant, child or adult, integrated to age 70 years. Additional dose coefficients for exposure of the fetus and the breastfed infant following an intake of radionuclides by the mother have more recently been reported by ICRP (ICRP, 2001; ICRP 2004).

In the ICRP (1994) model of the human respiratory tract, three inhalation classes are considered, S, M and F, which have slow, medium and fast clearance times, respectively for the absorption of activity to blood. For ingested material the fraction of each element reaching blood is determined by the gut transfer fraction. The fraction of each element transferred from blood to each organ and the retention of the element in these organs are given in the relevant ICRP publication (see ICRP, 1996 for a full reference list)

The number of transformations in a source organ is evaluated from the distribution of each radionuclide among the various organs and their retention. Resulting non-penetrating radiations (α, β) are generally assumed to be absorbed in the source organ. However, for the skeleton, two radiosensitive tissues, red bone marrow and endosteal cells, are considered separately (see, for example, ICRP, 1989). For penetrating radiations (γ) only a fraction of the energy emitted in the source organ is deposited in that organ and further fractions are deposited in other organs (see, for example, ICRP, 1989).

Equivalent doses in each organ are derived from the absorbed dose in each organ using the appropriate radiation weighting factors (ICRP, 1991); see Chapter 2. Effective doses are then obtained by summing the equivalent organ doses weighted by the appropriate tissue weighting factors (ICRP, 1991), given in Chapter 2.

Effective doses for 1 and 10 year old children and adults are considered in this report. For children age-dependent biokinetics are taken into account (see ICRP, 1996 for a list of relevant references for each radioelement) and the doses are integrated to age 70 from the time of intake.

The committed effective doses per unit intake (dose coefficients) are summarised in Tables 6.1 and 6.2 for intake by ingestion and inhalation, respectively. For illustration purposes only those radionuclides that are included in the common nuclide list within PC-CREAM 08 are listed here. Doses coefficients are given for 1 and 10 year old children and for adults. Although dose coefficients for other age groups (3 months, 5 years and 15 years) are also available, current advice from ICRP (2007b) is that generally only doses to 1 year olds, 10 year olds and adults need be considered when assessing the dose to the general population and, hence, these are the only ones included by default within the model. A report from the HPA (HPA, 2008) provides guidance on the inclusion of the fetus and the breastfed infant within a dose assessment and recommends that for most radionuclides an assessment would not need to include these age groups. This is because the dose to one of the age groups normally considered, namely 1 year olds, 10 year olds or adults, would either be higher than the dose to the fetus or breastfed infant or would be sufficiently similar as to not warrant the additional effort required to assess them individually. However, for four radionuclides, ^{32}P, ^{33}P, ^{45}Ca and ^{89}Sr, the recommendation is to always assess the dose to the fetus and the breastfed infant as the dose to these age groups could be up to an order of magnitude higher than that to the other age groups. Although PC-CREAM 08 does not include the ability to assess the dose to the fetus or the breastfed infant directly, guidance on using PC-CREAM 08 to assess the dose to these groups for these radionuclides is given in Section 7.5.

The values presented in Tables 6.1 and 6.2 are taken from ICRP publication 72 (ICRP, 1996). Tables 6.1 and 6.2 also present the default gut transfer factor and the lung class assumed within PC-CREAM 08, these being the same as those recommended by ICRP (ICRP, 1996) for an unspecified material. Although ICRP recommendations are generally used within PC-CREAM 08 it is worth noting that for some radionuclides additional assumptions have been made, as follows.

Tritium

Tritium emits weak beta radiation that does not penetrate the outer layer of the skin. Thus exposure occurs only when tritium has entered the body. This could occur via three mechanisms: inhalation of water vapour, ingestion of water or as a result of the physical absorption of tritium through the skin.

The ICRP inhalation dose coefficient (ICRP, 1996) for tritium does not take into account skin absorption. PC-CREAM 08 assesses the dose from skin absorption of tritium by the use of a multiplication factor applied to the inhalation dose. From information presented in ICRP publication 71 (ICRP, 1995b), which states that for a sedentary adult approximately one third of the total tritium intake for a given atmospheric concentration is attributable to absorption through the skin, a multiplication factor equal to 1.5 was derived for use within PC-CREAM 08. This factor represents a rounded value for use with all age groups assuming that the CREAM default inhalation rates are used. If these inhalation rates are changed then this tritium absorption factor may also need to be revised.

Carbon-14

PC-CREAM 08 assumes that inhalation of carbon-14 occurs in particulate form. Previously PC-CREAM 98 (Mayall et al, 1997) assumed that inhalation of carbon-14 was 50% particulate and 50% vapour and any comparison with results calculated using PC-CREAM 98 should take this change into account.

Radon

At the time of writing this report the Main Commission of ICRP is in the process of developing dose coefficients for inhalation of radon isotopes. Consequently, the method previously used in PC-CREAM 98 to estimate the inhalation dose coefficient for radon-222 progeny has been used in PC-CREAM 08. The approach uses a dose coefficient for radon exposure in homes of 1.1 Sv per J h m^{-3} and a conversion factor of 5.56 10^{-9} J h m^{-3} per Bq h m^{-3} both taken from ICRP publication 65 (ICRP, 1993a). This gives a dose coefficient for radon and its progeny of approximately 6 nSv per Bq h m^{-3}. An equilibrium concentration factor, F, is also applied to this value to take account of the disequilibrium that may exist between radon and its progeny. UNSCEAR (United Nations Scientific Committee on the Effects of Atomic Radiation) (UN, 2000) recommends a value for F of between 0.2 and 1.0 in the outdoor environment. In PC-CREAM 08 a value of 1 is used which is a conservative estimate and means that the progeny are in equilibrium with their parent nuclide. This is a reasonable assumption given that the precise location of the individuals exposed is unknown. This dose coefficient is used for all age groups.

The dose coefficient for thoron (^{220}Rn) used in PC-CREAM 08 is taken from UNSCEAR (UN, 2000) which gives 40 nSv per Bq h m^{-3} for a concentration of thoron in equilibrium with its progeny. This dose coefficient is used for all age groups.

6.2 References

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6.3 Tables

TABLE 6.1 Committed effective dose per unit intake by ingestion (Sv per Bq) of the "common" radionuclides within PC-CREAM 08 (from ICRP, 1996)

Radionuclide^{a}	Fractional absorption, F_1	1 year old	10 year old	Adult
^{3}H	1	4.8 10^{-11}	2.3 10^{-11}	1.8 10^{-11}
^{14}C	1	1.6 10^{-9}	8.0 10^{-10}	5.8 10^{-10}
^{32}P	0.8	1.9 10^{-8}	5.3 10^{-9}	2.4 10^{-9}
^{33}P	0.8	1.8 10^{-9}	5.3 10^{-10}	2.4 10^{-10}
^{35}S	1	5.4 10^{-9}	1.6 10^{-9}	7.7 10^{-10}
^{51}Cr	0.1	2.3 10^{-10}	7.8 10^{-11}	3.8 10^{-11}
^{54}Mn	0.1	3.1 10^{-9}	1.3 10^{-9}	7.1 10^{-10}
^{59}Fe	0.1	1.3 10^{-8}	4.7 10^{-9}	1.8 10^{-9}
^{58}Co	0.1	4.4 10^{-9}	1.7 10^{-9}	7.4 10^{-10}
^{60}Co	0.1	2.7 10^{-8}	1.1 10^{-8}	3.4 10^{-9}
^{65}Zn	0.5	1.6 10^{-8}	6.4 10^{-9}	3.9 10^{-9}
^{89}Sr	0.3	1.8 10^{-8}	5.8 10^{-9}	2.6 10^{-9}
^{90}Sr	0.3	7.3 10^{-8}	6.0 10^{-8}	2.8 10^{-8}
^{95}Zr	0.01	5.6 10^{-9}	1.9 10^{-9}	9.5 10^{-10}
^{95}Nb	0.01	3.2 10^{-9}	1.1 10^{-9}	5.8 10^{-10}
^{106}Ru	0.05	4.9 10^{-8}	1.5 10^{-8}	7.0 10^{-9}
^{122}Sb	0.1	1.2 10^{-8}	3.7 10^{-9}	1.7 10^{-9}
^{124}Sb	0.1	1.6 10^{-8}	5.2 10^{-9}	2.5 10^{-9}
^{125}Sb	0.1	6.1 10^{-9}	2.1 10^{-9}	1.1 10^{-9}
^{129}I	1	2.2 10^{-7}	1.9 10^{-7}	1.1 10^{-7}
^{131}I	1	1.8 10^{-7}	5.2 10^{-8}	2.2 10^{-8}
^{132}I	1	2.4 10^{-9}	6.2 10^{-10}	2.9 10^{-10}
^{133}I	1	4.4 10^{-8}	1.0 10^{-8}	4.3 10^{-9}
^{135}I	1	8.9 10^{-9}	2.2 10^{-9}	9.3 10^{-10}
^{134}Cs	1	1.6 10^{-8}	1.4 10^{-8}	1.9 10^{-8}
^{137}Cs	1	1.2 10^{-8}	1.0 10^{-8}	1.3 10^{-8}
^{140}Ba	0.2	1.8 10^{-8}	5.8 10^{-9}	2.6 10^{-9}
^{140}La	0.0005	1.3 10^{-8}	4.2 10^{-9}	2.0 10^{-9}
^{141}Ce	0.0005	5.1 10^{-9}	1.5 10^{-9}	7.1 10^{-10}
^{144}Ce	0.0005	3.9 10^{-8}	1.1 10^{-8}	5.2 10^{-9}
^{195m}Pb	0.2	1.6 10^{-10}	5.2 10^{-11}	2.9 10^{-11}
^{198}Pb	0.2	4.8 10^{-10}	1.7 10^{-10}	1.0 10^{-10}
^{199}Pb	0.2	2.6 10^{-10}	9.4 10^{-11}	5.4 10^{-11}
^{200}Pb	0.2	2.0 10^{-9}	7.0 10^{-10}	4.0 10^{-10}
^{201}Pb	0.2	7.8 10^{-10}	2.7 10^{-10}	1.6 10^{-10}
^{202}Pb	0.2	1.6 10^{-8}	1.9 10^{-8}	8.8 10^{-9}
^{202m}Pb	0.2	6.1 10^{-10}	2.3 10^{-10}	1.3 10^{-10}
^{203}Pb	0.2	1.3 10^{-9}	4.3 10^{-10}	2.4 10^{-10}
^{203}Po	0.5	2.4 10^{-10}	8.5 10^{-11}	4.6 10^{-11}
^{205}Pb	0.2	9.9 10^{-10}	6.1 10^{-10}	2.8 10^{-10}

TABLE 6.1 Committed effective dose per unit intake by ingestion (Sv per Bq) of the "common" radionuclides within PC-CREAM 08 (from ICRP, 1996)

Radionuclide[a]	Fractional absorption, F_1	1 year old	10 year old	Adult
^{205}Po	0.5	$2.8\ 10^{-10}$	$1.1\ 10^{-10}$	$5.8\ 10^{-11}$
^{207}Po	0.5	$5.7\ 10^{-10}$	$2.1\ 10^{-10}$	$1.1\ 10^{-10}$
^{209}Pb	0.2	$3.8\ 10^{-10}$	$1.1\ 10^{-10}$	$5.7\ 10^{-11}$
^{210}Pb	0.2	$3.6\ 10^{-6}$	$1.9\ 10^{-6}$	$6.9\ 10^{-7}$
^{210}Po	0.5	$8.8\ 10^{-6}$	$2.6\ 10^{-6}$	$1.2\ 10^{-6}$
^{211}Pb	0.2	$1.4\ 10^{-9}$	$4.1\ 10^{-10}$	$1.8\ 10^{-10}$
^{212}Pb	0.2	$6.3\ 10^{-8}$	$2.0\ 10^{-8}$	$6.0\ 10^{-9}$
^{214}Pb	0.2	$1.0\ 10^{-9}$	$3.1\ 10^{-10}$	$1.4\ 10^{-10}$
^{238}Pu	0.0005	$4.0\ 10^{-7}$	$2.4\ 10^{-7}$	$2.3\ 10^{-7}$
^{239}Pu	0.0005	$4.2\ 10^{-7}$	$2.7\ 10^{-7}$	$2.5\ 10^{-7}$
^{240}Pu	0.0005	$4.2\ 10^{-7}$	$2.7\ 10^{-7}$	$2.5\ 10^{-7}$
^{241}Pu	0.0005	$5.7\ 10^{-9}$	$5.1\ 10^{-9}$	$4.8\ 10^{-9}$
^{241}Am	0.0005	$3.7\ 10^{-7}$	$2.2\ 10^{-7}$	$2.0\ 10^{-7}$
^{242}Cm	0.0005	$7.6\ 10^{-8}$	$2.4\ 10^{-8}$	$1.2\ 10^{-8}$
^{243}Cm	0.0005	$3.3\ 10^{-7}$	$1.6\ 10^{-7}$	$1.5\ 10^{-7}$
^{244}Cm	0.0005	$2.9\ 10^{-7}$	$1.4\ 10^{-7}$	$1.2\ 10^{-7}$

a Noble gases are omitted from this table as the dose coefficients are zero for ingestion. Also omitted are radionuclides where ICRP have not recommended a dose coefficient; for these radionuclides the dose coefficient is assumed to be zero in PC-CREAM 08.

TABLE 6.2 Committed effective dose per unit intake by inhalation (Sv per Bq) of the "common" radionuclides within PC-CREAM 08

Radionuclide[a]	Lung absorption type[b]	1 year old	10 year old	adult
^{3}H	V	$7.2\ 10^{-11}$	$3.5\ 10^{-11}$	$2.7\ 10^{-11}$
^{14}C	M	$6.6\ 10^{-9}$	$2.8\ 10^{-9}$	$2.0\ 10^{-9}$
^{32}P	F	$7.5\ 10^{-9}$	$1.8\ 10^{-9}$	$7.7\ 10^{-10}$
^{33}P	F	$7.8\ 10^{-10}$	$2.0\ 10^{-10}$	$9.2\ 10^{-11}$
^{35}S	M	$4.5\ 10^{-9}$	$2.0\ 10^{-9}$	$1.4\ 10^{-9}$
^{51}Cr	S	$2.1\ 10^{-10}$	$6.6\ 10^{-11}$	$3.7\ 10^{-11}$
^{54}Mn	M	$6.2\ 10^{-9}$	$2.4\ 10^{-9}$	$1.5\ 10^{-9}$
^{58}Co	M	$6.5\ 10^{-9}$	$2.4\ 10^{-9}$	$1.6\ 10^{-9}$
^{59}Fe	M	$1.3\ 10^{-8}$	$5.5\ 10^{-9}$	$3.7\ 10^{-9}$
^{60}Co	M	$3.4\ 10^{-8}$	$1.5\ 10^{-8}$	$1.0\ 10^{-8}$
^{65}Zn	M	$6.5\ 10^{-9}$	$2.4\ 10^{-9}$	$1.6\ 10^{-9}$
^{89}Sr	M	$2.4\ 10^{-8}$	$9.1\ 10^{-9}$	$6.1\ 10^{-9}$
^{90}Sr	M	$1.1\ 10^{-7}$	$5.1\ 10^{-8}$	$3.6\ 10^{-8}$
^{95}Nb	M	$5.2\ 10^{-9}$	$2.2\ 10^{-9}$	$1.5\ 10^{-9}$
^{95}Zr	M	$1.6\ 10^{-8}$	$6.8\ 10^{-9}$	$4.8\ 10^{-9}$
^{106}Ru	M	$1.1\ 10^{-7}$	$4.1\ 10^{-8}$	$2.8\ 10^{-8}$
^{122}Sb	M	$5.7\ 10^{-9}$	$1.8\ 10^{-9}$	$1.0\ 10^{-9}$
^{124}Sb	M	$2.4\ 10^{-8}$	$9.6\ 10^{-9}$	$6.4\ 10^{-9}$
^{125}Sb	M	$1.6\ 10^{-8}$	$6.8\ 10^{-9}$	$4.8\ 10^{-9}$
^{129}I	F	$8.6\ 10^{-8}$	$6.7\ 10^{-8}$	$3.6\ 10^{-8}$
^{131}I	F	$7.2\ 10^{-8}$	$1.9\ 10^{-8}$	$7.4\ 10^{-9}$
^{132}I	F	$9.6\ 10^{-10}$	$2.2\ 10^{-10}$	$9.4\ 10^{-11}$
^{133}I	F	$1.8\ 10^{-8}$	$3.8\ 10^{-9}$	$1.5\ 10^{-9}$
^{134}Cs	F	$7.3\ 10^{-9}$	$5.3\ 10^{-9}$	$6.6\ 10^{-9}$
^{135}I	F	$3.7\ 10^{-9}$	$7.9\ 10^{-10}$	$3.2\ 10^{-10}$
^{137}Cs	F	$5.4\ 10^{-9}$	$3.7\ 10^{-9}$	$4.6\ 10^{-9}$
^{140}Ba	M	$2.0\ 10^{-8}$	$7.6\ 10^{-9}$	$5.1\ 10^{-9}$
^{140}La	M	$6.3\ 10^{-9}$	$2.0\ 10^{-9}$	$1.1\ 10^{-9}$
^{141}Ce	M	$1.1\ 10^{-8}$	$4.6\ 10^{-9}$	$3.2\ 10^{-9}$
^{144}Ce	M	$1.6\ 10^{-7}$	$5.5\ 10^{-8}$	$3.6\ 10^{-8}$
^{195m}Pb	M	$1.5\ 10^{-10}$	$4.6\ 10^{-11}$	$2.5\ 10^{-11}$
^{198}Pb	M	$4.0\ 10^{-10}$	$1.3\ 10^{-10}$	$6.6\ 10^{-11}$
^{199}Pb	M	$2.2\ 10^{-10}$	$7.1\ 10^{-11}$	$3.6\ 10^{-11}$
^{200}Pb	M	$1.7\ 10^{-9}$	$5.7\ 10^{-10}$	$3.3\ 10^{-10}$
^{201}Pb	M	$6.4\ 10^{-10}$	$2.1\ 10^{-10}$	$1.1\ 10^{-10}$
^{202}Pb	M	$8.9\ 10^{-9}$	$6.7\ 10^{-9}$	$6.3\ 10^{-9}$
^{202m}Pb	M	$5.6\ 10^{-10}$	$1.9\ 10^{-10}$	$9.5\ 10^{-11}$
^{203}Pb	M	$1.0\ 10^{-9}$	$3.6\ 10^{-10}$	$2.0\ 10^{-10}$
^{203}Po	M	$2.1\ 10^{-10}$	$6.7\ 10^{-11}$	$3.5\ 10^{-11}$
^{205}Pb	M	$7.7\ 10^{-10}$	$3.2\ 10^{-10}$	$2.5\ 10^{-10}$
^{205}Po	M	$3.1\ 10^{-10}$	$1.1\ 10^{-10}$	$6.5\ 10^{-11}$
^{207}Po	M	$5.1\ 10^{-10}$	$1.6\ 10^{-10}$	$7.8\ 10^{-11}$

TABLE 6.2 Committed effective dose per unit intake by inhalation (Sv per Bq) of the "common" radionuclides within PC-CREAM 08

Radionuclide[a]	Lung absorption type[b]	1 year old	10 year old	adult
^{209}Pb	M	2.7 10^{-10}	9.2 10^{-11}	5.6 10^{-11}
^{210}Pb	M	3.7 10^{-6}	1.5 10^{-6}	1.1 10^{-6}
^{210}Po	M	1.1 10^{-5}	4.6 10^{-6}	3.3 10^{-6}
^{211}Pb	M	4.5 10^{-8}	1.9 10^{-8}	1.1 10^{-8}
^{212}Pb	M	4.6 10^{-7}	2.2 10^{-7}	1.7 10^{-7}
^{214}Pb	M	4.6 10^{-8}	1.9 10^{-8}	1.4 10^{-8}
^{238}Pu	M	7.4 10^{-5}	4.4 10^{-5}	4.6 10^{-5}
^{239}Pu	M	7.7 10^{-5}	4.8 10^{-5}	5.0 10^{-5}
^{240}Pu	M	7.7 10^{-5}	4.8 10^{-5}	5.0 10^{-5}
^{241}Am	M	6.9 10^{-5}	4.0 10^{-5}	4.2 10^{-5}
^{241}Pu	M	9.7 10^{-7}	8.3 10^{-7}	9.0 10^{-7}
^{242}Cm	M	1.8 10^{-5}	7.3 10^{-6}	5.2 10^{-6}
^{243}Cm	M	6.1 10^{-5}	3.1 10^{-5}	3.1 10^{-5}
^{244}Cm	M	5.7 10^{-5}	2.7 10^{-5}	2.7 10^{-5}

a Noble gases are omitted from this table because, with the exception of some radon isotopes (see Section 6.1.2), the dose coefficients are zero for inhalation. Also omitted are radionuclides where ICRP have not recommended a dose coefficient; for these radionuclides the dose coefficient is assumed to be zero in PC-CREAM 08.

b The particulate lung absorption rates are: S = slow, M = moderate, F = fast (see ICRP, 1996). Radionuclides with type “V” are in the vapour form only. The absorption rates listed are the defaults recommended by ICRP for an unspecified compound.

7 APPLICATION OF THE METHODOLOGY TO ESTIMATE COLLECTIVE DOSES AND DOSES TO INDIVIDUALS

7.1 Introduction

The models described in Chapters 3, 4 and 5 of this report can be used to predict the transport of radionuclides in the environment following their release to the atmospheric or aquatic environments. This chapter describes how this predictive ability can be used to estimate doses to man. The methodology, CREAM, can be applied to the estimation of both collective doses to populations and doses to individuals. In the assessment of collective doses the average habits of the exposed population are generally sufficient to obtain reliable estimates of the doses. However, in the assessment of doses to individuals additional detailed information is often required on the habits of the individuals concerned. Such detailed information will be site-specific and cannot be addressed within a generic methodology of this nature. Some default information is therefore presented on the habits of the population of the EU which will enable the estimation of doses to individuals. Such estimated doses to individuals should not replace calculations based on detailed local habit surveys designed for demonstrating compliance with national regulations.

7.2 Estimation of doses from atmospheric discharges

7.2.1 Collective doses

The definition of collective effective dose is given in section 2.3. The estimation of this quantity for an exposed population requires the integration over time and space of the individual-related effective dose in that population. In its most general form this integration can be written as

$$\int_{d}\int_{\theta}\int_{t} \mathrm{N}(\mathrm{d},\theta,\mathrm{t})\,\dot{\mathrm{H}}(\mathrm{d},\theta,\mathrm{t})\,\mathrm{dd}\,\mathrm{d}\theta\,\mathrm{dt} \qquad (7.1)$$

where $\mathrm{N}(\mathrm{d},\theta,\mathrm{t}) =$ population at time t, at distance d and angle θ relative to the point of discharge,

$\dot{\mathrm{H}}(\mathrm{d},\theta,\mathrm{t}) =$ individual effective dose rate at d, θ and t.

Several pathways need to be considered in assessing the doses received by individuals at any particular location following a discharge to atmosphere; these comprise:

a external irradiation (β, γ) from the cloud;
b inhalation of radionuclides in the cloud;
c external irradiation (β, γ) from deposited activity;
d inhalation of resuspended activity;
e ingestion of contaminated foodstuffs.

