Generic Models for Use in Assessing the Impact of Discharges of Radioactive Substances to the Environment

GENERIC MODELS FOR
USE IN ASSESSING THE IMPACT OF
DISCHARGES OF RADIOACTIVE
SUBSTANCES TO THE
ENVIRONMENT

The following States are Members of the International Atomic Energy Agency:

AFGHANISTAN	GUATEMALA	PANAMA
ALBANIA	HAITI	PARAGUAY
ALGERIA	HOLY SEE	PERU
ANGOLA	HUNGARY	PHILIPPINES
ARGENTINA	ICELAND	POLAND
ARMENIA	INDIA	PORTUGAL
AUSTRALIA	INDONESIA	QATAR
AUSTRIA	IRAN, ISLAMIC REPUBLIC OF	REPUBLIC OF MOLDOVA
BANGLADESH	IRAQ	ROMANIA
BELARUS	IRELAND	RUSSIAN FEDERATION
BELGIUM	ISRAEL	SAUDI ARABIA
BENIN	ITALY	SENEGAL
BOLIVIA	JAMAICA	SIERRA LEONE
BOSNIA AND HERZEGOVINA	JAPAN	SINGAPORE
BRAZIL	JORDAN	SLOVAKIA
BULGARIA	KAZAKHSTAN	SLOVENIA
BURKINA FASO	KENYA	SOUTH AFRICA
CAMBODIA	KOREA, REPUBLIC OF	SPAIN
CAMEROON	KUWAIT	SRI LANKA
CANADA	LATVIA	SUDAN
CHILE	LEBANON	SWEDEN
CHINA	LIBERIA	SWITZERLAND
COLOMBIA	LIBYAN ARAB JAMAHIRIYA	SYRIAN ARAB REPUBLIC
COSTA RICA	LIECHTENSTEIN	THAILAND
COTE D'IVOIRE	LITHUANIA	THE FORMER YUGOSLAV
CROATIA	LUXEMBOURG	REPUBLIC OF MACEDONIA
CUBA	MADAGASCAR	TUNISIA
CYPRUS	MALAYSIA	TURKEY
CZECH REPUBLIC	MALI	UGANDA
DEMOCRATIC REPUBLIC	MALTA	UKRAINE
OF THE CONGO	MARSHALL ISLANDS	UNITED ARAB EMIRATES
DENMARK	MAURITIUS	UNITED KINGDOM OF
DOMINICAN REPUBLIC	MEXICO	GREAT BRITAIN AND
ECUADOR	MONACO	NORTHERN IRELAND
EGYPT	MONGOLIA	UNITED REPUBLIC
EL SALVADOR	MOROCCO	OF TANZANIA
ESTONIA	MYANMAR	UNITED STATES OF AMERICA
ETHIOPIA	NAMIBIA	URUGUAY
FINLAND	NETHERLANDS	UZBEKISTAN
FRANCE	NEW ZEALAND	VENEZUELA
GABON	NICARAGUA	VIET NAM
GEORGIA	NIGER	YEMEN
GERMANY	NIGERIA	YUGOSLAVIA
GHANA	NORWAY	ZAMBIA
GREECE	PAKISTAN	ZIMBABWE

GENERIC MODELS FOR USE IN ASSESSING THE IMPACT OF DISCHARGES OF RADIOACTIVE SUBSTANCES TO THE ENVIRONMENT

INTERNATIONAL ATOMIC ENERGY AGENCY
VIENNA, 2001

FOREWORD

The concern of society in general for the quality of the environment and the realization that all human activities have some environmental effect has led to the development of a procedure for environmental impact analysis. This procedure is a predictive one, which forecasts probable environmental effects before some action, such as the construction and operation of a nuclear power station, is decided upon. The method of prediction is by the application of models that describe the environmental processes in mathematical terms in order to produce a quantitative result which can be used in the decision making process.

This report describes such a procedure for application to radioactive discharges and is addressed to the national regulatory bodies and technical and administrative personnel responsible for performing environmental impact analyses. The report is also intended to support the recently published IAEA Safety Guide on Regulatory Control of Radioactive Discharges to the Environment. It expands on and supersedes previous advice published in IAEA Safety Series No. 57 on Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases.

This Safety Report was developed through a series of consultants meetings and three Advisory Group Meetings. The IAEA wishes to express its gratitude to all those who assisted in its drafting and review. The IAEA officers responsible for the preparation of this report were C. Robinson, M. Crick and G. Linsley of the Division of Radiation and Waste Safety.

CONTENTS

TABLES CONTAINED IN THIS SAFETY REPORT

1. INTRODUCTION

1.1. BACKGROUND

The International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources (BSS) [1] establish basic and detailed requirements for protection against the risks associated with exposure to radiation and for the safety of radiation sources that may deliver such exposure. The standards are based primarily on the 1990 Recommendations of the International Commission on Radiological Protection (ICRP) [2] and other IAEA Safety Series publications. The BSS [1] place requirements on both the Regulatory Authority and on the legal person responsible for a source. These requirements and the procedures required to fulfill them are outlined in more detail in Ref. [3]. This Safety Report supports that publication and, in particular, provides the information necessary to allow the legal person responsible to "make an assessment of the nature, magnitude and likelihood of the exposures attributed to the source" [1]. It provides a practical generic methodology for assessing the impact of radionuclide discharges in terms of the resulting individual and collective radiation doses.

Previous guidance on models for predicting environmental transfer for assessing doses to the most exposed individuals (critical groups) was given in Safety Series No. 57 [4]. Since the publication of that report, the IAEA has produced Safety Series No. 100 on methods for evaluating the reliability of environmental transfer model predictions [5]. A handbook of transfer data for the terrestrial and freshwater environment [6] has also been produced which brings together relevant information from the major data collections in the world. Many of the parameter values used in this report are derived from the data in that handbook [6]. While Safety Series No. 57 contained much useful information and has become, to some extent, a standard text, in practice it was incomplete since it did not include all the models that were needed for assessment purposes. Moreover, considerable skill, expertise and resources were needed to derive and use appropriate data in the models.

This Safety Report expands on and supersedes the previous report [4]. It includes a new section on radiation dosimetry for intakes of radionuclides by members of the public and revised sections on atmospheric and aquatic dispersion. A section on calculating collective doses for screening purposes is also included to help determine whether further optimization procedures would be warranted. This Safety Report is intended to be a complete and self-contained manual describing a simple but robust assessment methodology that may be implemented without the need for special computing facilities.

1.2. OBJECTIVES

The main purpose of this Safety Report is to provide simple methods for calculating doses[1] arising from radioactive discharges into the environment, for the purpose of evaluating suitable discharge limits and to allow comparison with the relevant dose limiting criteria specified by the relevant Regulatory Authority.

1.3. SCOPE

The models in this Safety Report have been developed for the purpose of screening proposed radioactive discharges (either from a new or existing practice); that is for determining through a simplified but conservative assessment the likely magnitude of the impact, and whether it can be neglected from further consideration or whether more detailed analysis is necessary. The use of simple screening models for dose assessment is one of the first steps in registering or licensing a practice, as explained in more detail in Ref. [3]. A dose assessment will normally be required either to demonstrate that the source may be exempted from the requirements of the BSS, or as part of the authorization or licence application. A step-wise procedure for setting discharge limits is outlined in Ref. [3]. The function of the dose assessment within this process, and the value of an iterative procedure in which the complexity of the dose assessment method increases as the magnitude of the predicted doses increase, is outlined in Ref. [3] and discussed in Sections 2 and 8 of this report.

This Safety Report provides the information required to assess rapidly doses using a minimum of site specific information. Two alternative methods are presented — a 'no dilution' approach that assumes members of the public are exposed at the point of discharge, and a generic environmental screening methodology that takes account of dilution and dispersion of discharges into the environment.

The screening models contained in this report are expected to be particularly useful for assessing the radiological impact of discharges from small scale facilities, for example hospitals or research laboratories. In these situations the development of special local arrangements for dose assessment is likely to be unwarranted because the environmental discharges will usually be of a low level, and the methodology described in this report will usually be adequate. However, for many larger scale

---

[1] Unless otherwise stated, the term 'dose' refers to the sum of the effective dose from external exposure in a given period and the committed effective dose from radionuclides taken into the body in the same period.

2

nuclear facilities the assessed doses from the screening models presented in this report are more likely to approach the dose limiting criteria set by the Regulatory Authority (e.g. dose constraint), and users are more likely to need to follow a screening calculation with a more realistic, site specific and detailed assessment. Such a re-evaluation may necessitate consultation with professionals in radiological assessment and the application of more advanced models. The description of these advanced models is outside the scope of this Safety Report.