A number of approximations are made in order to evaluate the integrals in equation (7.1). The spatial distributions of the population and of the radionuclides in the environment will be continuously varying functions of distance d and angle θ relative to the discharge point. These variations have been approximated in PC-CREAM 08 in the manner indicated in Figure 7.1. The area around each discharge point is divided into a number of annular segments within which the population and agricultural production distributions are assumed to be uniform, as are the distributions of individual dose and radionuclide concentrations in the environment. The collective dose is then calculated for each annular segment and the total collective dose obtained by summation over the segments. In addition, it is assumed that the spatial distribution and size of the population remain constant, as do the spatial distribution and size of agricultural production (see section 2.4). The selection of the annular segments is a compromise between minimising computational effort, the availability of site-specific meteorological data, and ensuring that the errors introduced by such an approximation are small in comparison with other uncertainties in the assessment. Typical values for the radii of the annuli are given in Table 7.1. The number of sectors is determined by the available meteorological data and in PC-CREAM 08 the number of sectors is 8, 12 or 18, depending on the situation. Radionuclide concentrations in the environment are taken to be those calculated for the mid-point of each segment. Hence, equation 7.1 is reduced to

$$\sum_{n} N(d_n, \theta_n) \int_{t} \dot{H}(d_n, \theta_n, t)\, dt \tag{7.2}$$

where the subscript n refers to annular segment n and the summation is over all annular segments. The evaluation of the individual effective dose rate for external exposure pathways is carried out as described in Chapter 3. The evaluation of the individual doses for the inhalation pathways is performed by

$$\dot{H}(d_n, \theta_n, t) = \chi(d_n, \theta_n, t)\, H_{inh}\, I_{inh} \tag{7.3}$$

where $\chi(d_n,\theta_n,t)$ = radionuclide concentration in air in annular segment n at time t (Bq m^{-3}),

H_{inh} = effective dose per unit intake by inhalation (Sv Bq^{-1}),

I_{inh} = inhalation rate (m^3 y^{-1}).

The calculation of the collective doses for each annular segment is relatively straightforward for those pathways which are directly dependent on the location of the population, that is, pathways a to d. For pathway e, ingestion of contaminated foodstuffs, a slightly different approach is used. This arises because foodstuffs are, in general, widely distributed from production point to final consumer. Hence radionuclide concentrations in foodstuffs close to a particular population group will not, in general, determine the intake of activity by that population. The approach adopted is therefore to evaluate the total radionuclide content of a particular foodstuff by combining the calculated distribution of activity concentration with the spatial distribution of production of that foodstuff within the region under consideration. It is assumed that the total radionuclide content of the foodstuff represents, after allowance for losses, the radionuclide intake by the population of that region from that foodstuff. Hence, imports into and exports from the region are assumed to be zero. This approach can be used to determine the intake by foodstuff ingestion by populations in various regions, although the assumption of zero imports into and exports from the region will generally become less valid as the size of the region decreases. Some information is available on imports and exports of food together with the distribution of food within parts of the EU (Sinnaeve and Gerber, 1991; Haywood, Smith and Walmsley, 1991; FAOSTAT, 2003).

Hence, the collective dose from ingestion of a particular radionuclide in a particular foodstuff is obtained as

$$H_{ing} \sum_{n} P(d_n, \theta_n) \int_{t} C(d_n, \theta_n, t)\, dt \qquad (7.4)$$

where H_{ing} = organ dose equivalent or effective dose per unit intake by ingestion (Sv Bq^{-1}),

$P\ (d_n, \theta_n)$ = yield of foodstuff in annular segment n (kg),

$C\ (d_n, \theta_n, t)$ = radionuclide concentration in the foodstuff in annular segment n at time t (Bq kg^{-1}), after allowing for the delay between production and consumption.

The total collective dose from ingestion of foodstuffs is thus obtained by summation of equation 7.4 over all foodstuffs and radionuclides.

As described in section 2.4, in the estimation of collective doses it is sufficient to assume that the population is entirely adult, so that the inhalation rate and dose coefficients for internal irradiation are taken to be those for adults. Values of dose coefficients, integrated to age 70 following intake at age 20, are given in Chapter 6 and the inhalation rate is discussed below.

For the radionuclides tritium, carbon-14, krypton-85 and iodine-129 there will also be contributions to the collective dose arising from their global circulation, additional to the 'first-pass' collective doses estimated by the methods described above. The calculation of these additional contributions is described in Chapter 5.

A recent study for the EC considered the assessment of collective doses in more detail, including the possibility of breaking down the collective dose received by different population groups and as a function of time (Smith et al, 2006).

7.2.2 Doses to individuals

The methodology can equally be applied to the estimation of doses to individuals as well as collective doses, although reliable estimates of the former require fairly detailed habit data. The models described in Chapter 3 can be used to calculate the concentration in air and the deposition rate at a particular distance from the discharge in a particular sector. The same pathways apply in calculating doses to individuals as in the calculation of collective doses (see section 7.2.1). The dose rates arising from inhalation of the plume and from external irradiation from the plume are directly proportional to the instantaneous radionuclide distribution in the air. The dose rates arising from deposited activity (inhalation of resuspended activity, external irradiation from deposited activity and ingestion of contaminated foodstuffs) are dependent on the instantaneous concentrations in the relevant media, which are a function of both current and previous deposition. Hence, given a constant discharge rate, dose rates from deposited activity may increase with time, depending upon the radioactive half-life of the radionuclide and its environmental behaviour.

The dose rate arising from inhalation of the plume is obtained using equation 7.3. It is reasonable to assume that, for continuous discharges, the concentration in air inside a building will be the same as that outside the building so that no allowance for occupation of buildings is necessary for this exposure pathway. Different values of the dose coefficient should be used for individuals of different ages (see Chapter 6). Doses arising from inhalation of resuspended activity are also obtained using equation 7.3, substituting the radionuclide concentration in air due to resuspension for the concentration in the plume. An estimate of the maximum likely dose from resuspended activity would assume that the individual was always in the area where resuspension took place. A more realistic estimate of the dose would take into account time spent in other areas (see discussion of external irradiation below). The time at a location would potentially affect all non-food ingestion pathways but indoor location factors for inhalation, beta irradiation from the cloud and deposited material are currently set as 1, 1, and 0, respectively, in PC-CREAM 08.

The dose rate arising from external irradiation from the plume is obtained using the models described in Chapter 3. To obtain an estimate of the maximum likely dose from this pathway it would be appropriate to consider the individual to be outdoors in a rural environment all the time. A more realistic dose estimate would take into account the amount of time spent indoors and the shielding afforded by buildings. In the UK approximately 90% of an individual's time is spent indoors (Brown, 1983) and this is used as the default value for all age groups within PC-CREAM 08. Similar values to the UK are likely in northern EC countries but it is likely that people will spend longer periods out of doors in the warmer south. There is also some evidence that different age groups could spend different fractions of the time inside and out, with children potentially spending less time inside even in northerly countries such as the UK (BBC, 1984). A value for indoor occupancy of 80% is therefore probably more appropriate for the EC as a whole and for certain age groups although site specific surveys would be needed to determine if this is the case for any specific assessment. The shielding afforded by buildings is discussed in Chapter 3. Similar considerations apply to the calculation of external irradiation doses arising from deposited activity.

For ingestion of contaminated foodstuffs it is not possible to estimate doses to individuals unless some assumption is made regarding the source of the individual's dietary intake. As an extreme hypothesis, the individual is assumed to derive his total diet from his local habitat. A less extreme estimate of the dose to a member of the critical group could be made assuming that only a fraction of the diet is obtained locally, the fraction being different for different foodstuffs. Another alternative would be to assume that an individual's food intake was obtained entirely from the area within a certain distance of the discharge point, for example, 5 km. The assumptions to be adopted in any particular case will be site-specific. The dose rate arising from a particular radionuclide in a particular foodstuff is obtained as

$$C_f(t)\, I_f\, H_{ing} \tag{7.5}$$

where $C_f\,(t)$ = radionuclide concentration in the edible part of foodstuff f at time t, allowing for delay between production and consumption (Bq kg^{-1}),

I_f = ingestion rate of foodstuff f (kg y^{-1}).

H_{ing} = organ dose equivalent or effective dose per unit intake by ingestion (Sv Bq^{-1})

7.3 Estimation of doses from aquatic discharges

A detailed description of the methods used within PC-CREAM 08 for estimating individual doses following discharges into a river system are given in Section 4.2 and the method used to calculate the individual or collective doses following a marine discharge are given in Section 4.4. In general both river and marine systems are represented by a number of compartments with the radionuclide concentrations, and hence doses, being homogeneous within each compartment. A variety of exposure pathways are possible and these are listed in Table 4.1.

For marine discharges collective doses can be calculated from the radionuclide concentrations in each compartment and the total collective dose obtained by summation. For both marine and river discharges doses to individuals will often be highest close to the discharge point because of dispersion in the water. However, the river model incorporates the movement of bed sediment at a slower rate than the movement of water. Radionuclide concentrations in the bed sediment can be higher some distance from the discharge point than close to the discharge point (see section 4.2.2). Hence, for doses associated with a river discharge, care must be taken in the estimation of individual doses arising from pathways dependent upon radionuclide concentrations in bed sediment. For marine environments, sediment movement between compartments is not considered important and is not modelled.

7.4 Habits of individuals

In order to estimate doses to individuals it is necessary to make some assumptions about their habits, such as consumption rates of different foodstuffs. It is clear that these habits can vary widely from one individual to another and also in the same individual from one time to another. There are also regional differences across the EC. It may also be necessary to consider different sets of habit data so that doses to both average individuals and to critical groups can be assessed.

The assumed inhalation rate is an important parameter in the calculation of doses from inhalation. Representative values for inhalation rate are given in Table 7.2 (ICRP, 1994).

Consumption of water is an important pathway for radionuclides discharged to fresh water. Water intake rates are given in Table 7.3 (Smith and Jones, 2003); these are appropriate for both critical individuals and average individuals.

The intake of foodstuffs has been studied extensively within the EC (see, for example, the information presented within Jones et al, 2006). Tables 7.4 and 7.5 give the default average and critical terrestrial food ingestion rates, respectively, included within PC-CREAM 08. Table 7.6 gives the ingestion rates for aquatic foods for both average and critical consumers. These ingestion rates are based on UK data (Smith and Jones, 2003). These values should be replaced by representative values for a particular country or location where available.

7.5 Age groups assessed

The default age groups that are considered in PC-CREAM 08 are the 1 year old infant, the 10 year old child and the adult (assumed to be aged 20), and in all cases committed doses from radionuclides taken into the body are integrated to age 70 (see Chapter 6). This is done with the use of dose coefficients for ingestion and inhalation from ICRP (1996). Recent publications (for example from ICRP, 2007) have investigated the relative importance of the dose to other age groups. For most radionuclides there were found to be no significant differences in the dose between these other age groups (for example, the 5 year old and the 15 year old) and the default age groups included in PC-CREAM 08. Therefore, it is sufficient to consider the 1 year old, the 10 year old and the adult when assessing the dose to a member of the critical group.

ICRP has also published dose coefficients for the fetus (ICRP, 2001) and for breastfed infants (ICRP, 2004). In most cases the doses to the fetus or breastfed infant following intake of radionuclides by the mother will be less than the dose to the mother (HPA, 2008). However, doses to the unborn infant via an intake by the mother are expected to be significantly higher, by up to an order of magnitude, than those to one of the default age groups for ^{32}P, ^{33}P, ^{45}Ca and ^{89}Sr (HPA, 2008).

Although PC-CREAM 08 does not explicitly estimate the dose to the fetus or the breast fed infant, the dose to these age groups can be estimated, where necessary, from the estimated adult doses. The suggested method is to use PC-CREAM 08 to estimate the dose to an adult for each pathway that involves an intake (inhalation from the plume, ingestion of food etc) and then multiply this dose by the ratio of the appropriate fetal (ICRP, 2001) to adult (ICRP, 1996) dose coefficients to get the dose to the fetus/breastfed infant. The overall estimated dose to the fetus/breastfed infant is then the sum of the doses from all the applicable pathways, noting that the dose from external irradiation can be assumed to be equal for both the adult and the fetus/breastfed infant. For further information regarding the estimation of the dose to the fetus/breastfed infant reference should be made to HPA (2008).

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7.7 Tables

TABLE 7.1 Radii typical of those used in PC-CREAM 08 to specify annular distance bands in the estimation of collective doses to UK from discharges to the atmosphere

Radial limits of annulus, km
0 – 1
1 - 2
2 - 3
3 - 5
5 - 7
7 - 10
10 - 15
15 - 20
20 - 30
30 - 40
40 - 50
50 - 65
65 - 80
80 - 100
100 - 200
200 - 300
300 - 500
500 - 750
750 - 1000
1000 - 1500

TABLE 7.2 Inhalation rates[a,b]

Age	Inhalation rate, m^3 y^{-1}
Infant	1900
Child	5600
Adult male	8100

a These inhalation rates are considered appropriate for average individuals and critical groups.

b Taken from ICRP (1994 and 1975).

TABLE 7.3 Water intake rates[a,b]

Age	Water intake rate, $m^3\ y^{-1}$
Infant	0.26
Child	0.35
Adult	0.60

a These values exclude intakes of water by food, including milk, by oxidation of food, by inhalation or by absorption through the skin.

b Taken from Smith and Jones (2003).

Table 7.4 Default average food ingestion rates[a] used in PC-CREAM 08 for terrestrial foods, kg y^{-1}

Food type	Adult	10 year old	1 year old
Cow liver	2.75	1.5	0.5
Cow meat	15	15	3
Cow milk	95	110	130
Cow milk products	20	15	15
Fruit	20	15	9
Grain	50	45	15
Green vegetables	35	15	5
Root vegetables	60	50	15
Sheep liver	2.75	1.5	0.5
Sheep meat	8	4	0.8

a These rates were obtained from Smith and Jones (2003) and are for the UK. They maybe suitable for other countries although country specific information should be used if available.

Table 7.5 Default critical food ingestion rates[a] used in PC-CREAM 08 for terrestrial foods, kg y^{-1}

Food type	Adult	10 year old	1 year old
Cow liver	10	5	2.75
Cow meat	45	30	10
Cow milk	240	240	320
Cow milk products	60	45	45
Fruit	75	50	35
Grain	100	75	30
Green vegetables	80	35	15
Root vegetables	130	95	45
Sheep liver	10	5	2.75
Sheep meat	25	10	3

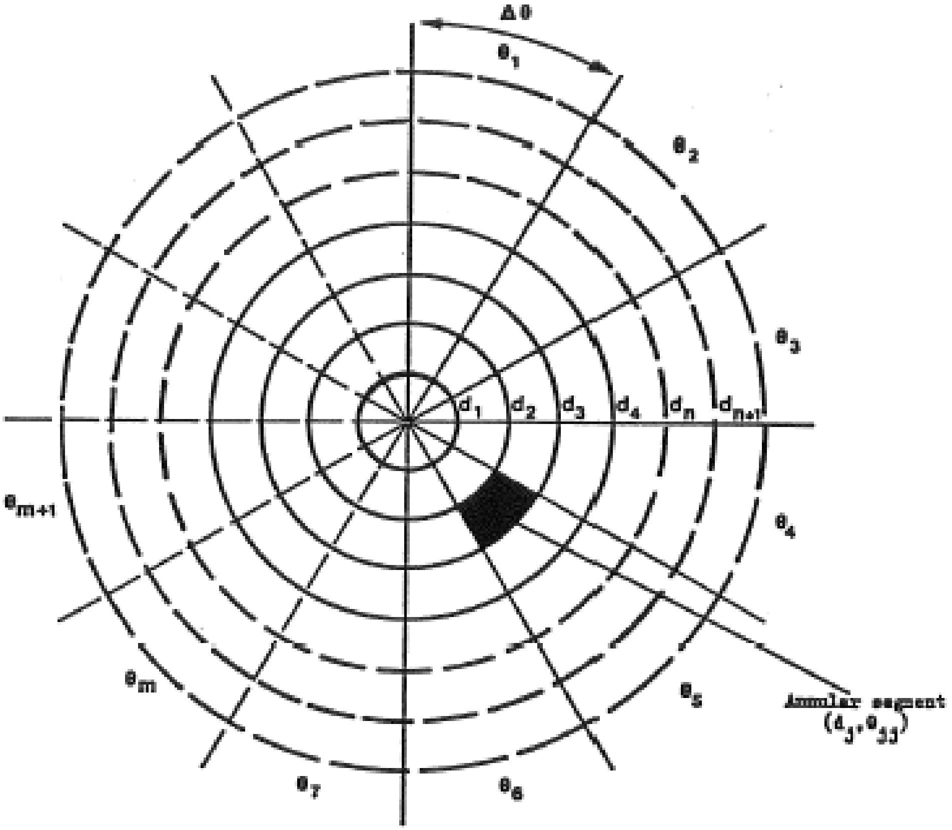
a These rates were obtained from Smith and Jones (2003) and are for the UK. They maybe suitable for other countries although country specific information should be used if available.

TABLE 7.6 Representative average and critical group aquatic food intake rates[a]

	Average intake rate kg y^{-1}			Critical intake rate kg y^{-1}		
	Adult	Child	Infant	Adult	Child	Infant
Marine fish	15	6	3.5	100	20	5
Freshwater fish	1	0.7	0.3	20	5	1
Crustacea	1.75	1.25	-	20	5	-
Mollusca	1.75	1.25	-	20	5	-
Seaweed	-	-	-	-	-	-

a These data are applicable to the UK and are discussed in more detail in Smith and Jones (2003).

7.8 Figure



$\Delta\theta = \frac{360^\circ}{N}$ where N is the number of sectors

FIGURE 7.1 Illustration of the scheme of annular segments adopted to represent the spatial distribution of population and activity in various parts of the environment

8 THE ESTIMATION OF HEALTH EFFECTS IN AN EXPOSED POPULATION

Exposure of a population to radiation may lead to the incidence of deterministic and stochastic effects in the population itself, together with hereditary stochastic effects in its descendants. Deterministic effects occur only if particular threshold levels of dose are exceeded. These levels are far in excess of doses typically encountered due to normal releases of effluents from the nuclear fuel cycle or other controlled sources and deterministic effects can, therefore, be disregarded in the present context. The important stochastic effects are cancer in the exposed population and hereditary effects in its descendants. The probability of occurrence of stochastic effects is assumed proportional to dose, without threshold.

The appearance of stochastic effects, and thus the expression of the risk following irradiation is not immediate but extends over a considerable period which maybe as large as several tens of years. Age at exposure has therefore a considerable influence on the probability that a particular dose will induce a given health effect; the probability will decrease when life expectancy at the time of irradiation becomes comparable with or smaller than the median time taken for the appearance of the effect. A rigorous evaluation of the incidence of stochastic effects in an exposed population must, therefore, take account of the age distribution of the population since for each age at exposure or intake, the temporal distribution of dose will affect the probability of appearance of the effect. The latter is particularly important for internally incorporated radionuclides with long effective half-lives in the body when exposure continues over an extended period. It is also important to recognise that the nominal risk coefficients used by ICRP (2007) apply to an averaged "world population" rather than any specific population group. In addition, the tissue weighting factors used in the calculation of committed effective dose are rounded, sex- and age-averaged, values that are suitable for protection purposes but have the potential to give misleading results when considering risks to specific population groups from internally incorporated radionuclides.

When this methodology was originally developed it was designed to estimate total health detriment as defined in ICRP publication 26 (ICRP, 1977) and a conservative approach was adopted where risk coefficients were applied to the estimated collective doses to predict possible health effects (NRPB/CEA, 1979). For the revision of the methodology in 1995 (Simmonds, Lawson and Mayall, 1995) the definition of health detriment was updated to that given in ICRP publication 60 (ICRP, 1991). The software based on this methodology, PC-CREAM (Mayall et al, 1997) also included an evaluation of health effects based on the ICRP Publication 60 risk coefficients as outlined in Simmonds, Lawson and Mayall (1995). In its latest recommendations ICRP have again reviewed the definition of health detriment and the related risk coefficients (ICRP, 2007). However, ICRP no longer recommend using collective effective dose to estimate health effects but state that: ‘*Collective effective dose is not intended as a tool for epidemiological studies and it is inappropriate to use it in risk projections. This is because the assumptions implicit in the calculation of collective effective dose conceal large biological and statistical uncertainties. Specifically, the computation of cancer* *deaths based on collective effective doses involving trivial exposures to large populations is not reasonable and should be avoided'* (ICRP, 2007). Following this advice the estimation of health effects is not directly included in PC-CREAM 08.

ICRP also advocates that collective effective dose should be disaggregated by determining the collective dose delivered in different time periods and to different populations. Using PC-CREAM 08 it is possible to obtain collective effective doses integrated over different time periods and to some extent to different populations (eg to the UK, Europe and the World). Ideally the levels of individual dose associated with the collective dose would also be estimated. However, as noted in Chapter 7 the collective doses from ingestion is based on the production of terrestrial foods or the catches of seafood and information on the associated individual dose is not readily available. A recent EC study considered methods for estimating collective doses (Smith et al, 2006) and this report considers these issues in more detail.

Although PC-CREAM 08 does not include the estimation of health effects it is possible to use the results obtained to estimate health effects using appropriate risk coefficients. Table 8.1 summarises the overall risk coefficients for the general population recommended by ICRP in Publication 103 for ease of reference. This gives the risk factors for hereditary disease and radiation-induced fatal cancer, together with the total weighted detriment allowing for radiation-induced non-fatal cancers as well as years of life lost for both cancers and hereditary disease.

Because of the considerable uncertainties, estimates of health effects from collective doses received from long-lived radionuclides over many hundreds of years into the future should be viewed with extreme caution.

8.1 References

ICRP (1991). 1990 Recommendations of the International Commission on Radiological Protection. ICRP Publication 60. *Ann ICRP*, **21** (1–3).

ICRP (1977) Recommendations of the International Commission on Radiological Protection. ICRP Publication 26. *Ann ICRP*, **1** (3).

ICRP (2007). The 2007 Recommendations of the International Commission on Radiological Protection. ICRP Publication 103. *Ann ICRP*, **37** (2-4).

NRPB/CEA (1982). Methodology for evaluating the radiological consequences of radioactive effluents released in normal operations. CEC Doc No V/3865/1/79, 2nd impression.

Smith KR, Bexon AP, Sihra K, Simmonds JR, Lochard J, Schneider T and Bataille (2006). Guidance on the calculation, presentation and use of collective doses for routine discharges. EC, Radiation Protection 144, Luxembourg. Available from: http://ec.europa.eu/energy/nuclear/radiation_protection/publications_en.htm

Mayall A, Cabianca T, Attwood C, Fayers C, Smith JG, Penfold JSS, Steadman D, Martin G, Morris TP and Simmonds JR (1997). PC-CREAM. Installing and using the PC system for assessing the radiological impact of routine releases. EUR 11791 EN, Chilton, NRPB-SR296. Code updated in 1998.

8.2 Table

TABLE 8.1 Nominal risk coefficients for stochastic effects for exposure to radiation at low dose rate for the whole population (ICRP, 2007)

	Risk (10^{-2} Sv^{-1})
Cancer	5.5
Heritable effects	0.2
Total	5.7

Notes

(1) Taken from ICRP publication 103 (ICRP, 2007).

9 CONCLUSIONS

This report outlines the methodology, CREAM (Consequences of Releases to the Environment: Assessment Methodology), for the assessment of the radiological consequences of routine releases of radioactive effluents as applied in the computer system, PC-CREAM 08. The methodology is based on that originally published by the European Commission in 1995 but there have been some changes made, notably to the models for the transfer of radionuclides in the marine environment. The methodology is intended for application in the European Community. However, as a generalised approach has been adopted the models and methods provided are appropriate for wider use. Using CREAM and PC-CREAM 08, both individual and collective radiation doses can be determined.

Individual exposures can be compared with the appropriate dose limits or constraints as required by regulatory procedures. Collective doses can be used to give an indication of possible health detriment. Both individual and collective doses can also form an essential input into the optimisation procedure for effluent treatment systems.

Mathematical models have been developed which represent the transfer of a wide range of radionuclides through atmospheric, terrestrial and aquatic pathways. These models enable the spatial and temporal distribution of radioactivity in the environment to be predicted and can be used to estimate individual and collective doses.

Default values have been given for many parameters and have been used to determine illustrative results. The choice of such values is necessarily a compromise taking into account the range of possible values. For specific sites and applications it will often be appropriate to choose alternative values. The models adopted in CREAM are those considered appropriate for routine releases. They have generally been subject to extensive verification and validation. In each case the limitations of the models have been discussed in this report and, where appropriate more detailed models referred to.

Although, the aim is generally to make a "best estimate" of radiation doses it has sometimes been necessary to adopt cautious assumptions. This should be recognised, particularly in applications such as optimisation studies where more realistic estimates may be required.

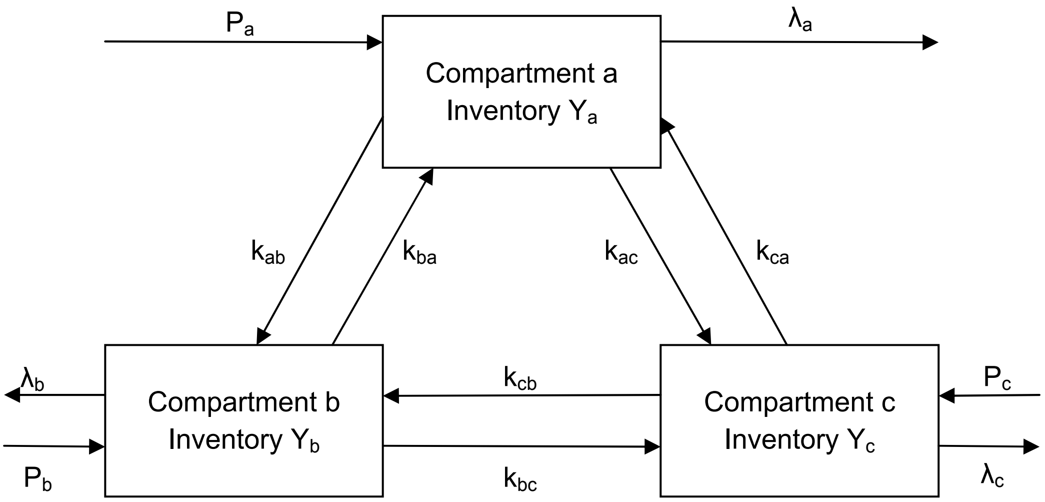
Any radiological assessment has associated uncertainties, due to the models and parameter values adopted. In particular applications it may be necessary to quantify such uncertainties. In addition, sensitivity analyses may be carried out to identify important parameters where uncertainty in their value has a significant effect on the overall result. Such analyses give an indication of areas where research effort should be concentrated to improve the overall accuracy of the assessment.

The methodology CREAM and the associated software PC-CREAM 08 are expected to find wide application in estimating individual and collective doses primarily from routine discharges of radioactive effluents to the environment. Further development of the methodology and the PC-CREAM software is planned for the future and this report will be updated when new versions of PC-CREAM are released.

APPENDIX A General form of compartmental model

A1 COMPARTMENTAL MODEL

General form of compartmental model



A dynamic compartmental model has been used in which the transfer rate of material between compartments is proportional to the inventory of material in the source compartment. The relationship can be generally represented by:

$$\frac{dY_a}{dt} = \sum_{b=1}^{n} k_{ba} Y_b - \left[\sum_{b=1}^{n} k_{ab} \right] Y_a - \lambda_a Y_a + P_a$$

where k_{ab} and k_{ba} are transfer coefficients between two compartments having inventories Y_a and Y_b.

λ_a is an effective transfer coefficient from compartment a which takes account of loss of material from the compartment without transfer to another, for example, radioactive decay.

P_a is a source of continuous input into compartment a. The time integral of the inventory in any compartment is obtained as:

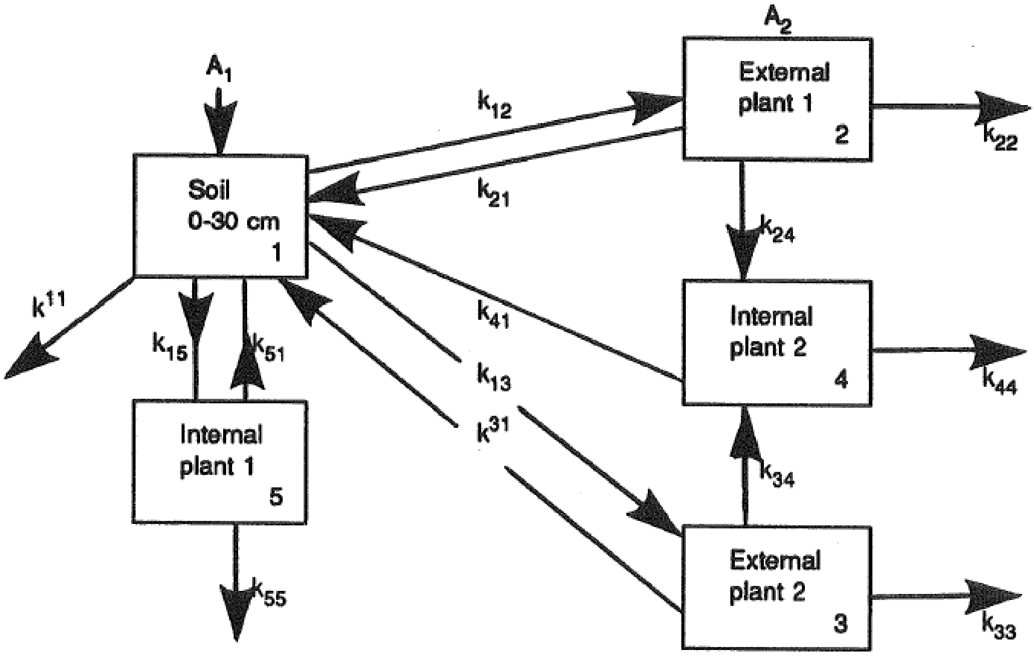
$$Y_a' = \int_0^t Y_a dt$$

The time variation and time integrals of the respective inventories are obtained by solution of the sets of simultaneous equations shown above.

APPENDIX B Transfer coefficients used in the terrestrial foodchain models

The transfer coefficients used in each of the terrestrial foodchain models are summarised in the following sections. The rate constants are calculated in PC-CREAM 08 for each set of input values and the values given here are based on the default inputs. Each model is illustrated schematically. In some cases the models differ from the simplified versions presented in the main text; the modifications are made to facilitate computation.

B1 GREEN VEGETABLE MODEL



Inputs A_1 = (1 - p) x deposition rate

A_2 = p x deposition rate

where p is the interception factor (see Table 3.18).

Notes

1 External plant (1) is for direct deposition and initial resuspension.
External plant (2) is for soil contamination.

2 Internal plant (1) is for root uptake.

Internal plant (2) is for translocation of the surface deposit.

3 k_{12} represents initial resuspension on to external plant.

k_{21} represents removal due to weathering processes with a 14-day half-life.

4 The translocation process is represented using the transfer coefficients k_{24}, k_{34} and k_{41}.

5 Periodic cropping of the plant throughout the year is represented by the transfer coefficients k_{22}, k_{33}, k_{44} and k_{55}. The value for these transfer coefficients is based on 2 crops per year.

Element independent transfer coefficients

Transfer coefficients	Value, d^{-1}
k_{11}	$1.90\ 10^{-5}$
k_{12}	$2.07\ 10^{-6}$
k_{21}	$4.95\ 10^{-2}$
k_{13}	$3.84\ 10^{-2}$
k_{31}	$8.64\ 10^{4}$
k_{51}	$8.64\ 10^{4}$
k_{22}, k_{33}, k_{44}, k_{55}	$5.48\ 10^{-3}$

Mobility dependent transfer coefficients for translocation

Mobility of element[1]	Value, d^{-1}		
	k_{41}	k_{24}	k_{34}
Mobile	$3.43\ 10^{-2}$	$2.34\ 10^{-3}$	$2.34\ 10^{-3}$
Semi-mobile	$1.35\ 10^{-2}$	$2.46\ 10^{-4}$	$2.46\ 10^{-4}$
Immobile	0.0	0.0	0.0

Note:

1 Elements are grouped by their mobility and transfer coefficients are chosen reflecting this. Details of element mobility are given in Table 3.19.

Element dependent transfer coefficients for root uptake

	Value, d^{-1}
Element	$k_{1\,5}$
Sulphur	$1.15\ 10^{2}$
Chromium	$5.76\ 10^{-2}$
Manganese	$1.92\ 10^{1}$
Iron	$3.84\ 10^{-2}$
Cobalt	$1.92\ 10^{0}$
Zinc	$1.92\ 10^{2}$
Strontium	$3.84\ 10^{1}$
Yttrium	$1.92\ 10^{0}$
Zirconium	$1.92\ 10^{-2}$
Niobium	$1.92\ 10^{0}$
Ruthenium	$1.92\ 10^{0}$
Antimony	$1.92\ 10^{0}$
Tellurium	$5.76\ 10^{-1}$
Iodine	$3.84\ 10^{0}$
Caesium	$1.92\ 10^{0}$
Barium	$1.92\ 10^{0}$
Lanthanum	$5.76\ 10^{-1}$
Cerium	$1.92\ 10^{-1}$
Plutonium	$1.92\ 10^{-3}$
Americium	$9.60\ 10^{-3}$
Curium	$9.60\ 10^{-3}$

B1.1 Running model for routine release applications

Application: continuous deposition per unit area for 1 year

The growth of the green vegetable crop is modelled explicitly for the first year.

A growing period of 120 days is assumed. For the remainder of the first year the land is assumed to be fallow and transfers from the soil to the plant are set to be zero.

At the start of the second year the inventories of all the plant boxes are set to zero and deposition onto plant and soil is stopped.

The processes of initial resuspension and weathering are also stopped at the start of the second year; ie transfer coefficients k_{12} and k_{21} are set to zero.

The model is run continuously in this mode for the times required, with continuous cropping at two crops per year represented by the transfer coefficients k_{33}, k_{44} and k_{55}. It is assumed that compartment 2 also continues to be cropped but the inventory in, and therefore loss from, this compartment is 0 from the start of the second year.

B1.2 Calculations of activity concentrations in green vegetables

For a continuous deposition rate (Bq km^{-2} s^{-1})

Time-integrated concentration in first year = concentration at 120 days (Bq y kg^{-1})

$$= \frac{I_2 \times F_r + I_3 \times F_r + I_4 + I_5}{Y}$$

where I_2, I_3, I_4 and I_5 are the inventories of activity in compartments 2, 3, 4 and 5.

F_r is the fraction of activity retained after processing (see Table 3.18) and Y is the yield of produce in kg km^{-2} (see Table 3.18).

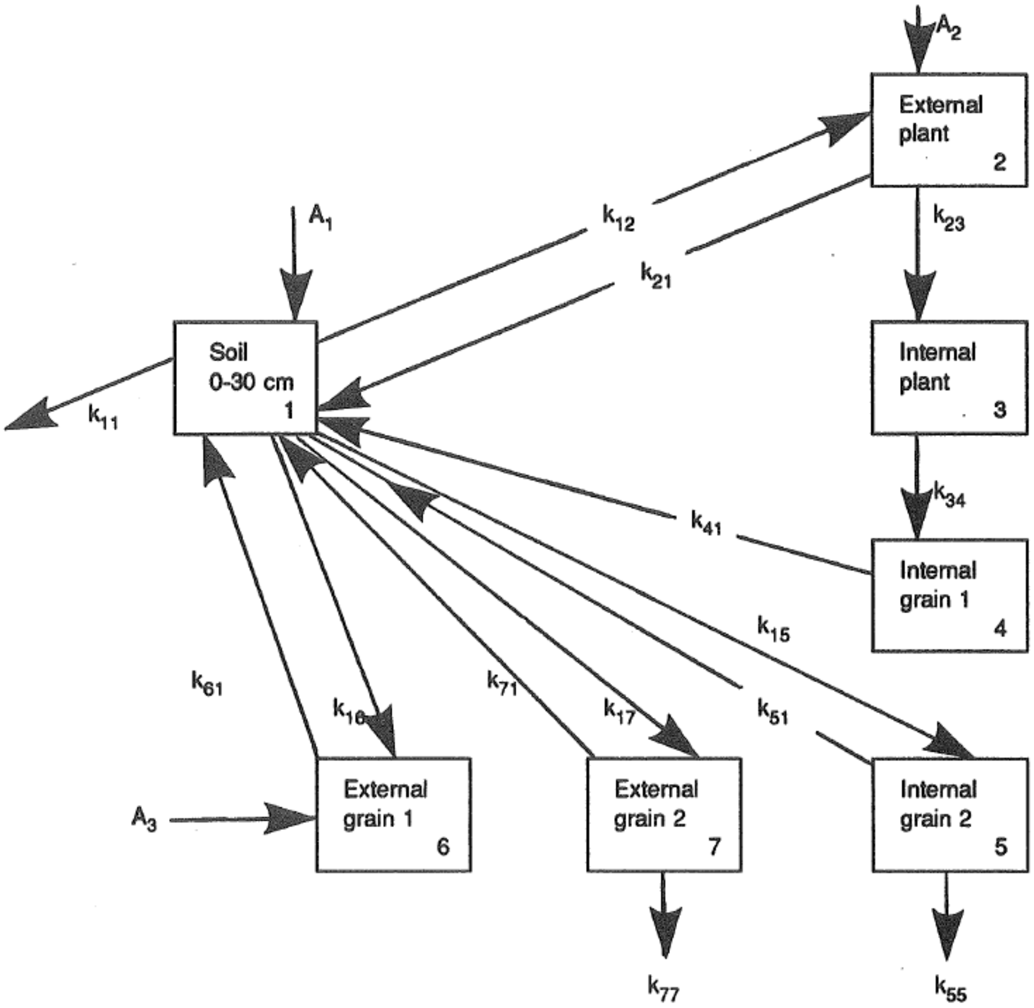
Time-integrated concentration to subsequent times (t) after 1 year = time-integrated

concentration at time t

$$= \frac{I_3 \times F_r + I_4 + I_5}{Y}$$

where I_3, I_4 and I_5 are the integrals of activity in compartments 3, 4 and 5.

B2 GRAIN MODEL



Inputs A_1 = $1 - (P_1 + P_2) \times$ deposition rate

A_2 = $P_1 \times$ deposition rate

A_3 = $P_2 \times$ deposition rate

where P_1 and P_2 are interception factors for the external plant and external grain respectively (see Table 3.18).

Notes:

1 External plant is for initial resuspension and direct deposition on to the whole cereal plant.

2 Internal grain (2) is for root uptake.
External grain (1) is for initial resuspension and direct deposition on to the grain seed.
External grain (2) is for soil contamination of grain.

3 k_{12} represents initial resuspension on to the whole cereal plant in the period immediately after the input.
k_{21} represents removal due to weathering processes from the whole cereal plant.

4 k_{16} represents initial resuspension on to grain seed in the period immediately after the input.
k_{61} represents removal due to weathering processes from grain.

5 The translocation process is represented by transfer coefficients k_{23}, k_{34} and k_{41}.

Element independent transfer coefficients

Transfer coefficient	Value, d^{-1}
k_{11}	$1.90\ 10^{-5}$
k_{12}	$2.07\ 10^{-6}$
k_{21}	$4.95\ 10^{-2}$
k_{51}	$8.64\ 10^{4}$
k_{16}	$8.29\ 10^{-8}$
k_{61}	$4.81\ 10^{-2}$
k_{17}	$6.91\ 10^{-3}$
k_{71}	$8.64\ 10^{4}$
k_{55}, k_{77}	$2.74\ 10^{-3}$

Element dependent transfer coefficients for translocation

Mobility of element[1]	Value, d^{-1}		
	k_{23}	k_{34}	k_{41}
Mobile	$3.41\ 10^{-2}$	$6.44\ 10^{-2}$	$5.21\ 10^{-2}$
Semi-mobile	$3.71\ 10^{-2}$	$6.90\ 10^{-2}$	$4.52\ 10^{-1}$
Immobile	0.0	0.0	0.0

Note:

1 Elements are grouped by their mobility and transfer coefficients are chosen reflecting this. Details of element mobility are given in Table 3.19.

Element dependent transfer coefficients for root uptake

	Value, d^{-1}
Element	k_{15}
Sulphur	$4.61\ 10^{+1}$
Chromium	$2.30\ 10^{-2}$
Manganese	$7.68\ 10^{+0}$
Iron	$3.07\ 10^{-2}$
Cobalt	$3.84\ 10^{-1}$
Zinc	$7.68\ 10^{+1}$
Strontium	$7.68\ 10^{+0}$
Yttrium	$7.68\ 10^{-1}$
Zirconium	$7.68\ 10^{-3}$
Niobium	$7.68\ 10^{-1}$
Ruthenium	$7.68\ 10^{-1}$
Antimony	$7.68\ 10^{-1}$
Tellurium	$2.30\ 10^{-1}$
Iodine	$1.54\ 10^{+0}$
Caesium	$7.68\ 10^{-1}$
Barium	$7.68\ 10^{-1}$
Lanthanum	$2.30\ 10^{-1}$
Cerium	$7.68\ 10^{-2}$
Plutonium	$1.54\ 10^{-3}$
Americium	$3.84\ 10^{-3}$
Curium	$1.54\ 10^{-3}$

B2.1 Running the model for routine release applications

Application: continuous deposition per unit area for 1 year.

The growth of the grain crop is modelled explicitly for the first year.

A growing period of 120 days is assumed. For the remainder of the first year the land is assumed to be fallow and transfers from the soil to the plant are set to zero. In addition, the transfer coefficient k_{21} is set to $8.64\ 10^{4}\ d^{-1}$ at the start of the fallow period to represent the ploughing back of the remaining plant into the soil.

At the start of the second year the inventories of all the plant compartments are set to zero and deposition onto plant, grain and soil is stopped.

The processes of initial resuspension and weathering onto the plant and the grain seed is also stopped, ie, transfer coefficients k_{12}, k_{21}, k_{16} and k_{61} are set to zero.

The model is run continuously in this mode for the remaining times required with continuous cropping at 1 crop/year represented by the transfer coefficients k_{55} and k_{77}.

B2.2 Calculation and activity concentrations in grain

For a continuous deposit rate (Bq km^{-2} s^{-1})

Time-integrated concentration in first year = concentration at 120 days (Bq y kg^{-1})

$$= \frac{I_4 + I_5 + I_6 \times F_r + I_7 \times F_r}{Y}$$

where I_4, I_5, I_6 and I_7 are the inventories of activity in compartments 4, 5, 6 and 7.

F_r is the fraction of activity retained after processing (see Table 3.18) and Y is the yield of produce in kg km^{-2} (see Table 3.18).

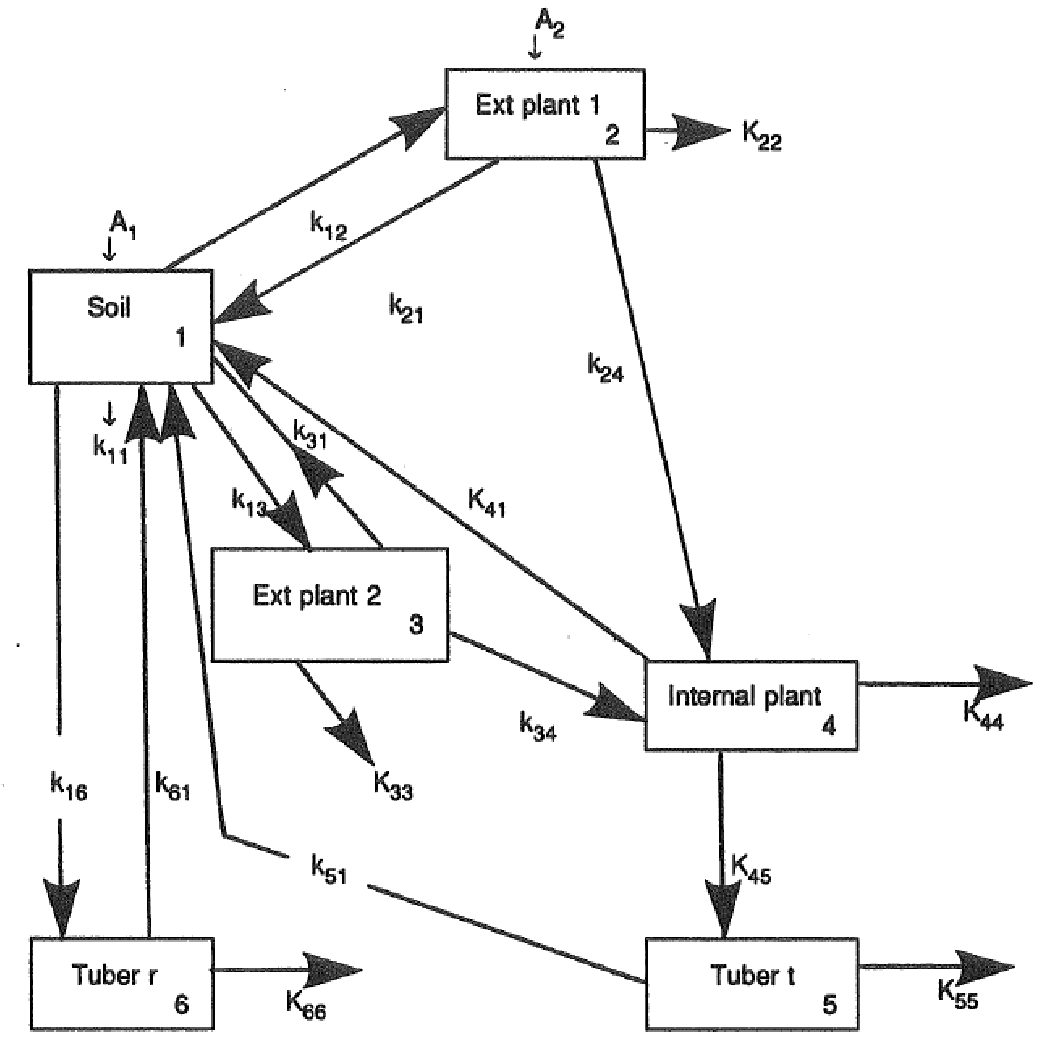
Time-integrated concentration to subsequent times (t) after 1 year =

time-integrated concentration at time t

$$= \frac{I_5 + I_7 \times F_r}{Y}$$

where I_5 and I_7 are the integrals of activity in compartments 5 and 7.

B3 POTATO MODEL



Inputs A_1 = (1 - p) x deposition rate

A_2 = p x deposition rate

where p is the interception factor (see Table 3.18).

Notes

1 External plant 1 is for direct deposition and initial resuspension.
External plant 2 is for soil contamination.

2 Internal plant is for translocation of the surface deposit.

3 Tuber t is for translocation of the surface deposit.
Tuber r is for root uptake.

4 k_{12} represents initial resuspension onto the plant.
k_{21} represents removal due to weathering processes with a 14 day half-life.

5 The translocation process is represented using the transfer coefficients k_{24}, k_{34}, k_{45} and k_{51}.

6 Cropping of the plant throughout the year is represented by the transfer coefficients k_{55} and k_{66} based on one crop per year.