Doses calculated using the screening models presented in this report do not represent actual doses received by particular individuals. Furthermore, it would not be reasonable to use these models to reconstruct discharges from environmental monitoring measurements, because the pessimistic nature of the models might lead to a significant underestimation of the magnitude of the release.

The modelling approaches described in this report are applicable to continuous or prolonged releases into the environment when it is reasonable to assume that an equilibrium or quasi-equilibrium has been established with respect to the released radionuclides and the relevant components of the environment. The approaches described here are not intended for application to instantaneous or short period releases such as might occur in uncontrolled or accident situations.

1.4. STRUCTURE

Section 2 provides an overview of the assessment methodology and discusses the basic procedures for screening radionuclide discharges. The parameters and models for assessing the transfer between various environmental compartments for releases of radionuclides to the atmosphere, into surface waters and to sewerage systems are described in Sections 3 to 5 of this report. Section 6 provides the necessary dosimetric data and the equations by which individual doses may be evaluated. Section 7 considers collective doses, and Section 8 discusses the procedures to be followed when calculated doses approach the relevant dose limiting criteria.

In each section a simplified modelling procedure is described. Limitations in the models and their use are discussed. Default values are provided for each of the parameters needed for the assessment — these are chosen from observed values in such a manner as to produce only a small probability of underestimation of doses.

Annex I provides two types of dose calculation factors. The first, known as no dilution factors, allow rapid estimates to be made of the critical group doses arising from a concentration in air or water (resulting from a discharge to the atmosphere or a river). These factors are intended to be used with the predicted maximum radionuclide concentrations at the point of discharge. This approach is likely to overestimate significantly the doses received by members of the critical group in

reality. It is expected that these data will provide a useful screening method to determine whether the discharge source may be automatically exempted from the requirements of the BSS (see Refs [1] and [3] for further discussion). Annex I also provides generic dose calculation factors based on the generic environmental methods presented in this report, and standardized assumptions regarding the discharge conditions and the location of the critical group. These factors give the dose for a unit discharge to the atmosphere or to a river or sewer. It is recommended that site conditions should be taken into account in generic assessments if predicted doses exceed a reference level, as explained in more detail in Section 2.

Radionuclide half-lives and decay constants are provided in Annex II, and special methods for calculating doses from ^{3}H and ^{14}C are described in Annex III. Annex IV provides a number of example calculations that illustrate the main features of the model.

Annexes V–VII provide more detailed information on some of the models included in this report. Annex V is a description of the Gaussian plume model, Annex VI covers the model for radionuclide transport in surface waters and Annex VII gives an explanation of the methods used to assess collective doses.

A full listing of the parameter symbols used in the equations that describe the model is provided at the end of the report. These symbols are listed, for ease of reference, by the section in which they are used. A glossary of the terms used in this Safety Report is also provided.

2. PROCEDURES FOR
SCREENING RADIONUCLIDE DISCHARGES

2.1. DOSE CRITERIA AND CHOICE OF MODEL

An operation or practice that discharges radioactive materials into the environment is subject to evaluation according to the basic principles of radiation protection. These principles are described in the BSS [1], and the specific issues relating to the control of discharges into the environment are described in a recent IAEA Safety Guide [3]. As indicated in Ref. [3], the calculation of critical group doses is a necessary component of the development of a discharge authorization. This Safety Report provides a simple screening approach for assessing critical group

doses[2] and collective doses from discharges of radioactive substances into the environment.

Accurately assessing doses that could be received by members of the public can be a complex and time consuming process. In many situations, where the doses likely to be experienced by members of the public are very low and the expense of undertaking a site specific assessment would not be warranted, it is possible to make some simplifying and generally pessimistic assumptions that remove the necessity of applying complex modelling procedures or of gathering site specific data. This Safety Report provides the information needed to perform such simplified assessments and recommends a structured iterative approach for increasing the complexity of modelling as predicted doses approach or exceed a reference level which is related to dose limiting criteria specified by the Regulatory Authority.