Element independent transfer coefficients

Rate constant	Value, d^{-1}
$k_{1\,1}$	$1.90\ 10^{-5}$
$k_{1\,2}$	$2.76\ 10^{-6}$
$k_{2\,1}$	$4.95\ 10^{-2}$
$k_{1\,3}$	$1.15\ 10^{-1}$
$k_{3\,1}$	$8.64\ 10^{+4}$
$k_{6\,1}$	$8.64\ 10^{+4}$
$k_{5\,5}$ $k_{6\,6}$	$2.74\ 10^{-3}$

Element dependent transfer coefficients for translocation

Mobility of element[1]	Value, d^{-1}			
	k_{24}	k_{34}	k_{45}	k_{51}
Mobile	$4.20\ 10^{-2}$	$4.20\ 10^{-2}$	$5.80\ 10^{-2}$	$4.15\ 10^{-2}$
Semi-mobile	$4.46\ 10^{-5}$	$4.46\ 10^{-5}$	$3.73\ 10^{-3}$	$2.18\ 10^{-4}$
Immobile	0	0	0	0

Note:

1 Elements are grouped by their mobility and transfer coefficients are chosen reflecting this. Details of element mobility are given in Table 3.19.

Element dependent transfer coefficients for root uptake

	Value , d^{-1}
Element	Potatoes k_{16}
Sulphur	$3.46\ 10^{+2}$
Chromium	$1.73\ 10^{-1}$
Manganese	$5.76\ 10^{+1}$
Iron	$1.73\ 10^{-1}$
Cobalt	$5.76\ 10^{+0}$
Zinc	$2.88\ 10^{+2}$
Strontium	$2.30\ 10^{+1}$
Yttrium	$5.76\ 10^{+0}$
Zirconium	$5.76\ 10^{-2}$
Niobium	$5.76\ 10^{+0}$
Ruthenium	$5.76\ 10^{+0}$
Antimony	$5.76\ 10^{+0}$
Tellurium	$5.76\ 10^{-1}$
Iodine	$1.15\ 10^{+1}$
Caesium	$3.46\ 10^{+0}$
Barium	$2.88\ 10^{+0}$
Lanthanum	$1.73\ 10^{+0}$
Cerium	$5.76\ 10^{-1}$
Plutonium	$2.88\ 10^{-2}$
Americium	$4.61\ 10^{-2}$
Curium	$1.73\ 10^{-2}$

B3.1 Running model for routine release applications

Application: continuous deposition per unit area for one year

The growth of the root vegetable or potato crop is modelled explicitly for the first year.

A growing period of 120 days is assumed. For the remainder of the first year the land is assumed to be fallow and transfers from the soil to the plant are set to zero. In addition, the transfer coefficient k_{21} is set to $8.64\ 10^{4}\ d^{-1}$ at the start of the fallow period to represent the ploughing back of the remaining plant into the soil.

At the start of the second year the inventories of all the plant and tuber compartments are set to zero and deposition onto plant and soil is stopped.

The process of initial resuspension and weathering onto the plant is also stopped, ie, transfer coefficients k_{12} and k_{21} are set to zero.

The model is run continuously in this mode for the remaining times required with continuous cropping at 1 crop/year represented by the transfer coefficients k_{55} and k_{66}.

B3.2 Calculation and activity concentrations in potatoes

For a continuous deposition rate (Bq km^{-2} s^{-1})

Time-integrated concentration in first year = concentration at 120 days (Bq y kg^{-1})

$$= \frac{I_5 \times F_r + I_6 \times F_r}{Y}$$

where I_5, and I_6 are the inventories of activity in compartments 5 and 6.

F_r is the fraction of activity retained after processing (see Table 3.18) and Y is the yield of produce in kg km^{-2} (see Table 3.18).

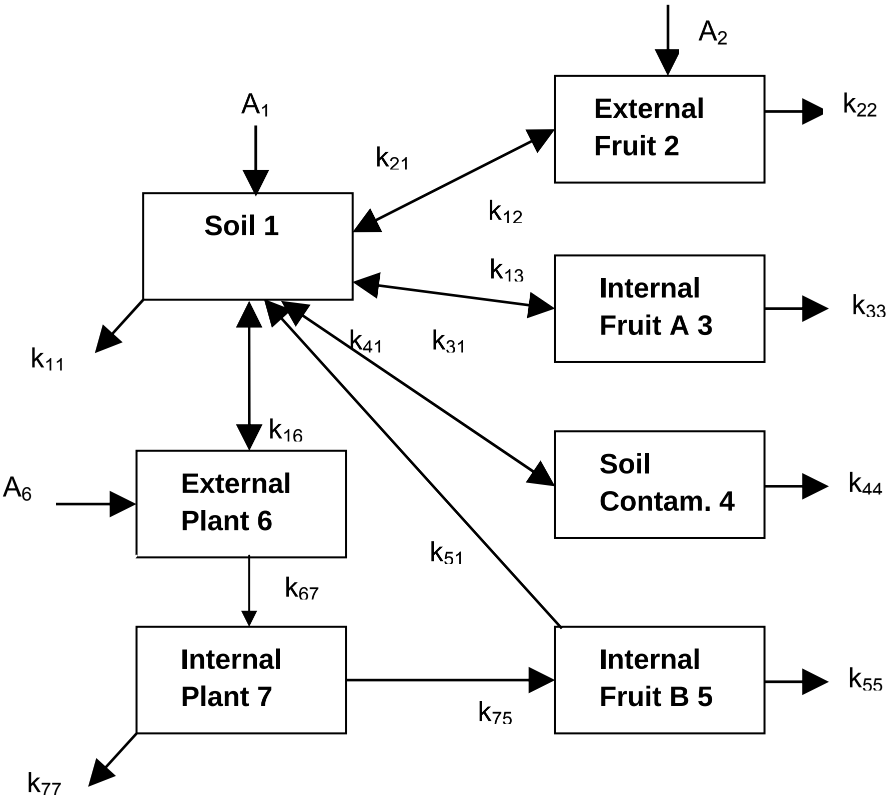
Time-integrated concentration to subsequent times (t) after 1 year =

time-integrated concentration at time t

$$= \frac{I_5 \times F_r + I_6 \times F_r}{Y}$$

where I_5 and I_6 are the integrals of activity in compartments 5 and 6.

B4 FRUIT MODEL



Inputs A_1 = $(1 - p_f - p_p)$ x deposition rate

A_2 = p_f x deposition rate

A_6 = p_p x deposition rate

where p_f and p_p are the interception factors for fruit and plant respectively (see Table 3.18).

Notes

1 External fruit is for direct deposition and initial resuspension onto the fruit.
External plant is for deposition and initial resuspension onto the plant.

2 Internal fruit A is for root uptake.

3 Soil contamination is for soil contamination of the fruit.

4 Internal fruit B and internal plant model the translocation of activity from the plant surface through transfer coefficients k_{67}, k_{75} and k_{51}.

5 After the first year's harvest, loss from External plant through leaf-fall is modelled as a fast loss process to soil through k_{61}.

6 Cropping of the plant throughout the year is represented by the transfer coefficients k_{22}, k_{33}, k_{44}, k_{55} and k_{77} based on one crop per year.

Element independent transfer coefficients

Rate constant	Value, d^{-1}
$k_{2\,1}$	$4.95\ 10^{-2}$
$k_{6\,1}$	$4.95\ 10^{-2}$
$k_{3\,1}$	$8.64\ 10^{4}$
$k_{1\,4}$	$5.06\ 10^{-2}$
$k_{4\,1}$	$8.64\ 10^{4}$
$k_{1\,2}$	$3.94\ 10^{-8}$
$k_{1\,6}$	$4.16\ 10^{-4}$
k_{11}	$1.90\ 10^{-5}$
$k_{2\,2}$, k_{33}, k_{44}, k_{55}, k_{66},	$2.74\ 10^{-3}$

Element dependent transfer coefficients for translocation

Mobility of element[1]	Value, d^{-1}		
	K_{67}	K_{75}	K_{51}
Mobile	$1.0\ 10^{-1}$	$3.0\ 10^{-2}$	$7.0\ 10^{-2}$
Semi-mobile	$1.5\ 10^{-3}$	$3.0\ 10^{-2}$	$7.0\ 10^{-2}$
Immobile	0	0	0

Note:

1 Elements are grouped by their mobility and transfer coefficients are chosen reflecting this. Details of element mobility are given in Table 3.19.

Element dependent transfer coefficients for root uptake

Element	Value d^{-1} k_{13} (orchard fruit)
Sulphur	$1.95\ 10^{2}$
Chromium	$9.73\ 10^{-2}$
Manganese	$3.24\ 10^{1}$
Iron	$1.30\ 10^{-1}$
Cobalt	$1.62\ 10^{0}$
Zinc	$3.24\ 10^{2}$
Strontium	$6.49\ 10^{0}$
Yttrium	$3.24\ 10^{0}$
Zirconium	$3.24\ 10^{-2}$
Niobium	$3.24\ 10^{0}$
Ruthenium	$3.24\ 10^{0}$
Antimony	$3.24\ 10^{0}$
Tellurium	$9.73\ 10^{-1}$
Iodine	$6.49\ 10^{0}$
Caesium	$9.73\ 10^{-1}$
Barium	$3.24\ 10^{0}$
Lanthanum	$9.73\ 10^{-1}$
Cerium	$3.24\ 10^{-1}$
Plutonium	$3.24\ 10^{-3}$
Americium	$3.24\ 10^{-3}$
Curium	$6.49\ 10^{-3}$

B4.1 Running model for routine release applications

Application: continuous deposition per unit area for one year

The growth of the fruit is modelled explicitly for the first year.

A growing period of 150 days is assumed. For the remainder of the first year the land is assumed to be fallow and transfers from the soil to the plant are set to zero. In addition, the transfer coefficient k_{61} is set to $8.64\ 10^{4}\ d^{-1}$ at the start of the fallow period to represent leaf-fall and so the remaining activity deposited on the leaves being transferred to the soil compartment. Fast-loss ($8.64\ 10^{4}\ d^{-1}$) transfers are applied to all fruit compartments from the start of the fallow season to represent cropping.

At the start of the second year the inventories of all the plant and fruit compartments are set to zero. Deposition onto plant and soil is stopped.

The process of initial resuspension and weathering onto the plant is also stopped, ie, transfer coefficients k_{12}, k_{21}, k_{16} and k_{61} are set to zero.

The model is run continuously in this mode for the remaining times required with continuous cropping at 1 crop/year represented by the transfer coefficients k_{22}, k_{33}, k_{44}, k_{55} and k_{77}.

B4.2 Calculation and activity concentrations in fruit

For a continuous deposit rate (Bq km^{-2} s^{-1})

Time-integrated concentration in first year = concentration at 150 days (Bq y kg^{-1})

$$= \frac{I_2 \times I_3 + I_4 \times I_5}{Y}$$

where I_2, I_3, I_4 and I_5 are the inventories of activity in compartments 2, 3, 4 and 5.

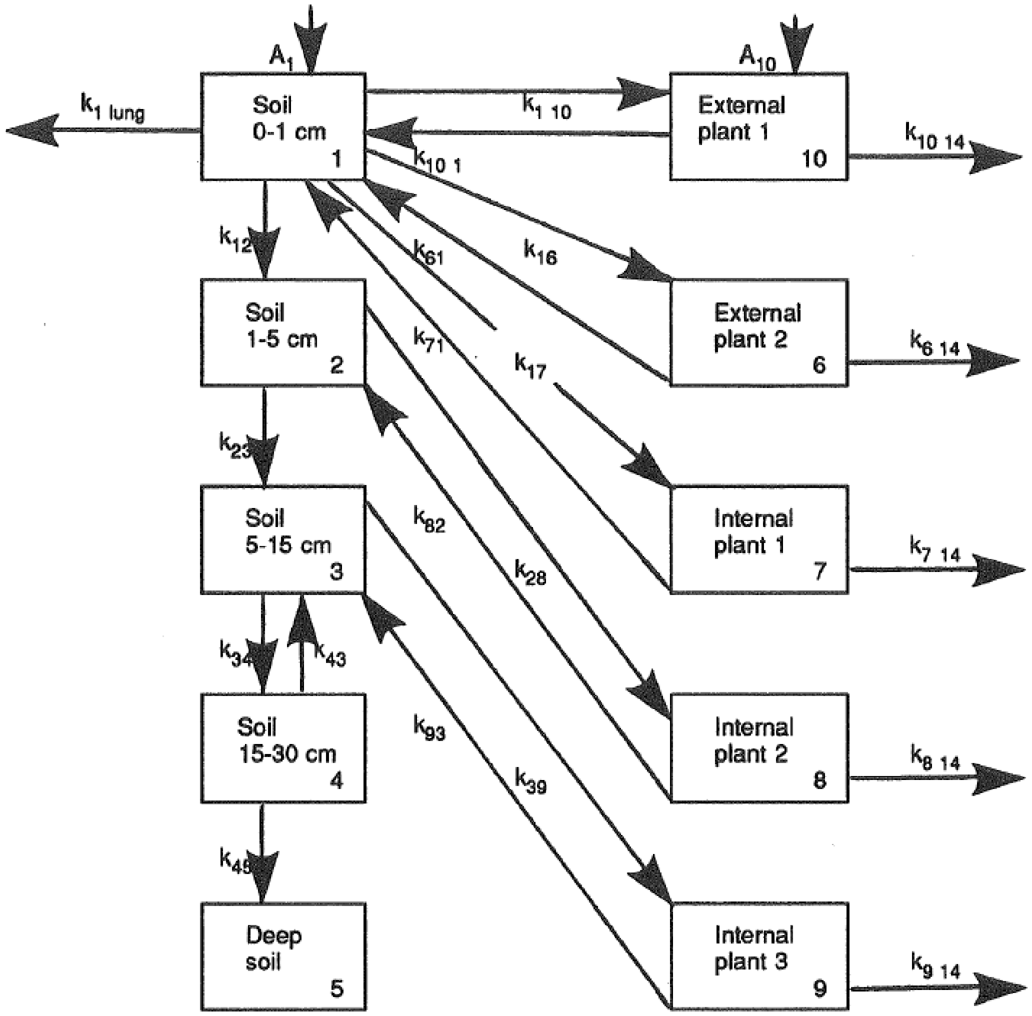
Y is the yield of produce in kg km^{-2} (see Table 3.18).

Time-integrated concentration to subsequent times (t) after 1 year = time-integrated concentration at time t

$$= \frac{I_2 \times I_3 + I_4 \times I_5}{Y}$$

where I_2, I_3, I_4 and I_5 are the integrals of activity in compartments 2, 3, 4 and 5.

B5 UNDISTURBED PASTURE MODELS



Inputs A_1 = (1 - p) x deposition rate

A_{10} = p x deposition rate

where p is the interception factor (see Table 3.18).

Notes

1. This is the basic model for undisturbed pasture. There is an additional part of the model for caesium which is described later.
2. External plant (1) is for direct deposition and initial resuspension.
   External plant (2) is for surface soil contamination of the plant, represents all soil consumed by an animal on the pasture.
3. The internal plant compartments represent root uptake from the different layers of soil.
4. $k_{1\,10}$ represents resuspension on to the plant surface, and $k_{10\,1}$, the losses due to weathering processes.
5. $k_{6\,14}$, $k_{7\,14}$, $k_{8\,14}$, $k_{9\,14}$ and $k_{10\,14}$ represent losses from the pasture due to its consumption by animals.
6. $k_{1\,lung}$ represents inhalation by the animal of resuspended material from the soil.

Element-independent transfer coefficients

Transfer coefficient	Value, d^{-1} Pasture
k_{12} [1]	$6.64\ 10^{-4}$
k_{23}	$1.72\ 10^{-4}$
k_{34}	$1.07\ 10^{-4}$
k_{43}	$4.03\ 10^{-6}$
k_{45}	$3.80\ 10^{-5}$
$k_{1\ 10}$	$2.16\ 10^{-7}$
$k_{10\ 1}$ [2]	$4.95\ 10^{-2}$
k_{16} [3]	$2.30\ 10^{1}$
k_{61}	$8.64\ 10^{4}$
k_{71}	$8.64\ 10^{4}$
k_{82}	$8.64\ 10^{4}$
k_{93} [4]	$8.64\ 10^{4}$

Notes:

1 k_{12} = $1.27\ 10^{-3}$ for strontium.

2 $k_{10\ 1}$ = $2.48\ 10^{-2}$ during winter months, ie, November - April.

3 k_{16} = $1.15\ 10^{2}$ for sheep.

4 k_{93} = 0 for caesium.

Animal dependent transfer coefficients

	Value, d^{-1}	
Transfer coefficient	Cows	Sheep
$k_{6\ 14}$, $k_{7\ 14}$, $k_{8\ 14}$, $k_{9\ 14}$, $k_{10\ 14}$	$5.2\ 10^{-2}$	$7.5\ 10^{-3}$
k1,lung[1]	$5.18\ 10^{-10}$	$4.32\ 10^{-11}$
k1,lung[2]	$3.27\ 10^{-10}$	$2.72\ 10^{-11}$

Notes:

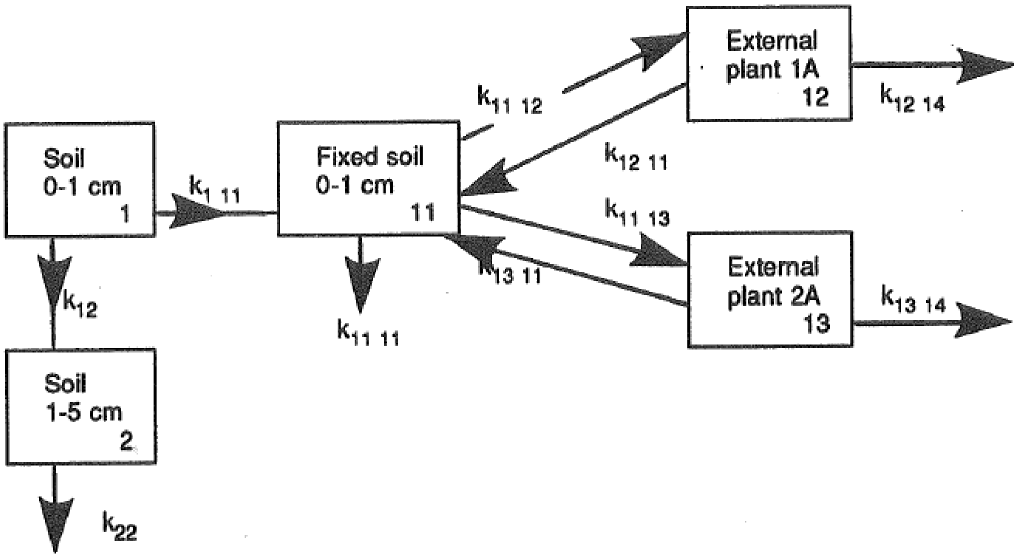
1 non-actinides

2 actinides

Element dependent transfer coefficients for root uptake

	Value, d^{-1}		
Element	k_{17}	k_{28}	k_{39}
Sulphur	$1.73\ 10^{3}$	$4.32\ 10^{2}$	$1.73\ 10^{2}$
Chromium	$8.64\ 10^{-1}$	$2.16\ 10^{-1}$	$8.64\ 10^{-2}$
Manganese	$2.88\ 10^{2}$	$7.20\ 10^{1}$	$2.88\ 10^{1}$
Iron	$1.15\ 10^{0}$	$2.88\ 10^{-1}$	$1.15\ 10^{-1}$
Cobalt	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Zinc	$2.88\ 10^{3}$	$7.20\ 10^{2}$	$2.88\ 10^{2}$
Strontium	$5.76\ 10^{2}$	$3.60\ 10^{1}$	$1.44\ 10^{1}$
Yttrium	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Zirconium	$2.88\ 10^{-1}$	$7.20\ 10^{-2}$	$2.88\ 10^{-2}$
Niobium	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Ruthenium	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Antimony	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Tellurium	$1.44\ 10^{1}$	$3.60\ 10^{0}$	$1.44\ 10^{0}$
Iodine	$5.76\ 10^{1}$	$1.44\ 10^{1}$	$5.76\ 10^{0}$
Caesium	$8.64\ 10^{1}$	$2.16\ 10^{1}$	$0.00\ 10^{0}$
Barium	$2.88\ 10^{1}$	$7.20\ 10^{0}$	$2.88\ 10^{0}$
Lanthanum	$8.64\ 10^{0}$	$2.16\ 10^{0}$	$8.64\ 10^{-1}$
Cerium	$2.88\ 10^{0}$	$7.20\ 10^{-1}$	$2.88\ 10^{-1}$
Plutonium	$2.88\ 10^{-1}$	$7.20\ 10^{-2}$	$2.88\ 10^{-2}$
Americium	$2.88\ 10^{0}$	$7.20\ 10^{-1}$	$2.88\ 10^{-1}$
Curium	$2.88\ 10^{0}$	$7.20\ 10^{-1}$	$2.88\ 10^{-1}$

B5.1 Additional compartments and transfer coefficients for caesium on pasture (used in conjunction with the basic model for undisturbed pasture)



Notes

1 External plant 1A is used for fixed activity resuspended on to plant surfaces.
External plant 2A is used for surface contamination by soil containing fixed activity, and includes any fixed activity consumed by animals.

2 $k_{1\,11}$ represents the process of fixation.

3 k_{22} represents the loss of activity due to the fixation process in the 1-5 cm layer of soil.

4 $k_{11\,12}$ is a loss term representing migration from the surface soil layer of fixed activity.

Additional transfer coefficients used for caesium

	Value, d^{-1}	
	Pasture	
Transfer coefficient	Cow	Sheep
$k_{1\,11}$	$2.11\ 10^{-3}$	$2.11\ 10^{-3}$
k_{22}	$2.11\ 10^{-3}$	$2.11\ 10^{-3}$
$k_{11\,11}$	$6.65\ 10^{-4}$	$6.65\ 10^{-4}$
$k_{11\,12}$	$2.16\ 10^{-7}$	$2.16\ 10^{-7}$
$k_{12\,11}$	$4.95\ 10^{-2}$	$4.95\ 10^{-2}$
$k_{11\,13}$	$2.30\ 10^{1}$	$1.15\ 10^{2}$
$k_{13\,11}$	$8.64\ 10^{4}$	$8.64\ 10^{4}$
$k_{12\,14}$, $k_{13\,14}$	$5.2\ 10^{-2}$	$7.5\ 10^{-3}$

B5.2 Running the model for routine release applications

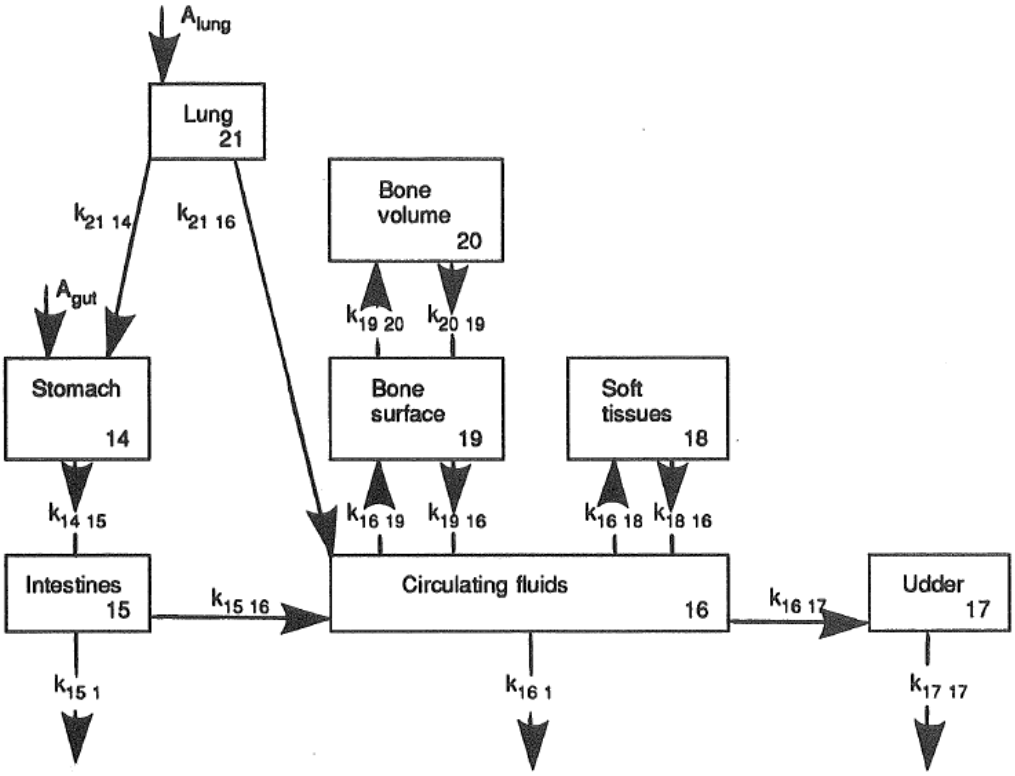
The pasture model is run continuously with deposition onto plant and soil for the period over which deposition occurs.

If the model is run for subsequent times with no deposition the transfer coefficients to external plant 1 representing initial resuspension and weathering (k_{110} and k_{101}) should be set to zero. For caesium, the transfer coefficients $k_{11\ 12}$, and $k_{12\ 11}$ should also be set to zero.

B6 COW MODELS

Several cow models of varying complexity are used as described in Section 3.3.3.4 of the main text. Details of the models are given below.

B6.1 Cow model for isotopes of strontium



Notes

1 $k_{16\ 1}$ and $k_{15\ 1}$ represent return to the soil (compartment 1) due to excretion processes.

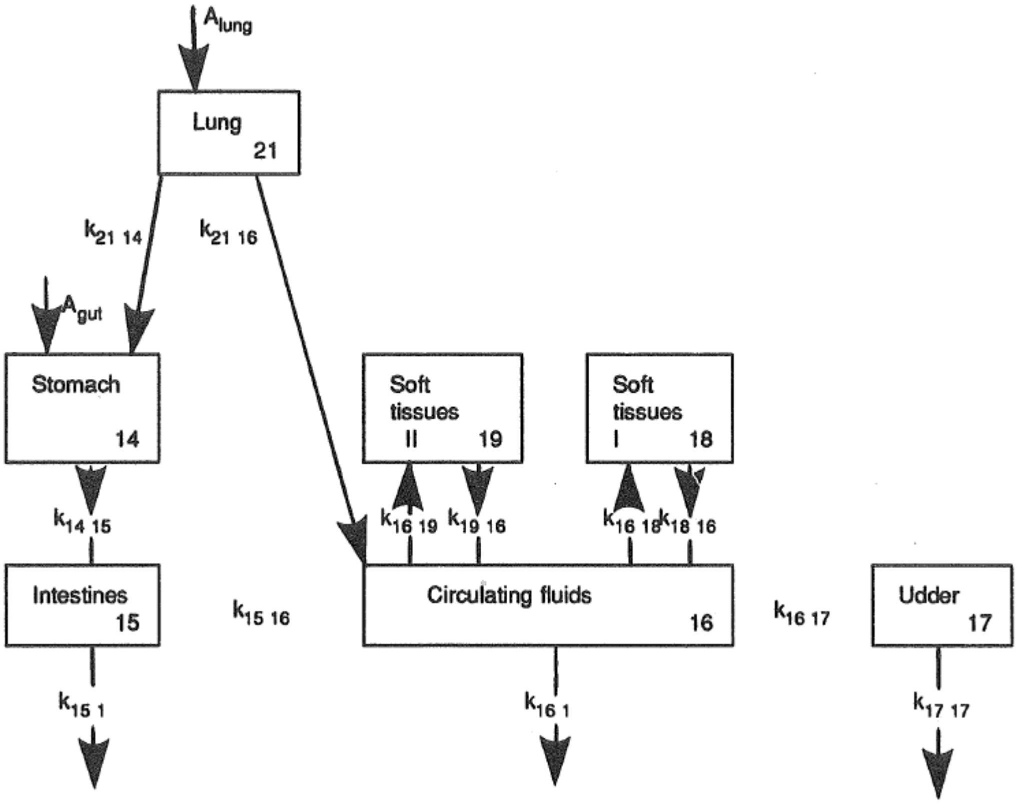
2 There is an additional loss from all compartments to represent the periodic slaughter of cows; the value of this transfer coefficient is 4.56 10^{-4} d^{-1}.

3 Compartment 18 (soft tissue) represents the rest of the body that is not modelled explicitly.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 15}$	$7.00\ 10^{-1}$
$k_{15\ 1}$	$5.00\ 10^{0}$
$k_{15\ 16}$	$5.56\ 10^{-1}$
$k_{16\ 17}$	$1.26\ 10^{-1}$
$k_{16\ 18}$	$2.24\ 10^{-1}$
$k_{18\ 16}$	$2.30\ 10^{-1}$
$k_{16\ 19}$	$1.61\ 10^{0}$
$k_{19\ 16}$	$6.43\ 10^{-2}$
$k_{19\ 20}$	$1.10\ 10^{-1}$
$k_{20\ 19}$	$8.91\ 10^{-3}$
$k_{21\ 14}$	$2.11\ 10^{1}$
$k_{21\ 16}$	$2.70\ 10^{1}$
$k_{17\ 17}$	$4.00\ 10^{0}$
$k_{16\ 1}$	$7.76\ 10^{-1}$

B6.2 Cow model for isotopes of caesium



Notes

1 Soft tissue (1) (compartment 18) represents the diffusion of caesium from the blood to the rest of the body.

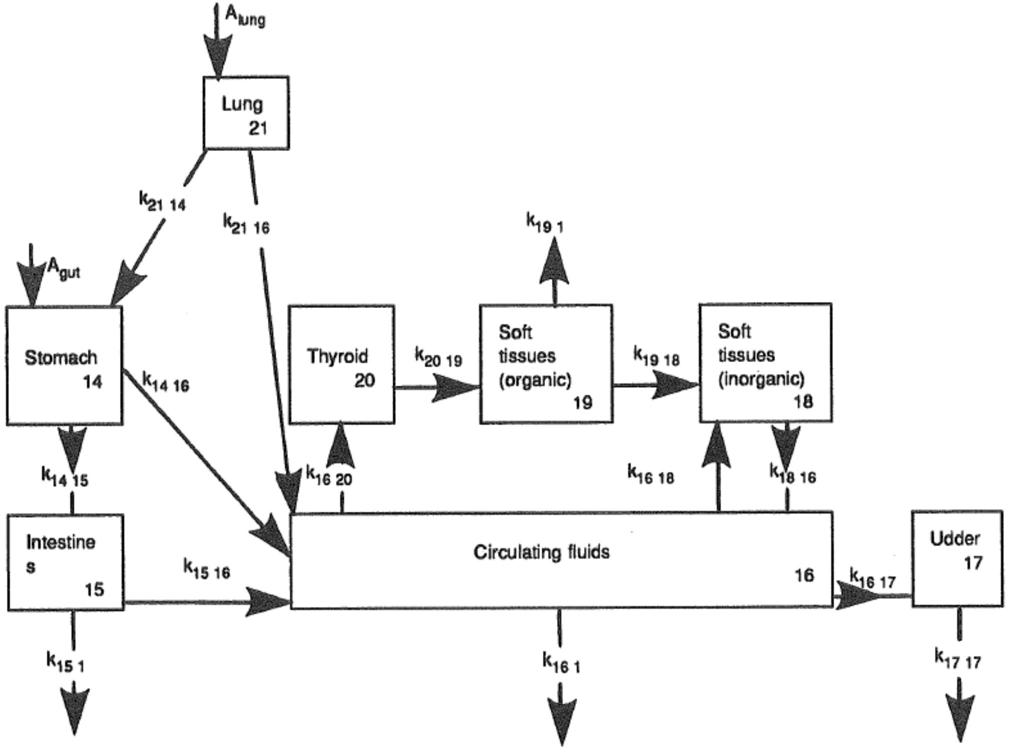
2 Soft tissue (2) represents a slower concentrating mechanism of caesium in the soft tissues.

3 Periodic slaughter is represented by losses from all compartments, the value of the transfer coefficient is $4.56\ 10^{-4}\ d^{-1}$.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 15}$	$7.00\ 10^{-1}$
$k_{15\ 1}$	$5.00\ 10^{0}$
$k_{15\ 16}$	$1.48\ 10^{1}$
$k_{16\ 1}$	$8.53\ 10^{-1}$
$k_{16\ 17}$	$6.11\ 10^{-2}$
$k_{16\ 19}$	$2.53\ 10^{-1}$
$k_{16\ 18}$	$5.65\ 10^{-1}$
$k_{17\ 17}$	$4.00\ 10^{0}$
$k_{18\ 16}$	$2.97\ 10^{-1}$
$k_{19\ 16}$	$2.65\ 10^{-2}$
$k_{21\ 14}$	$2.11\ 10^{1}$
$k_{21\ 16}$	$2.70\ 10^{1}$

B6.3 Cow model for isotopes of iodine



Notes

1 $k_{14\ 16}$ represents the early absorption of iodine from the rumen of the cow.

2 The storage of iodine in the soft tissues is represented by two compartments (18 and 19). The organic iodine produced in the thyroid is re-distributed throughout the soft tissues and organs of the body where it remains for some time before being broken down into inorganic iodine.

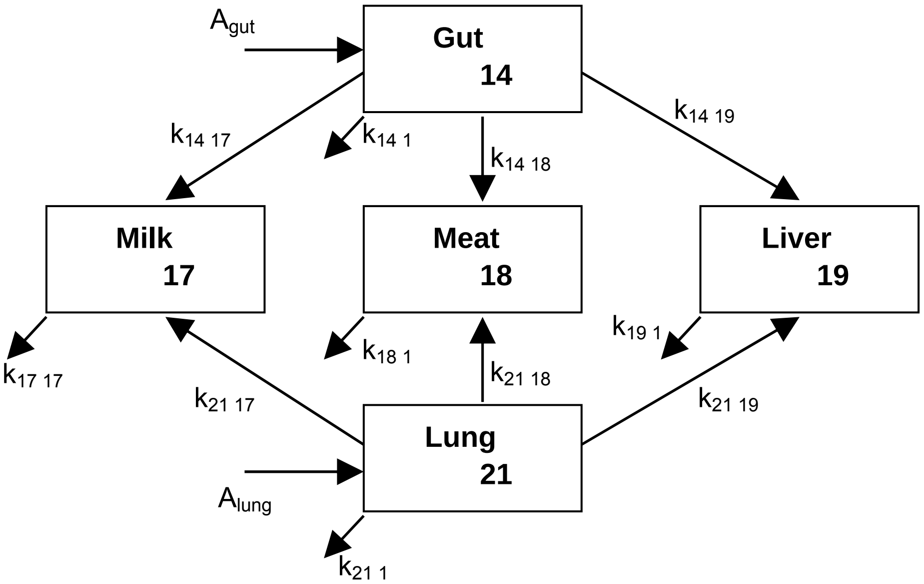
3 $k_{19\ 1}$ represents the excretion of the organic fraction of iodine in the circulating fluids.

4 Periodic slaughter is represented by losses from all compartments, the value of the transfer coefficient is $4.56\ 10^{-4}\ d^{-1}$.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\,15}$	$7.00\ 10^{-1}$
$k_{15\,1}$	$5.00\ 10^{0}$
$k_{15\,16}$	$8.00\ 10^{-1}$
$k_{14\,16}$	$4.90\ 10^{-1}$
$k_{16\,1}$	$2.50\ 10^{0}$
$k_{16\,17}$	$3.35\ 10^{-1}$
$k_{17\,17}$	$4.00\ 10^{0}$
$k_{16\,18}$	$9.19\ 10^{0}$
$k_{18\,16}$	$2.40\ 10^{0}$
$k_{16\,20}$	$7.21\ 10^{-1}$
$k_{20\,19}$	$7.34\ 10^{-2}$
$k_{19\,1}$	$1.12\ 10^{-1}$
$k_{19\,18}$	$5.71\ 10^{-2}$
$k_{21\,14}$	$2.11\ 10^{1}$
$k_{21\,16}$	$2.70\ 10^{1}$

B6.4 Cow model for other non-actinide elements



Note

1 Periodic slaughter is represented by losses from all compartments at a rate of $4.56\ 10^{-4}\ d^{-1}$.

Element-independent transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 1}$	$1.11\ 10^{0}$
$k_{21\ 1}$	$1.00\ 10^{0}$
$k_{17\ 17}$	$4.57\ 10^{-4}$

Element dependent transfer coefficients

Element	Transfer coefficient, d^{-1}							
	k_{14-18}	k_{18-1}	k_{14-19}	k_{19-1}	k_{14-17}	k_{21-18}	k_{21-19}	k_{21-17}
Sulphur	$1.38\ 10^{0}$	$6.33\ 10^{-3}$	$3.59\ 10^{-2}$	$6.33\ 10^{-3}$	$6.31\ 10^{-1}$	$1.91\ 10^{-1}$	$5.09\ 10^{-3}$	$8.93\ 10^{-2}$
Chromium	$2.77\ 10^{-2}$	$2.11\ 10^{-2}$	$7.21\ 10^{-4}$	$2.11\ 10^{-2}$	$1.14\ 10^{-3}$	$1.11\ 10^{-2}$	$2.89\ 10^{-4}$	$4.57\ 10^{-4}$
Manganese	$4.08\ 10^{-3}$	$3.17\ 10^{-2}$	$3.65\ 10^{-3}$	$2.71\ 10^{-2}$	$2.24\ 10^{-3}$	$4.23\ 10^{-3}$	$3.79\ 10^{-3}$	$2.33\ 10^{-3}$
Iron	$9.79\ 10^{-5}$	$3.80\ 10^{-4}$	$1.02\ 10^{-2}$	$3.80\ 10^{-4}$	$5.60\ 10^{-4}$	$1.02\ 10^{-4}$	$1.06\ 10^{-2}$	$5.82\ 10^{-4}$
Cobalt	$9.71\ 10^{-5}$	$3.80\ 10^{-3}$	$2.53\ 10^{-4}$	$3.80\ 10^{-3}$	$1.11\ 10^{-3}$	$1.01\ 10^{-4}$	$2.63\ 10^{-4}$	$1.15\ 10^{-3}$
Zinc	$1.25\ 10^{-3}$	$2.37\ 10^{-3}$	$3.26\ 10^{-5}$	$2.37\ 10^{-3}$	$3.44\ 10^{-2}$	$4.82\ 10^{-4}$	$1.26\ 10^{-5}$	$1.33\ 10^{-2}$
Yttrium	$1.21\ 10^{-5}$	$4.75\ 10^{-5}$	$3.16\ 10^{-6}$	$4.75\ 10^{-5}$	$2.22\ 10^{-4}$	$1.31\ 10^{-3}$	$3.43\ 10^{-4}$	$2.41\ 10^{-2}$
Zirconium	$2.42\ 10^{-6}$	$9.50\ 10^{-2}$	$6.32\ 10^{-8}$	$9.50\ 10^{-2}$	$6.66\ 10^{-6}$	$2.01\ 10^{-5}$	$5.24\ 10^{-7}$	$5.52\ 10^{-5}$
Niobium	$4.85\ 10^{-7}$	$6.33\ 10^{-3}$	$1.26\ 10^{-8}$	$6.33\ 10^{-3}$	$4.44\ 10^{-6}$	$4.02\ 10^{-6}$	$1.05\ 10^{-7}$	$3.68\ 10^{-5}$
Ruthenium	$6.93\ 10^{-4}$	$2.71\ 10^{-3}$	$1.81\ 10^{-5}$	$2.71\ 10^{-3}$	$1.11\ 10^{-5}$	$1.28\ 10^{-3}$	$3.33\ 10^{-5}$	$2.05\ 10^{-5}$
Antimony	$1.00\ 10^{-2}$	$3.80\ 10^{-2}$	$2.61\ 10^{-2}$	$3.80\ 10^{-2}$	$1.15\ 10^{-3}$	$1.04\ 10^{-2}$	$2.72\ 10^{-2}$	$1.20\ 10^{-3}$
Tellurium	$5.10\ 10^{-2}$	$3.80\ 10^{-2}$	$1.33\ 10^{-3}$	$3.80\ 10^{-2}$	$5.84\ 10^{-3}$	$2.49\ 10^{-2}$	$6.50\ 10^{-4}$	$2.85\ 10^{-3}$
Barium	$2.71\ 10^{-3}$	$2.11\ 10^{-2}$	$7.08\ 10^{-5}$	$2.11\ 10^{-2}$	$5.59\ 10^{-3}$	$1.72\ 10^{-3}$	$4.48\ 10^{-5}$	$3.54\ 10^{-3}$
Lanthanum	$2.43\ 10^{-4}$	$1.90\ 10^{-4}$	$2.53\ 10^{-4}$	$1.90\ 10^{-4}$	$2.22\ 10^{-4}$	$4.42\ 10^{-2}$	$4.61\ 10^{-2}$	$4.05\ 10^{-2}$
Cerium	$4.85\ 10^{-5}$	$1.90\ 10^{-4}$	$2.53\ 10^{-4}$	$1.90\ 10^{-4}$	$1.10\ 10^{-3}$	$1.01\ 10^{-2}$	$5.28\ 10^{-2}$	$2.32\ 10^{-1}$

For cow model for actinides see Section B8.

B6.5 Running the models for routine release applications

The assumption is made that cows are continuously grazing pasture throughout the year.

The cow model is linked to the undisturbed pasture model described in Section B5 in the following way:

(a) For cows on pasture:

Intake by ingestion: $A_{gut} = k_{6\ 14} + k_{7\ 14} + k_{8\ 14} + k_{9\ 14} + k_{10\ 14}$

For caesium: $A_{gut} = k_{6\ 14} + k_{7\ 14} + k_{8\ 14} + k_{9\ 14} + k_{10\ 14} + k_{12\ 14} + k_{13\ 14}$

Intake by inhalation: $A_{lung} = k_{1\ lung}$

(b) For cows inhaling:

Inhalation of the radioactive plume is not modelled in PC-CREAM 08

The models are run for the period required with periodic slaughter of the herd included.

B6.6 Calculation of activity concentrations in animal tissues and milk

(a) For cows on pasture and a continuous deposition rate (Bq km^{-2} s^{-1}):

Concentration in meat (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

strontium: concentration = $I_{18}/N_c \times M_c$

iodine and caesium: concentration = $(I_{18} + I_{19})/N_c \times M_c$

other non-actinide elements: concentration = ~~I_{17}~~ $I_{18}/N_c \times M_c$

for actinides see Section B8

where I_{18} and I_{19} are the inventories (or integrals) of activity in compartments 18 and 19 of the models representing meat.

N_c is the number of cows per unit area (km^{-2}) - see Table 3.20

M_c is the mass of meat per animal (kg)- see Table 3.20

Concentration in liver (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

strontium: concentration = $I_{18}/N_c \times M_L$

iodine and caesium: concentration = $(I_{18} + I_{19})/N_c \times M_L$

other non-actinide elements: concentration = $I_{19}/N_c \times M_L$

for actinides - see Section B8

where M_L is the mass of liver per animal (kg) - see Table 3.20.

I_{18} and I_{19} are the inventories (or integrals) of activity in compartments 18 and 19 representing liver.

Concentration in milk (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

strontium, iodine and caesium: concentration = $I_{17} \times k_{17\,17}/N_c \times Y_m$

other non-actinide elements: concentration = $(I_{14} \times k_{14\,17}) + (I_{21} \times k_{21\,17})/N_c \times Y_m$

where Y_m is the daily yield of milk per cow (l d^{-1})

I_{17} is the inventory (or integral) in compartment 17 of the model representing the udder.

I_{14} and I_{21} are the inventories (or integrals) in compartments 14 and 21 of the model representing the gut and lung respectively.

$k_{14\ 17}$ and $k_{21\ 17}$ are the transfer coefficients from gut to milk and lung to milk, respectively (d^{-1}).

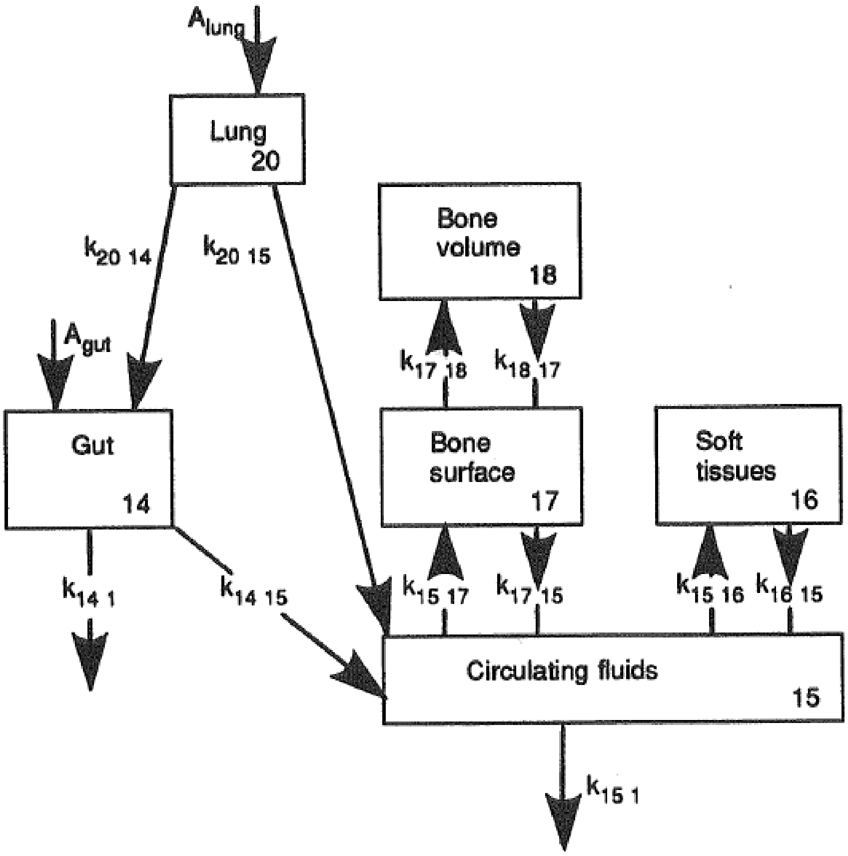
(b) For cows inhaling:

Inhalation of the radioactive plume is not modelled in PC-CREAM 08.

B7 SHEEP MODEL

Several sheep models of varying complexity are used as described in Section 3.3.3.4 of the main report. Details of the models are given below.

B7.1 Sheep model for isotopes of strontium



Notes

1 $k_{14\ 1}$ and $k_{15\ 1}$ represent return to the soil (compartment 1) due to excretion processes.