The first stage in the iterative approach recommended in this report is a very simple assessment based on the conservative assumption that members of the public are exposed at the point of discharge. This is referred to as the *no dilution model*. Dose calculation factors based on this approach are presented in Annex I of this report. As indicated in Fig. 1, it is recommended that a greater level of model complexity would be necessary if the critical group dose predicted by the no dilution model exceeds the relevant dose criterion (e.g. dose constraint). The second stage in the iterative process is to use a simple *generic environmental model* that accounts for the dispersion of radioactive materials in the environment. This model is explained in some detail in this Safety Report. Simple dose calculation factors, based on this approach, are also provided in Annex I. These factors are based on the generic environmental model and some standardized assumptions about discharge conditions, the location of food production and the habits and location of the critical group. As indicated in Fig. 1, if predicted doses based on this generic environmental model exceed a reference level, the next stage in the iterative assessment process is to examine the generic input data for applicability to the site in question. If the data are overly conservative or otherwise inapplicable, a modified generic assessment is called for. If the doses predicted using this approach also exceed the reference level it may be necessary to consult a relevant expert to undertake a full site specific assessment.

2.1.1. Reference level

The choice of a value for the reference level to indicate when a greater level of model complexity is needed warrants discussion. It is recommended that this level be

---

[2] Dose criteria for members of the public are generally expressed in terms of the average dose to the critical group. A critical group is representative of those members of the public likely to be most exposed (see the glossary).
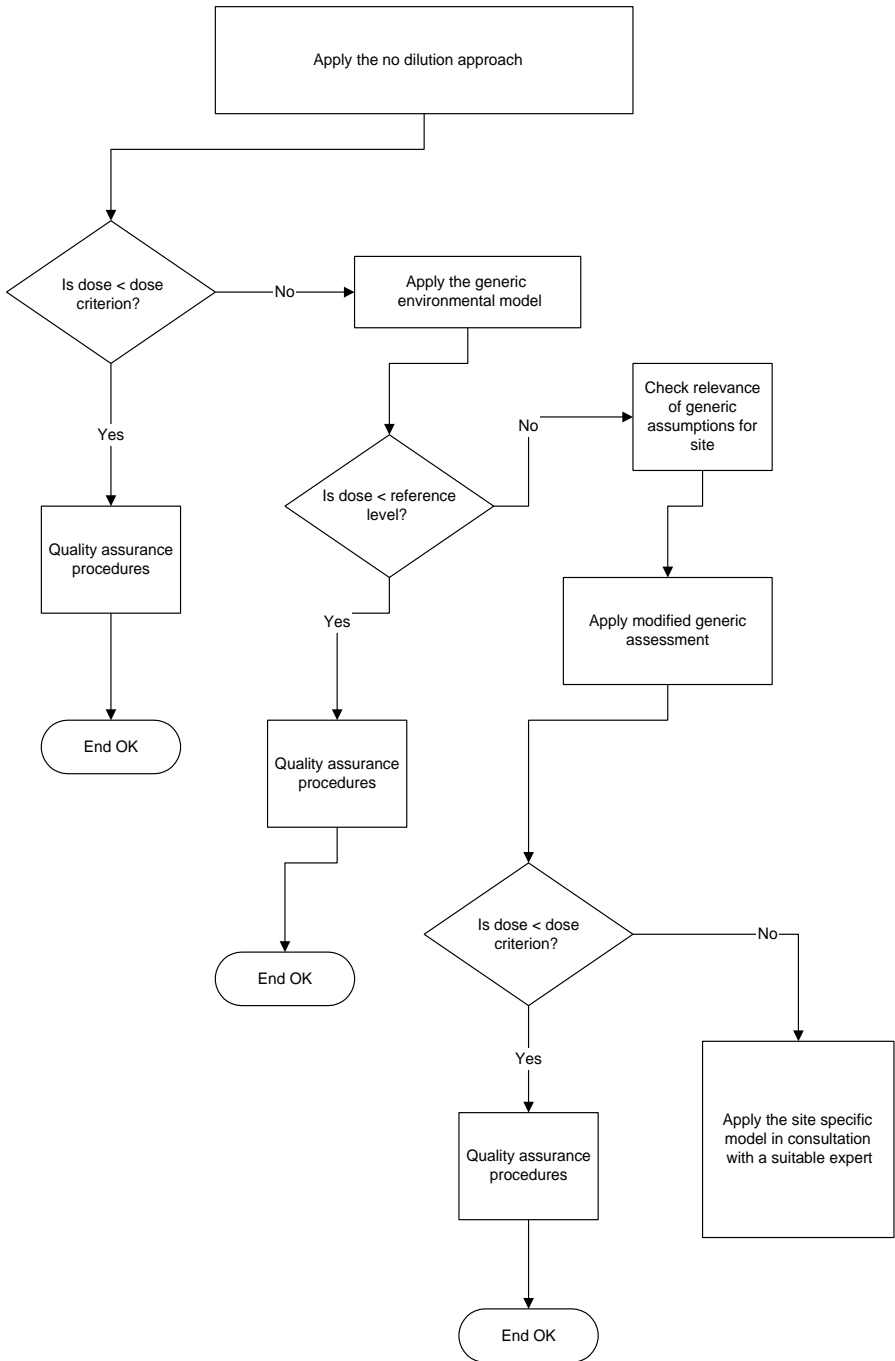
FIG. 1. Iterative approach for assessing critical group doses.