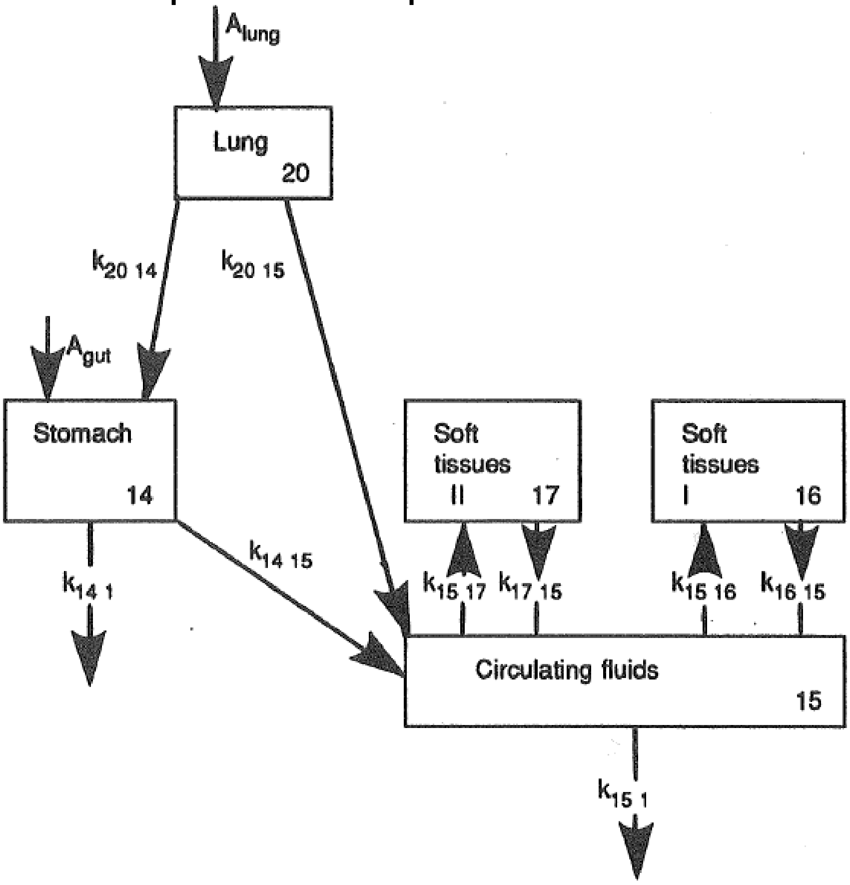
2 There is an additional loss from all compartments to represent the periodic slaughter of sheep; the value of this transfer coefficient is $2.74\ 10^{-3}\ d^{-1}$.

3 Compartment 16 (soft tissue) represents the rest of the body that is not modelled explicitly.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 1}$	$7.10\ 10^{-1}$
$k_{14\ 15}$	$1.97\ 10^{-1}$
$k_{15\ 1}$	$7.53\ 10^{-1}$
$k_{15\ 16}$	$1.50\ 10^{0}$
$k_{16\ 15}$	$8.93\ 10^{0}$
$k_{15\ 17}$	$5.75\ 10^{0}$
$k_{17\ 15}$	$4.64\ 10^{-1}$
$k_{17\ 18}$	$1.27\ 10^{-2}$
$k_{18\ 17}$	$1.64\ 10^{-3}$
$k_{20\ 14}$	$2.11\ 10^{1}$
$k_{20\ 15}$	$2.70\ 10^{1}$

B7.2 Sheep model for isotopes of caesium



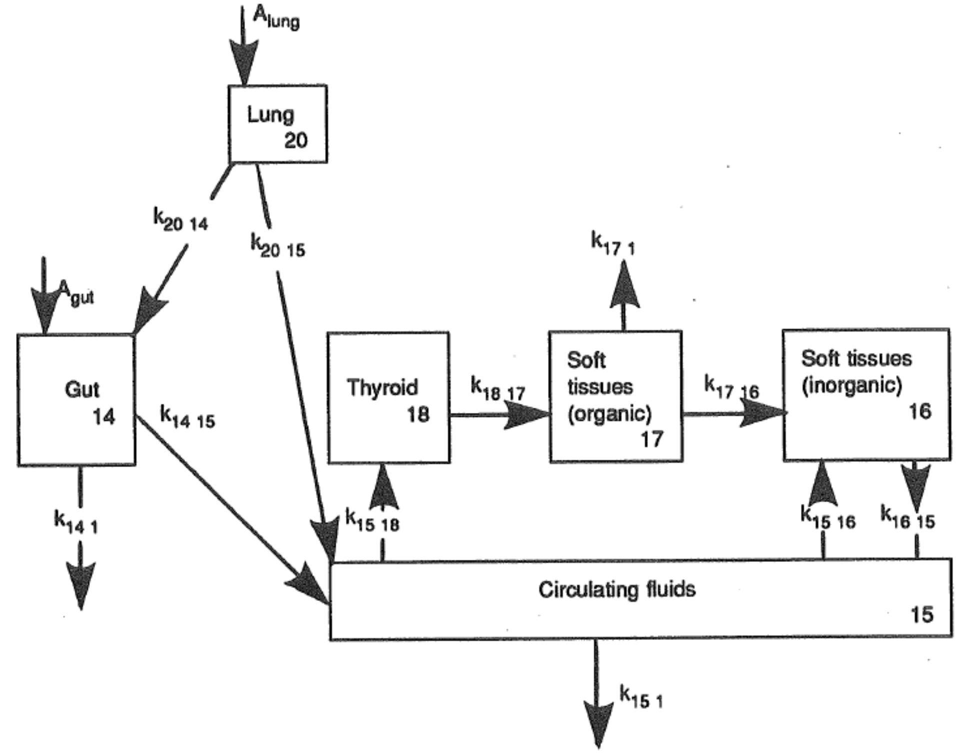
Notes

1. Soft tissue (1) (compartment 16) represents the diffusion of caesium from the blood to the rest of the body.
2. Soft tissue (2) represents a slower concentrating mechanism of caesium in the soft tissues.
3. Periodic slaughter is represented by losses from all compartments, the value of the transfer coefficient is 2.74 10^{-3} d^{-1}.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 1}$	7.10 10^{-1}
$k_{14\ 15}$	7.03 10^{2}
$k_{15\ 1}$	3.63 10^{-1}
$k_{15\ 16}$	2.03 10^{1}
$k_{16\ 15}$	4.80 10^{1}
$k_{15\ 17}$	9.75 10^{-2}
$k_{17\ 15}$	4.40 10^{-2}
$k_{20\ 14}$	2.11 10^{1}
$k_{20\ 15}$	2.70 10^{1}

B7.3 Sheep model for isotopes of iodine



Notes

1 The storage of iodine in the soft tissues is represented by two compartments (16 and 17). The organic iodine produced in the thyroid is re-distributed throughout the soft tissues and organs of the body where it remains for some time before being broken down into inorganic iodine.

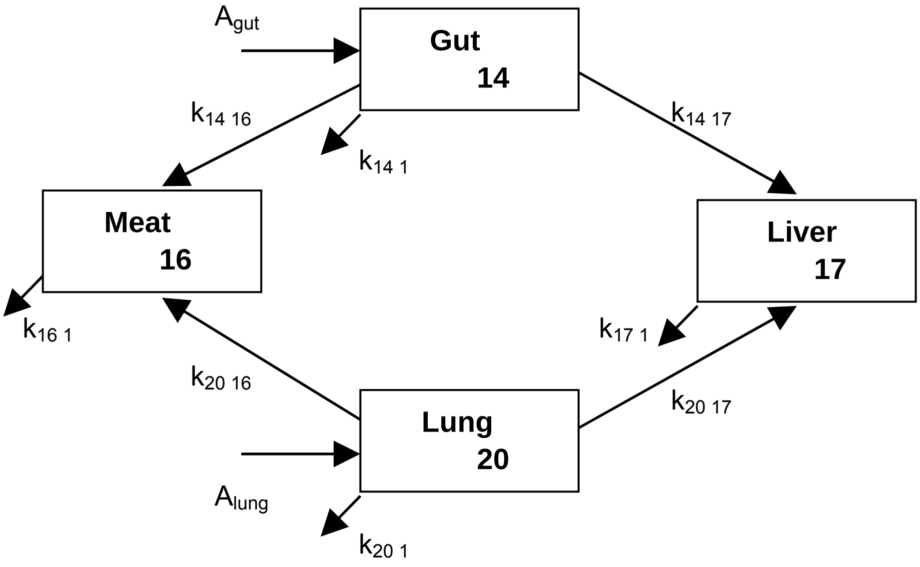
2 $k_{17\,1}$ represents the excretion of the organic fraction of iodine in the circulating fluids.

3 Periodic slaughter is represented by losses from all compartments, the value of the transfer coefficient is $2.74\ 10^{-3}\ d^{-1}$.

Transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 1}$	$7.10\ 10^{-1}$
$k_{14\ 15}$	$1.50\ 10^{-1}$
$k_{15\ 1}$	$5.88\ 10^{-1}$
$k_{15\ 16}$	$1.00\ 10^{1}$
$k_{16\ 15}$	$4.30\ 10^{0}$
$k_{15\ 18}$	$4.06\ 10^{-1}$
$k_{18\ 17}$	$7.46\ 10^{-2}$
$k_{17\ 1}$	$1.00\ 10^{-2}$
$k_{17\ 16}$	$1.30\ 10^{-1}$
$k_{20\ 14}$	$2.11\ 10^{1}$
$k_{20\ 15}$	$2.70\ 10^{1}$

B7.4 Sheep model for other non-actinide elements



Notes

1 $k_{14\ 1}$ and $k_{15\ 1}$ represent return to soil from excretion processes via blood.

2 $k_{16\ 1}$ and $k_{17\ 1}$ represent loss from the organs due to biological processes.

3 In addition, there is a loss from each compartment to represent the periodic slaughter of sheep; the value of this transfer coefficient is $2.74\ 10^{-3}\ d^{-1}$, ie, $k_{14\ 14}$, $k_{15\ 15}$, $k_{16\ 16}$, $k_{17\ 17}$.

Element-independent transfer coefficients

Transfer coefficient	Value, d^{-1}
$k_{14\ 1}$	$1.11\ 10^{0}$
$k_{20\ 1}$	$1.00\ 10^{0}$

Element dependent transfer coefficients

Element	Transfer coefficients, d^{-1}					
	k_{14-16}	k_{16-1}	k_{14-17}	k_{17-1}	k_{20-16}	k_{20-17}
Sulphur	$1.51\ 10^{0}$	$6.33\ 10^{-3}$	$2.68\ 10^{-2}$	$6.33\ 10^{-3}$	$2.47\ 10^{-1}$	$4.39\ 10^{-3}$
Chromium	$2.15\ 10^{-2}$	$2.11\ 10^{-2}$	$9.56\ 10^{-4}$	$2.11\ 10^{-2}$	$8.65\ 10^{-3}$	$3.85\ 10^{-4}$
Manganese	$3.18\ 10^{-3}$	$3.17\ 10^{-2}$	$4.85\ 10^{-3}$	$2.71\ 10^{-2}$	$3.31\ 10^{-3}$	$5.04\ 10^{-3}$
Iron	$7.66\ 10^{-5}$	$3.80\ 10^{-4}$	$1.02\ 10^{-2}$	$3.80\ 10^{-4}$	$7.95\ 10^{-5}$	$1.06\ 10^{-2}$
Cobalt	$7.59\ 10^{-5}$	$3.80\ 10^{-3}$	$3.37\ 10^{-4}$	$3.80\ 10^{-3}$	$7.87\ 10^{-5}$	$3.50\ 10^{-4}$
Zinc	$9.49\ 10^{-4}$	$2.37\ 10^{-3}$	$4.22\ 10^{-5}$	$2.37\ 10^{-3}$	$3.72\ 10^{-4}$	$1.66\ 10^{-5}$
Yttrium	$9.49\ 10^{-6}$	$4.75\ 10^{-5}$	$4.22\ 10^{-6}$	$4.75\ 10^{-5}$	$1.00\ 10^{-3}$	$4.46\ 10^{-4}$
Zirconium	$1.90\ 10^{-6}$	$9.50\ 10^{-2}$	$8.43\ 10^{-8}$	$9.50\ 10^{-2}$	$1.57\ 10^{-5}$	$6.99\ 10^{-7}$
Niobium	$3.79\ 10^{-7}$	$6.33\ 10^{-3}$	$1.69\ 10^{-8}$	$6.33\ 10^{-3}$	$3.14\ 10^{-6}$	$1.40\ 10^{-7}$
Ruthenium	$5.42\ 10^{-4}$	$2.71\ 10^{-3}$	$2.41\ 10^{-5}$	$2.71\ 10^{-3}$	$1.00\ 10^{-3}$	$4.44\ 10^{-5}$
Antimony	$7.88\ 10^{-3}$	$3.80\ 10^{-2}$	$3.50\ 10^{-2}$	$3.80\ 10^{-2}$	$8.22\ 10^{-3}$	$3.65\ 10^{-2}$
Tellurium	$3.93\ 10^{-2}$	$3.80\ 10^{-2}$	$1.75\ 10^{-3}$	$3.80\ 10^{-2}$	$1.94\ 10^{-2}$	$8.60\ 10^{-4}$
Barium	$2.11\ 10^{-3}$	$2.11\ 10^{-2}$	$9.39\ 10^{-5}$	$2.11\ 10^{-2}$	$1.34\ 10^{-3}$	$5.95\ 10^{-5}$
Lanthanum	$1.90\ 10^{-4}$	$1.90\ 10^{-4}$	$3.37\ 10^{-4}$	$1.90\ 10^{-4}$	$3.34\ 10^{-2}$	$5.95\ 10^{-2}$
Cerium	$3.80\ 10^{-5}$	$1.90\ 10^{-4}$	$3.37\ 10^{-4}$	$1.90\ 10^{-4}$	$6.51\ 10^{-3}$	$5.79\ 10^{-2}$

For actinides the model for sheep is described in Section B8.

B7.5 Running the models for routine release applications

The assumption is made that sheep are continually grazing pasture throughout the year.

The sheep model is linked to the undisturbed pasture model described in Section B5 in the following way:

(a) For sheep on pasture:

Intake by ingestion: $A_{gut} = k_{6\ 14} + k_{7\ 14} + k_{8\ 14} + k_{9\ 14} \mathbf{+\ k_{10\ 14}}$

For caesium: $A_{gut} = k_{6\ 14} + k_{7\ 14} + k_{8\ 14} + k_{9\ 14} + k_{10\ 14} + k_{12\ 14} + k_{13\ 14}$

Intake by inhalation: $A_{lung} = k_{1\ lung}$

(b) For sheep inhaling:

Inhalation of the radioactive plume is not modelled in PC-CREAM 08.

The models are run for the times required with periodic slaughter of the flock of sheep included.

B7.6 Calculation of activity concentrations in animal tissues

(a) For sheep on pasture and a continuous deposition rate (Bq km^{-2} s^{-1}):

Concentration in meat (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

strontium: concentration = $I_{16}/N_s \times M_c$

iodine and caesium: concentration = $(I_{16} + I_{17})/N_s \times M_c$

other non-actinide elements: concentration = $I_{16}/N_s \times M_c$

where I_{16}, and I_{17} are the inventories (or integrals) of activity in compartments 16 and 17 of the models representing meat.

N_s is the number of sheep per unit area (km^{-2}) (see Table 3.20)

M_c is the mass of meat per animal (kg) (see Table 3.20)

Concentration in liver (or integral of concentration)
(units kg^{-1} (or units y kg^{-1}))

strontium: concentration = $I_{16}/N_s \times M_L$

iodine and caesium: concentration = $(I_{16} + I_{17})/N_s \times M_L$

other non-actinide elements: concentration = $I_{17}/N_s \times M_L$

for actinides see Section B8

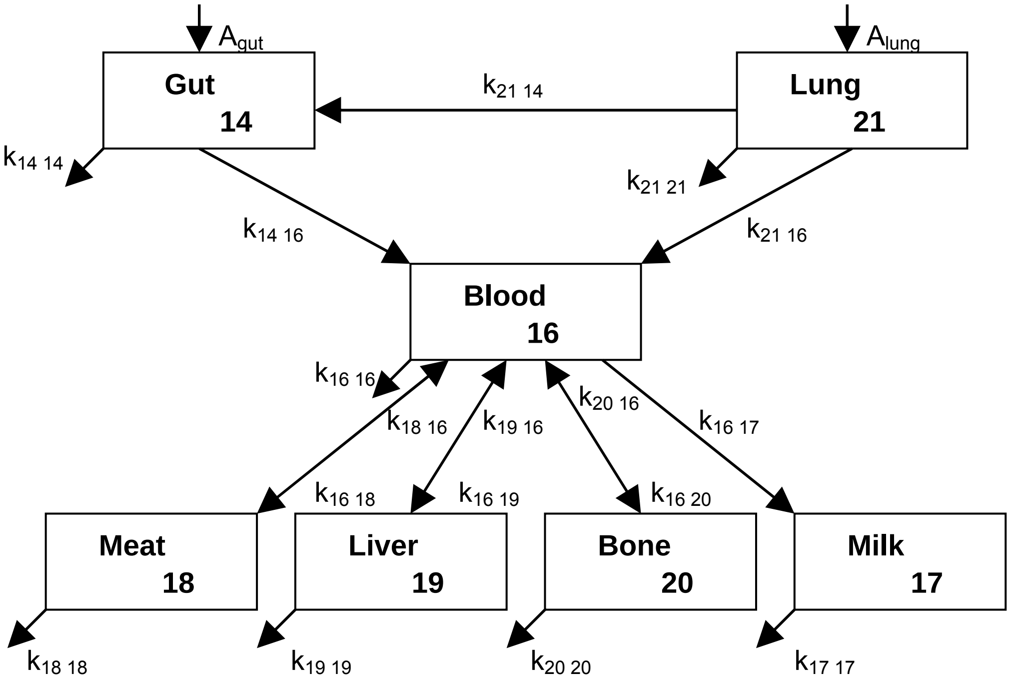
where M_L is the mass of liver per animal (kg) = see Table 3.20

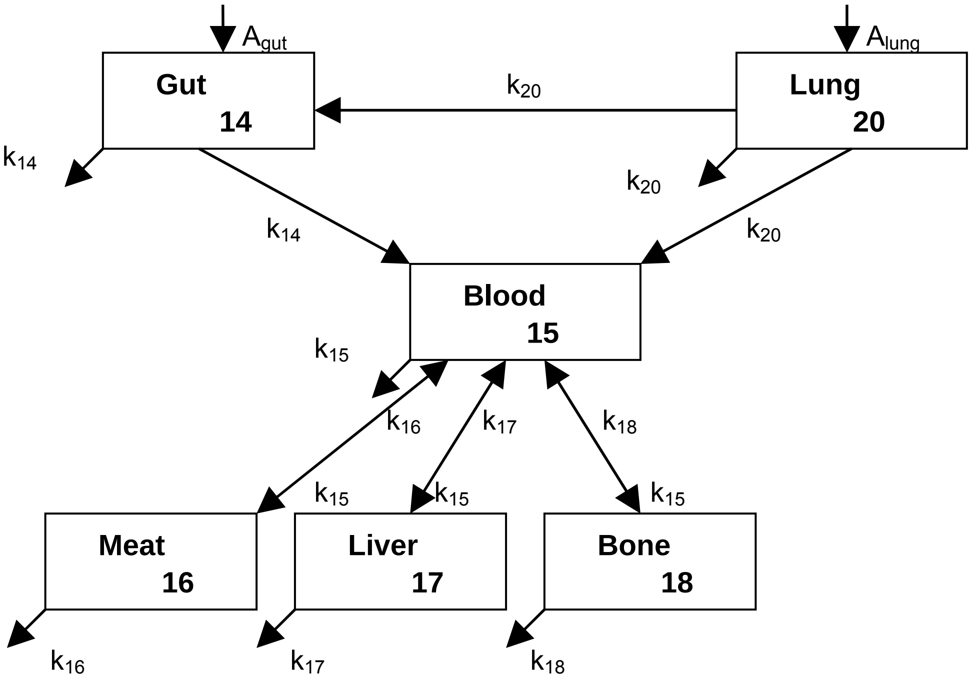
I_{16} and I_{17} are the inventories (or integrals) of activity in compartments 16 and 17 representing liver.

(b) For sheep inhaling:

Inhalation of the radioactive plume is not modelled in PC-CREAM 08.

B8 ANIMAL MODEL FOR THE ACTINIDES (CATTLE; SHEEP)





Notes

1 $k_{14\,1}$ and $k_{15\,1}$ represent return to the soil via excreta.

2 $k_{17\,17}$ represents losses due to regular milking.

3 Periodic slaughter of animals is represented by a loss from each compartment; the value of the transfer coefficient is 4.56 10^{-4} d^{-1} for cows and 2.74 10^{-3} d^{-1} for sheep.

Element independent transfer coefficients (cow)

Parameter	Value, d^{-1}
$k_{14\ 1}$	$1.11\ 10^{0}$
$k_{16\ 1}$	$1.00\ 10^{-2}$
$k_{14\ 16}$	$5.53\ 10^{-4}$
$k_{21\ 14}$	$1.04\ 10^{-1}$
$k_{21\ 16}$	$2.07\ 10^{-2}$
$k_{16\ 17}$	$1.00\ 10^{-2}$
$k_{17\ 17}$	$3.02\ 10^{0}$
$k_{16\ 18}$	$2.00\ 10^{0}$
$k_{18\ 16}$	$4.74\ 10^{-1}$
$k_{16\ 19}$	$2.00\ 10^{0}$
$k_{19\ 16}$	$1.48\ 10^{-1}$
$k_{16\ 20}$	$2.00\ 10^{0}$
$k_{20\ 16}$	$2.46\ 10^{-2}$

Element independent transfer coefficients (sheep)

Parameter	Value, d^{-1}
$k_{14\ 1}$	$1.11\ 10^{0}$
$k_{14\ 15}$	$5.53\ 10^{-4}$
$k_{15\ 1}$	$1.00\ 10^{-2}$
$k_{15\ 16}$	$2.00\ 10^{0}$
$k_{15\ 17}$	$2.00\ 10^{0}$
$k_{15\ 18}$	$2.00\ 10^{0}$
$k_{16\ 15}$	$4.76\ 10^{-1}$
$k_{17\ 15}$	$1.48\ 10^{-1}$
$k_{18\ 15}$	$2.46\ 10^{-2}$
$K_{20\ 14}$	$1.04\ 10^{-1}$
$k_{20\ 15}$	$2.07\ 10^{-2}$

B8.1 Running of the model for routine release applications

The model is run in the same way as for cows and sheep for other elements as described in Sections B6.5 and B7.5.

B8.2 Calculation of activity concentrations in animal tissues and milk

(a) For cows or sheep on pasture and a continuous deposition rate (Bq km^{-2} s^{-1})

Concentration in meat (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

Concentration $= I_{18}/N \times M$ for cows

$= I_{16}/N \times M$ for sheep

Where I_{18} and I_{16} are the inventory (or integral) of activity in compartments 18 and 16 which represent meat

N is the number of cows or sheep per unit area (km^{-2})	(see Table 3.20)
M is the mass of meat per animal (kg)	(see Table 3.20)

Concentration in liver (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

Concentration $= I_{19}/ N \times L$ for cows

$= I_{17}/ N \times L$ for sheep

Where I_{19} and I_{17} are the inventory (or integral) of activity in compartments 19 and 17 which represent liver

L is the mass of liver per animal (kg)	(see Table 3.20)

Concentration in milk (or integral of concentration)
(Bq kg^{-1} (or Bq y kg^{-1}))

Concentration $= I_{17}/N_c \times Y_m$

Where I_{17} is the inventory (or integral) of activity in compartment 17 representing milk

N_c is the number of cows per unit area (km^{-2})	(see Table 3.20)
Y_m is the yield of milk per cow (l d^{-1})	(see Table 3.20)

(b) For cows or sheep inhaling

Inhalation of the radioactive plume is not modelled in PC-CREAM 08.

APPENDIX C Some Major European Rivers

C1 EUROPEAN RIVERS

This appendix outlines the main features of four major European rivers for which a model implementation is available in PC-CREAM 08. Further details for the Loire, Rhône, Rhine and Meuse can be found in Simmonds, Lawson and Mayall (1995). The rivers considered here are the Rhine in Germany, Rhône and Loire in France and the Thames in the UK. A number of tributaries from some of these rivers are also included in the models provided with PC-CREAM 08. The tributaries of the Rhine which are considered are the Neckar, Main, Moselle and Lippe. The Vienne which is a tributary of the Loire is also included. Figure C4.1 shows the river networks of the Rhine, Loire and Rhone considered in PC-CREAM 08. Tables C4.1 to C4.4 give the parameter values used in the dynamic river model in PC-CREAM 08 and the locations of relevant nuclear sites (Harvey et al, 2008; Simmonds et al, 1995; Smith et al, 2002; Mobbs et al, 1990; Hilton et al, 2002).

The ingestion rates of freshwater fish vary for each European country. Suggested values are given in Table C4.5 (Jones et al, 2006). The default values in PC-CREAM 08 are considered appropriate for the UK but these can be changed by the user.

C2 REFERENCES

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C3 TABLES

Table C3.1 – Parameters used in PC-CREAM 08 for the River Rhine and tributaries with discharging sites (Smith et al, 2002)

River section	Flow ($m^3\ s^{-1}$)**	Width (m)	Depth (m)	Length (m)	Volume (m^3)	Sediment flow **($m^3\ s^{-1}$)	Suspended sediment load t m^{-3}	Discharging sites
1	6.96 10^{2}	1.05 10^{2}	3.90 10^{0}	1.20 10^{5}	4.91 10^{7}	5.36 10^{-3}	3.70 10^{-5}	Fessenheim
2	1.24 10^{3}	1.55 10^{2}	4.70 10^{0}	1.25 10^{5}	9.11 10^{7}	8.50 10^{-3}	3.80 10^{-5}	Karlsruhe Wak, Phillipsberg 1, Phillipsberg 2
4	1.39 10^{3}	1.77 10^{2}	4.90 10^{0}	7.50 10^{4}	6.50 10^{7}	8.95 10^{-3}	4.10 10^{-5}	Biblis A, Biblis B
6	1.58 10^{3}	1.99 10^{2}	5.30 10^{0}	9.50 10^{4}	1.00 10^{8}	7.76 10^{-3}	4.50 10^{-5}	
8	1.97 10^{3}	3.98 10^{2}	7.6 10^{0}	1.75 10^{5}	5.29 10^{8}	9.77 10^{-3}	4.50 10^{-5}	Mulheim-Karlich
10	2.21 10^{3}	3.25 10^{2}	6.8 10^{0}	2.40 10^{5}	5.30 10^{8}	5.36 10^{-3}	3.70 10^{-5}	Dodewaard
*Neckar (3)	9.86 10^{1}	6.06 10^{1}	2.90 10^{0}	1.65 10^{5}	2.90 10^{7}	1.02 10^{-3}	3.00 10^{-5}	Obringheim, Neckar 1, Neckar 2
* Main (5)	1.45 10^{2}	5.57 10^{1}	2.60 10^{0}	3.80 10^{5}	5.50 10^{7}	1.67 10^{-3}	3.50 10^{-5}	Grafenrheinfeld
* Moselle (7)	2.11 10^{2}	1.00 10^{2}	3.75 10^{0}	2.50 10^{5}	9.38 10^{7}	1.68 10^{-3}	3.00 10^{-5}	Cattenom
* Leppe (9)	3.40 10^{1}	3.00 10^{1}	1.90 10^{0}	1.50 10^{5}	8.55 10^{6}	5.36 10^{-4}	4.00 10^{-5}	Hamm-Uentrop – THTR-300 (closed)

* Neckar (3) flows into section 4 of the Rhine
* Main (5) flows into section 6 of the Rhine
* Moselle (7) flows into section 8 of the Rhine
* Leppe (9) flows into section 10 of the Rhine

** Flow is calculated from the velocity x depth of water or sediment x width of river. The sediment velocity is assumed to be 1 10^{-4} of the river water velocity.

Table C3.2 – Parameters used in PC-CREAM 08 for the River Rhone with discharging sites (Smith et al, 2002)

River section	Flow ($m^3\ s^{-1}$)*	Width (m)	Depth (m)	Length (m)	Volume (m^3)	Sediment flow *($m^3\ s^{-1}$)	Suspended sediment load t m^{-3}	Discharging sites
1	4.45 10^{2}	2.02 10^{2}	2.00 10^{0}	1.45 10^{5}	5.9 10^{7}	5.64 10^{-3}	2.50 10^{-5}	Creys Malville, Bugey B
2	5.96 10^{2}	1.80 10^{2}	3.30 10^{0}	2.00 10^{4}	1.2 10^{7}	6.50 10^{-3}	2.40 10^{-5}	
3	1.02 10^{3}	2.21 10^{2}	4.70 10^{0}	4.00 10^{4}	4.2 10^{7}	5.45 10^{-3}	2.70 10^{-5}	
4	1.02 10^{3}	2.22 10^{2}	5.00 10^{0}	6.00 10^{4}	6.7 10^{7}	7.32 10^{-3}	3.10 10^{-5}	St Alban
5	1.50 10^{3}	2.00 10^{2}	8.50 10^{0}	4.50 10^{4}	7.6 10^{7}	8.99 10^{-3}	3.50 10^{-5}	Cruas
6	1.50 10^{3}	2.00 10^{2}	9.00 10^{0}	4.00 10^{4}	7.2 10^{7}	1.02 10^{-2}	3.70 10^{-5}	Tricastin
7	1.54 10^{3}	2.11 10^{2}	9.20 10^{0}	5.50 10^{4}	1.1 10^{8}	1.02 10^{-2}	4.00 10^{-5}	Marcoule
8	1.64 10^{3}	2.20 10^{2}	9.90 10^{0}	3.50 10^{4}	7.6 10^{7}	8.59 10^{-3}	4.50 10^{-5}	
9	1.68 10^{3}	2.29 10^{2}	1.05 10^{1}	5.00 10^{4}	1.2 10^{8}	6.31 10^{-3}	5.00 10^{-5}	

* Flow is calculated from the velocity x depth of water or sediment x width of river. The sediment velocity is taken from Simmonds et al (1995).

Table C3.3 – Parameters used in PC-CREAM 08 for the River Loire and tributaries with discharging sites (Smith et al, 2002).

River section	Flow ($m^3\ s^{-1}$)**	Width (m)	Depth (m)	Length (m)	Volume (m^3)	Sediment flow **($m^3\ s^{-1}$)	Suspended sediment load t m^{-3}	Discharging sites
1	$3.98\ 10^{2}$	$2.04\ 10^{2}$	$1.50\ 10^{0}$	$1.70\ 10^{5}$	$5.2\ 10^{7}$	$6.73\ 10^{-3}$	$2.00\ 10^{-5}$	Belleville, Dampierre
2	$5.30\ 10^{2}$	$2.54\ 10^{2}$	$1.90\ 10^{0}$	$1.10\ 10^{5}$	$5.3\ 10^{7}$	$7.24\ 10^{-3}$	$3.00\ 10^{-5}$	St Luarent B
3	$9.03\ 10^{2}$	$2.83\ 10^{2}$	$4.00\ 10^{0}$	$1.15\ 10^{5}$	$1.3\ 10^{8}$	$5.38\ 10^{-3}$	$4.00\ 10^{-5}$	Chinon B
4	$1.26\ 10^{3}$	$3.05\ 10^{2}$	$5.90\ 10^{0}$	$1.00\ 10^{5}$	$1.8\ 10^{8}$	$2.90\ 10^{-3}$	$5.00\ 10^{-5}$	
Vienne (3a)*	$2.54\ 10^{1}$	$1.07\ 10^{2}$	2.5010^{0}	$6.00\ 10^{4}$	$1.60\ 10^{7}$	$1.63\ 10^{-4}$	$3.00\ 10^{-5}$	Civaux

* Vienne (3a) flows into section 3 of the Loire. Data for the Vienne are taken from Mobbs et al (1990).

* *Flow is calculated from the velocity x depth of water or sediment x width of river. The sediment velocity is taken from Simmonds et al (1995).

Table C3.4 – Parameters used in PC-CREAM 08 for the River Thames and tributaries with discharging sites, (Hilton et al, 2002).

River section		Flow ($m^3\ s^{-1}$)*	Width (m)	Depth (m)	Length (m)	Volume (m^3)	Sediment flow *($m^3\ s^{-1}$)	Suspended sediment load t m^{-3}	Discharging sites
1	Ray confluence to Cherwell confluence	$1.35\ 10^{1}$	$2.42\ 10^{1}$	$1.00\ 10^{0}$	$6.96\ 10^{4}$	$1.69\ 10^{6}$	$4.04\ 10^{-4}$	$2.50\ 10^{-5}$	
2	Cherwell confluence to Sutton Courtenay	$2.57\ 10^{1}$	$5.00\ 10^{1}$	$2.23\ 10^{0}$	$1.75\ 10^{4}$	$1.95\ 10^{6}$	$3.45\ 10^{-4}$	$2.50\ 10^{-5}$	
3	Sutton Courtenay to 1 km below	$2.57\ 10^{1}$	$4.90\ 10^{1}$	$2.15\ 10^{0}$	$1.00\ 10^{3}$	$1.05\ 10^{5}$	$3.58\ 10^{-4}$	$2.50\ 10^{-5}$	Harwell
4	1 km below Sutton Courtenay confluence to Kennet confluence	$3.54\ 10^{1}$	$5.45\ 10^{1}$	$2.27\ 10^{0}$	$4.85\ 10^{4}$	$6.00\ 10^{6}$	$4.67\ 10^{-4}$	$2.50\ 10^{-5}$	Aldermaston
5	Kennet confluence to Loddon confluence	$5.21\ 10^{1}$	$6.00\ 10^{1}$	$2.04\ 10^{0}$	$8.00\ 10^{3}$	$9.79\ 10^{5}$	$7.67\ 10^{-4}$	$2.50\ 10^{-5}$	Aldermaston **
6	Loddon confluence to Colne confluence	$5.67\ 10^{1}$	$5.00\ 10^{1}$	$1.79\ 10^{0}$	$5.80\ 10^{4}$	$5.19\ 10^{6}$	$9.50\ 10^{-4}$	$2.50\ 10^{-5}$	
7	Colne confluence to Wey confluence	$5.41\ 10^{1}$	$5.05\ 10^{1}$	$2.14\ 10^{0}$	$1.05\ 10^{4}$	$1.13\ 10^{6}$	$7.59\ 10^{-4}$	$2.50\ 10^{-5}$	Amersham ***
8	Wey confluence to Teddington Lock	$7.78\ 10^{1}$	$7.80\ 10^{1}$	$2.97\ 10^{0}$	$2.05\ 10^{4}$	$4.75\ 10^{6}$	$7.85\ 10^{-4}$	$2.50\ 10^{-5}$	

* Flow is calculated from the velocity x depth of water or sediment x width of river. The sediment velocity is $1\ 10^{-4}$ of water velocity.

** Aldermaston discharges into the River Kennet

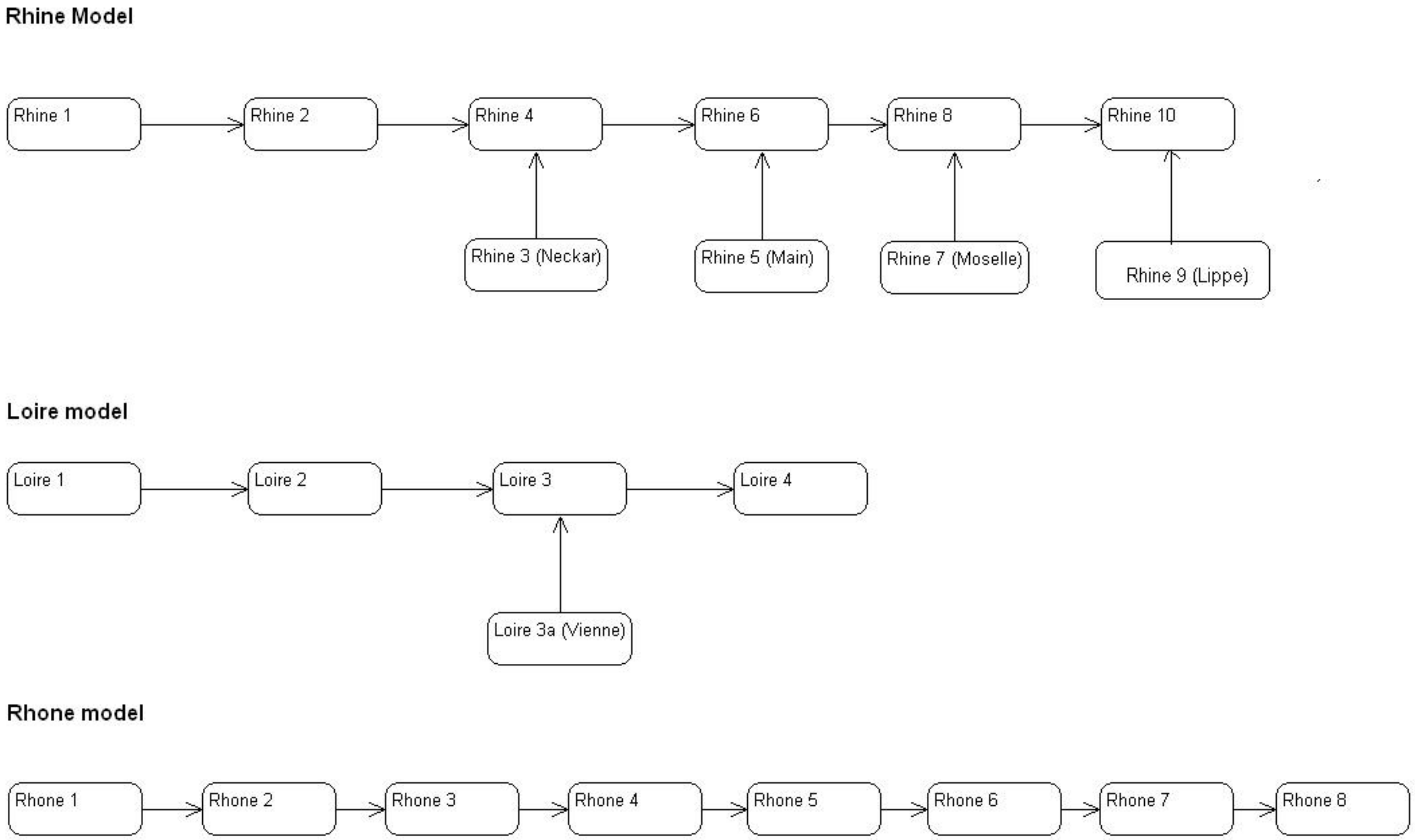
*** Amersham discharges into the River Colne.

Table C3.5 Suggested adult freshwater fish ingestion rates for use in the calculation of representative critical group doses arising from aquatic discharges (Jones et al, 2006)

Country	Ingestion rate (kg y^{-1})
Belgium	22
Finland	76
France	33
Germany	22
Portugal	4
Spain	18
Sweden	39
The Netherlands	25
United Kingdom	23

C4 FIGURE

Figure C4.1 Layout of rivers Rhine, Loire and Rhone as modelled in PC-CREAM 08



APPENDIX D Modelling in the Mediterranean Sea

D1 MEDITERRANEAN SEA

A model for the dispersion of radionuclides in the Mediterranean Sea has been developed by the Commissariat à l'Energie Atomique (CEA), France (Cigna et al, 1994). Details of the original model are given in Simmonds, Lawson and Mayall (1995). This model was modified for implementation in PC-CREAM 08 to reduce the number of compartments so that the code could be run efficiently. A brief description of the model as implemented in PC-CREAM 08 is given here. The Mediterranean Sea communicates with the Atlantic Ocean through the Straits of Gibraltar with a depth of about 300 m and the Black Sea communicates with the Mediterranean Sea through the 40 m deep Bosphorus and the 79 m deep Dardanelles. The mean depth of the Mediterranean Sea is about 1400 m. The Adventure Bank, south of Sicily, at a depth of less than 400 m, separates the western and the eastern basins of the Mediterranean Sea.

The Mediterranean Sea model consists of eleven surface water compartments, the geographical extent of these is shown in Figure D4.1. The number of model compartments has been reduced from that described in Simmonds et al (1995) from 21 to 16, by including the benthic boundary layer compartment into the water column above for five sea areas: Alboran Sea, Gulf of Lions, Western Basin, Eastern Basin and the Black Sea. Sedimentation occurs over the total water column for these sea areas and not just the benthic boundary layer as described in Simmonds et al (1995). The remaining oceans and seas of the world are described using three compartments, one for the Gulf of Cadiz which links the Mediterranean with the Atlantic Ocean, one for Northern Europe and one for the remainder of the world's oceans. Note that the Northern European compartment includes all oceans and seas in Northern Europe except for the Gulf of Cadiz and the Mediterranean. It is therefore larger than the Northern European waters compartment shown in Figure 4.4 of section 4 of the main text. Volumes and depths of all the compartments of the Mediterranean Sea model are given in Table D4.1. The exchanges between the compartments have been derived according to the methodology used in the REJMAR model (Chartier, 1987), as described in Simmonds et al (1995). The exchange rates for the model are given in Table D4.2

The sedimentation model adopted for the Mediterranean Sea is the same as the model used for North European waters (see section 4.4 of the main text). Parameter values used in the model are similar to those used for the Northern European waters, except that the value for the sedimentation rate and suspended sediment load is the same for all compartments with a sediment layer. Also the sediment porosity and sediment diffusion rate have two values for waters with depths greater or less than 200 m. The values are given in Table D4.3. These values have been adopted for all the compartments with the exception of the three compartments outside the Mediterranean

Sea for which the same values adopted in the regional marine model for North European waters have been used.

Total catches of fish, crustacea and molluscs in each compartment were calculated by MARINA-MED Working Group III (Cigna et al, 1994). Seafood catches taken from the United Nations Fisheries and Agricultural Organisation (FAO, 1985; 1990; 1991) and from national sources were used to allocate catches of the model compartments to EC member states. Information on import and export of EC member states was finally used to determine the quantities of seafood consumed by each EC member state which originated in the Mediterranean Sea. The final distribution is given in Tables D4.4 to D4.6. These data include the fraction lost due to processing given in Table 4.16 of the main text. The beach occupancy assumed for the Mediterranean is given in Table 4.18 of the main text. This is used together with the coastline length given in Table D4.7 to obtain the collective beach occupancy discussed in section 4.4.2.2 of the main text.

D2 REFERENCES

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D3 TABLES

Table D3.1 - Parameter values for the Mediterranean marine model compartments as implemented in PC-CREAM 08

Compartment number	Compartment name	Volume (m^3)	Depth (m)
1	Other oceans	$8.98\ 10^{17}$	$3.80\ 10^{3}$
2	Northern Europe	$4.65\ 10^{17}$	$2.94\ 10^{3}$
3	Gulf Of Cadiz	$2.30\ 10^{14}$	$1.70\ 10^{3}$
4	Alboran Sea (surface)	$5.00\ 10^{12}$	$1.00\ 10^{2}$
5	Alboran Sea (deep)	$2.43\ 10^{13}$	$4.85\ 10^{2}$
6	Liguro - Provencal Basin	$2.81\ 10^{13}$	$1.00\ 10^{2}$
7	Algerian Basin	$2.69\ 10^{13}$	$1.00\ 10^{2}$
8	Tyrrhenian Sea	$2.40\ 10^{13}$	$1.00\ 10^{2}$
9	Gulf of Lions	$5.36\ 10^{11}$	$8.00\ 10^{1}$
10	Western Basin	$1.11\ 10^{15}$	$1.41\ 10^{3}$
11	Adriatic Sea	$1.38\ 10^{13}$	$1.00\ 10^{2}$
12	Ionian Sea	$2.64\ 10^{13}$	$1.00\ 10^{2}$
13	Libyan Sea	$6.34\ 10^{13}$	$1.00\ 10^{2}$
14	Aegean Sea	$1.70\ 10^{13}$	$1.00\ 10^{2}$
15	Levantin Basin	$3.43\ 10^{13}$	$1.00\ 10^{2}$
16	Cyprus Sea	$1.11\ 10^{13}$	$1.00\ 10^{2}$
17	Eastern Basin	$2.20\ 10^{15}$	$1.33\ 10^{3}$
18	Black Sea (surface waters)	$4.53\ 10^{13}$	$1.00\ 10^{2}$
19	Black Sea (deep waters)	$4.98\ 10^{14}$	$1.10\ 10^{3}$

Table D3.2 - Exchange rates for Mediterranean regional marine model as implemented in PC-CREAM 08

From compartment no	Compartment name	To compartment no	Compartment name	Exchange rate (m^3 y^{-1})
11	Adriatic Sea	17	Eastern Basin	2.73 10^{13}
14	Aegean sea	17	Eastern Basin	5.36 10^{12}
14	Aegean sea	12	Ionian sea	4.70 10^{13}
14	Aegean sea	18	Black sea (surface)	1.89 10^{11}
4	Alboran sea surface	5	Alboran sea deep	1.58 10^{12}
4	Alboran sea surface	7	Algerian Basin	5.19 10^{13}
5	Alboran sea deep	4	Alboran sea surface	1.58 10^{12}
5	Alboran sea deep	3	Gulf of Cadiz	5.06 10^{13}
7	Algerian Basin	6	Liguro - Provencal Basin	1.73 10^{13}
7	Algerian Basin	8	Tyrrhenian sea	5.82 10^{13}
7	Algerian Basin	10	Western Basin	8.47 10^{12}
18	Black sea (surface)	14	Aegian sea	4.06 10^{11}
18	Black sea (surface)	19	Black sea (deep)	1.13 10^{9}
19	Black sea (deep)	18	Black sea (surface)	1.13 10^{9}
16	Cyprus sea	14	Aegian sea	2.52 10^{13}
16	Cyprus sea	17	Eastern Basin	1.87 10^{13}
17	Eastern Basin	11	Adriatic Sea	4.35 10^{12}
17	Eastern Basin	14	Aegian sea	5.36 10^{12}
17	Eastern Basin	16	Cyprus sea	3.50 10^{12}
17	Eastern Basin	12	Ionian sea	8.32 10^{12}
17	Eastern Basin	15	Levantin Basin	1.08 10^{13}
17	Eastern Basin	13	Libyan sea	2.00 10^{13}
17	Eastern Basin	10	Western Basin	3.80 10^{13}
3	Gulf of Cadiz	4	Alboran sea surface	5.29 10^{13}
3	Gulf of Cadiz	2	Northern Europe	5.68 10^{14}
9	Gulf of Lions	6	Liguro - Provencal Basin	1.58 10^{13}
12	Ionian sea	11	Adriatic Sea	2.69 10^{13}
12	Ionian sea	17	Eastern Basin	8.32 10^{12}
12	Ionian sea	13	Libyan sea	2.41 10^{13}
15	Levantin Basin	16	Cyprus sea	2.52 10^{13}
15	Levantin Basin	17	Eastern Basin	1.08 10^{13}
13	Libyan sea	17	Eastern Basin	2.00 10^{13}
13	Libyan sea	12	Ionian sea	3.73 10^{13}
13	Libyan sea	15	Levantin Basin	2.56 10^{13}
6	Liguro - Provencal Basin	7	Algerian Basin	2.42 10^{13}
6	Liguro - Provencal Basin	9	Gulf of Lions	1.58 10^{13}
6	Liguro - Provencal Basin	10	Western Basin	2.24 10^{13}
2	Northern Europe	3	Gulf of Cadiz	5.70 10^{14}
2	Northern Europe	1	Other Oceans	1.09 10^{15}
1	Other Oceans	2	Northern Europe	6.94 10^{14}
8	Tyrrhenian sea	13	Libyan sea	3.98 10^{13}

From compartment no	Compartment name	To compartment no	Compartment name	Exchange rate (m^3 y^{-1})
8	Tyrrhenian sea	6	Liguro - Provencal Basin	2.05 10^{13}
8	Tyrrhenian sea	10	Western Basin	7.56 10^{12}
10	Western Basin	5	Alboran sea deep	4.93 10^{13}
10	Western Basin	7	Algerian Basin	8.47 10^{12}
10	Western Basin	6	Liguro - Provencal Basin	8.85 10^{12}
10	Western Basin	8	Tyrrhenian sea	7.56 10^{12}

Table D3.3 - Sediment model parameters common to all compartments used in the Mediterranean Sea model as implemented in PC-CREAM 08 (Simmonds et al, 2002)*

Parameter	Default value	Description
Lt	0.1 m	Thickness of top sediment layer
Lm	1.9 m	Thickness of middle sediment layer
P	2.6 t m^{-3}	Sediment mineral density
RW	5 10^{-3} m y^{-1}	Sediment reworking rate for shallow seas up to 200m, also local compartment.
	5 10^{-4} m y^{-1}	Sediment reworking rate for deep seas greater than 200m.
RT	1 y^{-1}	Pore water turn over rate for shallow seas up to 200m, also local compartment.
	0.1 y^{-1}	Pore water turn over rate for deep seas greater than 200m.
D	3.15 10^{-2} m^2 y^{-1}	Sediment diffusion rate for shallow seas up to 200m, also local compartment.
	3.15 10^{-3} m^2 y^{-1}	Sediment diffusion rate for deep seas greater than 200m.
ε	0.75	Sediment porosity for shallow seas up to 200m, also local compartment.
	0.3	Sediment porosity for deep seas greater than 200m.
SR	7.5 10^{-5} t m^{-2} y^{-1}	Sedimentation rate for all Mediterranean compartments.