specified to take account of both the relevant dose limiting criterion (e.g. the dose constraint specified by the Regulatory Authority) and the level of uncertainty associated with the model predictions. In this context it is important to note that the generic environmental model and associated parameters presented in this report were derived such that

— Hypothetical critical group doses are generally likely to be overestimated,
— Under no circumstances would doses be underestimated by more than a factor of ten.

Thus it is fairly certain that doses experienced by the critical group will not exceed a particular dose criterion if the doses predicted using the generic model are less than one tenth of that criterion. This is consistent with the recommendation in Ref. [3] that a reference level of 10% (or one tenth) of the dose constraint is a reasonable basis for determining whether it is necessary to refine a dose assessment. The use of such a reference level to determine whether the no dilution approach is sufficient would be overcautious in view of the extremely conservative nature of this approach. Comparison with the relevant dose criterion is therefore recommended as the basis for deciding whether a more detailed assessment is necessary. A detailed description of the recommended iterative approach to be followed is given in Section 8.

2.2. GENERAL ASSESSMENT APPROACH

An overview of the assessment approach and the main parameters required to make an assessment are given in Fig. 2. The first step in this approach is to estimate the nature and magnitude of the proposed discharge of radioactive material into the environment, taking into account the period over which it is likely to occur. Transport of materials discharged to the atmosphere, surface water or a sewerage system is modelled and the concentrations of radionuclides at locations where people may be exposed is assessed. Discharges to sewerage systems are assumed to result in exposure of workers at the sewage plant only. Projected doses arising from the other discharge routes are calculated at the point of discharge for the no dilution model, or at the closest locations where members of the public have access (e.g. for external dose and inhalation dose calculations) or at the closest food production location (for ingestion doses) for the generic environmental model. The assumed location, habits and behaviour of members of the public are representative of those people likely to be most exposed (the critical group).