* Data for three compartments connected to Mediterranean are the same as for Gulf of Cadiz, Atlantic NE and Other Oceans in Table 4.11 of the main text.

TABLE D3.4 Annual fish consumption by Mediterranean Sea compartments used in PC-CREAM 08

Nation	Annual marine fish consumption t y^{-1}											
	Adriatic Sea	Aegean Sea	Alboran Sea (surface)	Algerian Basin	Black Sea (surface)	Black Sea (deep)	Cyprus Sea	Gulf of Cadiz	Gulf of Lions	Ionian Sea	Levantin Basin	Libyan Sea
Austria								1.5 10^{2}				
Belgium	2.9 10^{2}	1.5 10^{1}	1.3 10^{1}	5.0 10^{-1}	3.1 10^{0}	3.4 10^{1}	1.0 10^{0}	1.5 10^{2}	1.7 10^{2}	1.7 10^{2}	1.0 10^{1}	5.5 10^{0}
Denmark	1.3 10^{2}	1.8 10^{2}	5.0 10^{0}	4.0 10^{0}	1.4 10^{1}	1.5 10^{2}	1.9 10^{1}	5.0 10^{1}	2.1 10^{1}	1.2 10^{2}	6.0 10^{0}	0.0 10^{0}
Finland								1.5 10^{2}				
France	2.0 10^{3}	2.6 10^{2}	6.5 10^{2}	9.5 10^{1}	1.1 10^{2}	1.2 10^{3}	6.0 10^{1}	1.6 10^{3}	1.9 10^{3}	1.6 10^{3}	4.0 10^{1}	1.1 10^{3}
Germany	1.7 10^{3}	6.0 10^{2}	8.0 10^{1}	2.6 10^{1}	5.8 10^{1}	6.4 10^{2}	7.5 10^{1}	2.5 10^{2}	5.5 10^{2}	1.2 10^{3}	5.5 10^{1}	5.5 10^{1}
Greece	1.2 10^{3}	2.7 10^{4}	1.9 10^{2}	7.0 10^{0}	8.1 10^{1}	8.9 10^{2}	3.3 10^{3}	1.5 10^{2}	3.0 10^{1}	2.6 10^{3}	1.4 10^{2}	8.0 10^{1}
Ireland	5.0 10^{-1}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	0.0 10^{0}	1.0 10^{0}	5.0 10^{-1}	1.0 10^{0}	0.0 10^{0}
Italy	3.9 10^{4}	9.5 10^{3}	1.7 10^{3}	6.0 10^{2}	2.5 10^{2}	2.7 10^{3}	1.0 10^{3}	2.1 10^{3}	2.1 10^{3}	3.0 10^{4}	5.0 10^{2}	4.1 10^{3}
Netherlands	1.1 10^{3}	4.2 10^{2}	2.5 10^{1}	6.0 10^{2}	3.0 10^{1}	3.3 10^{2}	4.8 10^{1}	1.0 10^{2}	7.9 10^{1}	9.5 10^{2}	4.5 10^{1}	2.0 10^{0}
Norway												
Portugal	3.8 10^{2}	1.7 10^{1}	5.5 10^{2}	3.8 10^{2}	2.1 10^{0}	2.3 10^{1}	1.5 10^{0}	3.7 10^{4}	4.4 10^{2}	8.5 10^{1}	1.5 10^{0}	0.0 10^{0}
Spain	8.5 10^{2}	4.5 10^{2}	8.5 10^{3}	6.0 10^{3}	7.1 10^{0}	7.8 10^{1}	4.3 10^{1}	2.0 10^{4}	6.5 10^{3}	7.5 10^{2}	2.0 10^{1}	2.6 10^{2}
Sweden								5.0 10^{1}				
United Kingdom	2.3 10^{2}	1.3 10^{3}	4.2 10^{1}	1.8 10^{1}	9.0 10^{0}	9.9 10^{1}	1.3 10^{2}	5.0 10^{2}	7.0 10^{1}	2.0 10^{2}	1.8 10^{2}	2.8 10^{1}
World	4.6 10^{4}	4.0 10^{4}	1.2 10^{4}	7.7 10^{3}	5.6 10^{2}	6.2 10^{3}	4.7 10^{3}	6.9 10^{4}	1.2 10^{4}	3.8 10^{4}	1.0 10^{3}	5.6 10^{3}

* Zero catch for Alboran Sea (deep), Eastern Basin, Western Basin, Other Oceans.

TABLE D3.4 continued Annual fish consumption by Mediterranean Sea compartments used in PC-CREAM 08

Nation	Annual marine fish consumption t y^{-1}		
	Liguro - Provençal Basin	Northern Europe	Tyrrhenian Sea
Austria		$3.6\ 10^{3}$	
Belgium	$7.0\ 10^{2}$	$2.8\ 10^{4}$	$1.2\ 10^{2}$
Denmark	$9.0\ 10^{1}$	$1.5\ 10^{5}$	$7.0\ 10^{1}$
Finland		$3.8\ 10^{4}$	
France	$8.0\ 10^{3}$	$1.8\ 10^{5}$	$1.3\ 10^{3}$
Germany	$1.1\ 10^{3}$	$1.7\ 10^{5}$	$7.5\ 10^{2}$
Greece	$1.5\ 10^{2}$	$7.2\ 10^{3}$	$4.2\ 10^{2}$
Ireland	$3.5\ 10^{0}$	$3.4\ 10^{4}$	$5.0\ 10^{-1}$
Italy	$9.5\ 10^{3}$	$4.3\ 10^{4}$	$1.9\ 10^{4}$
Netherlands	$5.0\ 10^{2}$	$7.3\ 10^{4}$	$1.0\ 10^{2}$
Norway		$4.7\ 10^{4}$	
Portugal	$1.7\ 10^{3}$	$7.6\ 10^{4}$	$5.5\ 10^{1}$
Spain	$2.6\ 10^{4}$	$1.9\ 10^{5}$	$5.0\ 10^{2}$
Sweden		$1.0\ 10^{5}$	
United Kingdom	$2.9\ 10^{2}$	$3.3\ 10^{5}$	$7.5\ 10^{1}$

TABLE D3.5 Annual crustacean consumption by Mediterranean Sea compartments used in PC-CREAM 08

Nation	Annual marine crustacean consumption t y^{-1}										
	Adriatic Sea	Aegean Sea	Alboran Sea (surface)	Algerian Basin	Black Sea (surface)	Cyprus Sea	Gulf of Cadiz	Gulf of Lions	Ionian Sea	Levantin Basin	Libyan Sea
Austria											
Belgium	$3.5\ 10^{-1}$	$4.9\ 10^{0}$	$7.0\ 10^{-1}$	$3.5\ 10^{-1}$	$1.1\ 10^{1}$	$3.2\ 10^{0}$	$0.0\ 10^{0}$	$1.4\ 10^{0}$	$7.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Denmark	$0.0\ 10^{0}$	$3.5\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$2.1\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$3.5\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Finland											
France	$1.9\ 10^{1}$	$1.6\ 10^{1}$	$6.0\ 10^{0}$	$3.9\ 10^{0}$	$5.6\ 10^{1}$	$1.8\ 10^{1}$	$1.7\ 10^{1}$	$1.8\ 10^{1}$	$4.2\ 10^{1}$	$1.1\ 10^{0}$	$4.6\ 10^{1}$
Germany	$2.5\ 10^{0}$	$2.5\ 10^{0}$	$3.5\ 10^{-1}$	$0.0\ 10^{0}$	$1.1\ 10^{1}$	$3.2\ 10^{0}$	$0.0\ 10^{0}$	$1.4\ 10^{1}$	$5.6\ 10^{0}$	$2.5\ 10^{0}$	$1.8\ 10^{0}$
Greece	$5.3\ 10^{0}$	$1.1\ 10^{3}$	$1.8\ 10^{0}$	$1.4\ 10^{0}$	$0.0\ 10^{0}$	$3.5\ 10^{-1}$	$0.0\ 10^{0}$	$1.8\ 10^{0}$	$2.3\ 10^{1}$	$7.0\ 10^{-1}$	$0.0\ 10^{0}$
Ireland	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Italy	$2.5\ 10^{3}$	$5.6\ 10^{1}$	$7.0\ 10^{0}$	$4.9\ 10^{0}$	$6.3\ 10^{1}$	$1.9\ 10^{1}$	$1.7\ 10^{1}$	$1.2\ 10^{2}$	$5.6\ 10^{3}$	$2.1\ 10^{0}$	$4.6\ 10^{2}$
Netherlands	$1.1\ 10^{0}$	$7.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$4.9\ 10^{0}$	$1.4\ 10^{0}$	$0.0\ 10^{0}$	$3.5\ 10^{-1}$	$2.5\ 10^{0}$	$0.0\ 10^{0}$	$4.2\ 10^{0}$
Norway											
Portugal	$7.0\ 10^{-1}$	$2.3\ 10^{1}$	$1.1\ 10^{1}$	$7.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$2.3\ 10^{2}$	$1.1\ 10^{1}$	$1.8\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Spain	$2.8\ 10^{1}$	$3.3\ 10^{1}$	$4.2\ 10^{2}$	$2.6\ 10^{2}$	$8.8\ 10^{1}$	$2.6\ 10^{1}$	$1.1\ 10^{3}$	$2.8\ 10^{2}$	$6.7\ 10^{1}$	$0.0\ 10^{0}$	$5.3\ 10^{2}$
Sweden											
United Kingdom	$1.4\ 10^{0}$	$2.8\ 10^{0}$	$1.4\ 10^{0}$	$7.0\ 10^{-1}$	$1.5\ 10^{1}$	$4.6\ 10^{0}$	$0.0\ 10^{0}$	$1.8\ 10^{0}$	$3.5\ 10^{0}$	$1.4\ 10^{0}$	$3.5\ 10^{-1}$
World	$2.5\ 10^{3}$	$1.2\ 10^{3}$	$4.5\ 10^{2}$	$2.7\ 10^{2}$	$2.5\ 10^{2}$	$7.6\ 10^{1}$	$1.4\ 10^{3}$	$4.4\ 10^{2}$	$5.7\ 10^{3}$	$7.7\ 10^{0}$	$1.0\ 10^{3}$

* Zero catch for Alboran Sea (deep), Eastern Basin, Western Basin, Black Sea (deep), Other Oceans.

TABLE D3.5 continued Annual crustacean consumption by Mediterranean Sea compartments used in PC-CREAM 08

Nation	Annual marine crustacean consumption t y^{-1}		
	Liguro - Provençal Basin	Northern Europe	Tyrrhenian Sea
Austria		$1.5\ 10^{4}$	
Belgium	$4.9\ 10^{0}$	$1.2\ 10^{5}$	$7.0\ 10^{-1}$
Denmark	$3.5\ 10^{-1}$	$6.3\ 10^{5}$	$0.0\ 10^{0}$
Finland		$1.5\ 10^{5}$	
France	$7.7\ 10^{1}$	$7.6\ 10^{5}$	$2.6\ 10^{1}$
Germany	$6.0\ 10^{0}$	$7.2\ 10^{5}$	$2.8\ 10^{0}$
Greece	$6.3\ 10^{0}$	$2.9\ 10^{4}$	$4.6\ 10^{0}$
Ireland	$0.0\ 10^{0}$	$1.4\ 10^{5}$	$0.0\ 10^{0}$
Italy	$4.6\ 10^{2}$	$1.8\ 10^{5}$	$2.0\ 10^{3}$
Netherlands	$1.8\ 10^{0}$	$3.1\ 10^{5}$	$1.1\ 10^{0}$
Norway		$2.0\ 10^{5}$	
Portugal	$4.2\ 10^{1}$	$3.1\ 10^{5}$	$2.5\ 10^{0}$
Spain	$1.1\ 10^{3}$	$7.7\ 10^{5}$	$4.2\ 10^{1}$
Sweden		$4.1\ 10^{5}$	
United Kingdom	$7.7\ 10^{0}$	$1.3\ 10^{6}$	$2.1\ 10^{0}$

TABLE D3.6 Annual mollusc consumption by Mediterranean Sea compartments used in PC-CREAM 08

	Annual marine mollusc consumption t y^{-1}										
Nation	Adriatic Sea	Aegean Sea	Alboran Sea (surface)	Algerian Basin	Black Sea (surface)	Cyprus Sea	Gulf of Cadiz	Gulf of Lions	Ionian Sea	Levantin Basin	Libyan Sea
Austria											
Belgium	$1.2\ 10^{0}$	$1.8\ 10^{0}$	$3.0\ 10^{-1}$	$1.5\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$5.0\ 10^{0}$	$4.1\ 10^{0}$	$3.0\ 10^{-1}$	$0.0\ 10^{0}$	$9.8\ 10^{0}$
Denmark	$6.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.7\ 10^{0}$	$1.5\ 10^{-1}$	$1.5\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
Finland											
France	$1.5\ 10^{2}$	$1.8\ 10^{2}$	$1.2\ 10^{1}$	$9.8\ 10^{0}$	$1.0\ 10^{2}$	$2.3\ 10^{1}$	$8.3\ 10^{1}$	$4.8\ 10^{2}$	$3.8\ 10^{1}$	$3.2\ 10^{0}$	$1.3\ 10^{2}$
Germany	$8.7\ 10^{1}$	$4.8\ 10^{1}$	$2.3\ 10^{0}$	$1.5\ 10^{0}$	$6.8\ 10^{1}$	$1.4\ 10^{1}$	$1.7\ 10^{1}$	$6.3\ 10^{0}$	$2.1\ 10^{1}$	$0.0\ 10^{0}$	$1.7\ 10^{1}$
Greece	$2.1\ 10^{2}$	$9.8\ 10^{2}$	$1.1\ 10^{0}$	$4.5\ 10^{-1}$	$6.0\ 10^{1}$	$1.4\ 10^{1}$	$1.7\ 10^{1}$	$2.9\ 10^{0}$	$3.3\ 10^{1}$	$2.3\ 10^{0}$	$1.1\ 10^{2}$
Ireland	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$	$1.5\ 10^{-1}$
Italy	$1.9\ 10^{4}$	$5.0\ 10^{2}$	$2.9\ 10^{1}$	$3.2\ 10^{1}$	$4.9\ 10^{2}$	$1.0\ 10^{2}$	$1.7\ 10^{2}$	$3.3\ 10^{2}$	$4.8\ 10^{3}$	$1.2\ 10^{0}$	$1.1\ 10^{0}$
Netherlands	$3.3\ 10^{0}$	$1.3\ 10^{1}$	$4.5\ 10^{-1}$	$3.0\ 10^{-1}$	$9.0\ 10^{0}$	$7.5\ 10^{-1}$	$1.7\ 10^{0}$	$1.5\ 10^{0}$	$1.1\ 10^{0}$	$1.5\ 10^{-1}$	$1.7\ 10^{0}$
Norway											
Portugal	$3.8\ 10^{0}$	$2.7\ 10^{1}$	$6.8\ 10^{0}$	$4.7\ 10^{0}$	$3.0\ 10^{0}$	$3.0\ 10^{0}$	$1.8\ 10^{3}$	$5.7\ 10^{0}$	$1.5\ 10^{0}$	$0.0\ 10^{0}$	$4.5\ 10^{0}$
Spain	$9.2\ 10^{2}$	$3.5\ 10^{2}$	$2.7\ 10^{2}$	$1.8\ 10^{2}$	$4.3\ 10^{2}$	$8.4\ 10^{1}$	$1.8\ 10^{3}$	$2.4\ 10^{2}$	$2.4\ 10^{2}$	$9.0\ 10^{-1}$	$2.0\ 10^{2}$
Sweden											
United Kingdom	$3.8\ 10^{0}$	$1.5\ 10^{0}$	$3.0\ 10^{-1}$	$1.5\ 10^{-1}$	$0.0\ 10^{0}$	$1.5\ 10^{0}$	$8.3\ 10^{0}$	$7.5\ 10^{-1}$	$9.0\ 10^{-1}$	$0.0\ 10^{0}$	$0.0\ 10^{0}$
World	$2.0\ 10^{4}$	$2.1\ 10^{3}$	$3.2\ 10^{2}$	$2.3\ 10^{2}$	$1.2\ 10^{3}$	$2.4\ 10^{2}$	$4.2\ 10^{3}$	$1.1\ 10^{3}$	$5.1\ 10^{3}$	$7.7\ 10^{0}$	$4.7\ 10^{2}$

* Zero catch for Alboran Sea (deep), Eastern Basin, Western Basin, Black Sea (deep), Other Oceans.

TABLE D3.6 continued Annual mollusc consumption by Mediterranean Sea compartments used in PC-CREAM 08

Nation	Annual marine mollusc consumption t y^{-1}		
	Liguro - Provençal Basin	Northern Europe	Tyrrhenian Sea
Austria		$7.3\ 10^{3}$	$1.1\ 10^{0}$
Belgium	$2.0\ 10^{1}$	$6.0\ 10^{4}$	$1.5\ 10^{-1}$
Denmark	$3.0\ 10^{-1}$	$3.1\ 10^{5}$	
Finland		$7.7\ 10^{4}$	$9.6\ 10^{1}$
France	$2.0\ 10^{3}$	$3.8\ 10^{5}$	$1.1\ 10^{1}$
Germany	$2.3\ 10^{1}$	$3.6\ 10^{5}$	$1.1\ 10^{1}$
Greece	$9.0\ 10^{0}$	$1.5\ 10^{4}$	$0.0\ 10^{0}$
Ireland	$1.5\ 10^{-1}$	$6.9\ 10^{4}$	$2.6\ 10^{3}$
Italy	$7.4\ 10^{2}$	$8.8\ 10^{4}$	$6.0\ 10^{-1}$
Netherlands	$5.7\ 10^{0}$	$1.5\ 10^{5}$	
Norway		$9.5\ 10^{4}$	$7.5\ 10^{-1}$
Portugal	$2.3\ 10^{1}$	$1.5\ 10^{5}$	$1.3\ 10^{2}$
Spain	$8.9\ 10^{2}$	$3.8\ 10^{5}$	
Sweden		$2.0\ 10^{5}$	$7.5\ 10^{-1}$
United Kingdom	$3.0\ 10^{0}$	$6.6\ 10^{5}$	$2.8\ 10^{3}$

Table D3.7 Coastline lengths for Mediterranean Sea used in PC-CREAM 08

Nation	Coastline length (m)										
	Adriatic Sea	Aegean Sea	Alboran Sea (surface)	Algerian Basin	Black Sea surface	Cyprus Sea	Gulf of Lions	Ionian Sea	Levantin Basin	Libyan Sea	Liguro - Provençal Basin
Austria	-	-	-	-	-	-	-	-	-	-	-
Belgium	-	-	-	-	-	-	-	-	-	-	-
Denmark	-	-	-	-	-	-	-	-	-	-	-
Finland	-	-	-	-	-	-	-	-	-	-	-
France	-	-	-	-	-	-	$5.10\ 10^{5}$	-	-	-	$7.90\ 10^{5}$
Germany	-	-	-	-	-	-	-	-	-	-	-
Greece	-	$6.00\ 10^{6}$	-	-	-	-	-	$2.90\ 10^{6}$	$2.50\ 10^{5}$	$4.00\ 10^{5}$	-
Ireland	-	-	-	-	-	-	-	-	-	-	-
Italy	$1.50\ 10^{6}$	-	-	$3.50\ 10^{5}$	-	-	-	$9.50\ 10^{5}$	-	$3.00\ 10^{5}$	$8.00\ 10^{5}$
Netherlands	-	-	-	-	-	-	-	-	-	-	-
Norway	-	-	-	-	-	-	-	-	-	-	-
Portugal	-	-	-	-	-	-	-	-	-	-	-
Spain	-	-	$4.00\ 10^{5}$	$7.00\ 10^{5}$	-	-	$7.50\ 10^{4}$	-	-	-	$9.30\ 10^{5}$
Sweden	-	-	-	-	-	-	-	-	-	-	-
United Kingdom	-	-	-	-	-	-	-	-	-	-	-
World	$4.34\ 10^{6}$	$8.50\ 10^{6}$	$4.00\ 10^{5}$	$1.05\ 10^{6}$	$5.95\ 10^{6}$	$1.58\ 10^{6}$	$5.85\ 10^{5}$	$3.85\ 10^{6}$	$4.80\ 10^{5}$	$8.50\ 10^{5}$	$2.52\ 10^{6}$

* No coastline lengths for Alboran Sea (deep), Black Sea (deep), Eastern Basin, Western Basin.

Table D3.7 Continued - Coastline lengths for Mediterranean Sea used in PC-CREAM 08

Nation	Coastline length (m)		
	Northern Europe	Other Oceans	Tyrrhenian Sea
Austria	-	-	-
Belgium	$6.30\ 10^{4}$	-	-
Denmark	$1.63\ 10^{6}$	-	-
Finland	$2.57\ 10^{6}$	-	-
France	$2.28\ 10^{6}$	-	$2.50\ 10^{5}$
Germany	$9.90\ 10^{5}$	-	-
Greece	-	-	-
Ireland	$1.83\ 10^{6}$	-	-
Italy	-	-	$2.40\ 10^{6}$
Netherlands	$3.70\ 10^{5}$	-	-
Norway	$5.72\ 10^{6}$	-	-
Portugal	$1.17\ 10^{6}$	-	-
Spain	$1.22\ 10^{6}$	-	-
Sweden	$3.15\ 10^{6}$	-	-
United Kingdom	$6.27\ 10^{6}$	-	-
World	$7.46\ 10^{7}$	$2.38\ 10^{8}$	$2.65\ 10^{6}$

D4 FIGURE

FIGURE D4.1 Surface compartments of the Mediterranean Sea Model