The model is designed to estimate the maximum annual dose received during the period of the practice. The inventory of long lived radionuclides builds up in the environment, with the result that exposures may increase as the discharge continues.
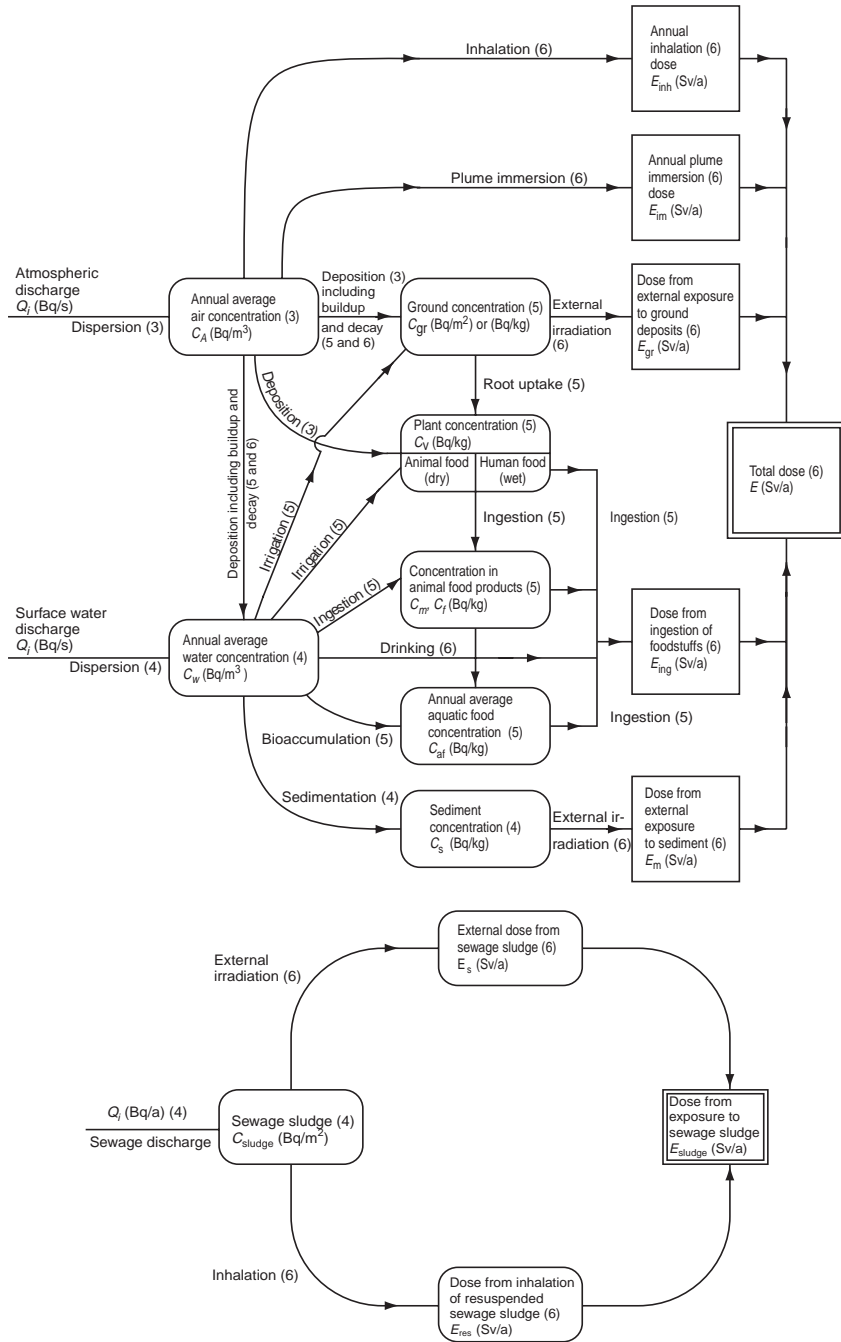
FIG. 2. Overview of the general assessment approach. Numbers in parentheses refer to the section of this report in which the process is discussed.

For generic model purposes the maximum annual dose is assumed to be the dose that would be received in the final year of the practice. A default discharge period of 30 years is assumed, with the result that doses are estimated for the 30th year of discharge, and include the contribution to the dose from all material discharged in the previous 29 years.

The exposure pathways considered, and the information needed to assess their contributions to the dose, are illustrated in Fig. 2. External exposures from immersion in the plume and from material deposited on surfaces are included. Methods for assessing internal exposure, from the inhalation of radionuclides in the air and the ingestion of radionuclides in food and water, are also provided. The recommended approach to account for exposures from multiple pathways is by simple summation over those pathways. In reality, it is unlikely that a member of the 'true' critical group[3] would be in the most exposed group for all exposure pathways. However, the significance of this potential compounding of pessimistic assumptions is somewhat lessened by the fact that the total dose is seldom dominated by more than a few radionuclides and exposure pathways.

2.2.1. Estimation of the annual average discharge rate

In order to estimate the annual discharge rates for the screening models, information is required on the quantities and types of radionuclides to be discharged, the mode of discharge and the discharge points. In order to apply these models, the discharge rate should be specified separately for the different release routes; that is for discharges to the atmosphere (used as input in Section 3) and for discharges into surface water or sewerage systems (used as input in Section 4).

The effects of any anticipated perturbations in the annual average discharge rate should be taken into account. For example, it is recommended that operational perturbations that are anticipated to occur with a frequency greater than 1 in 10 per year should be included in the discharge rate estimate. In making this assessment care should be taken to determine whether such perturbations are uniformly or randomly spaced over the year. If they are dominated by a single event, a different dose assessment approach may be needed.

The inherent conservatism in the screening model is sufficient to accommodate uncertainties in the discharge rate if these uncertainties are no larger than a factor of two. Therefore a more realistic discharge rate estimate may be used in preference to a pessimistically derived one if the uncertainty in its value is less than about a factor of two.

---

[3] In this context the true critical group is intended to represent those members of the public most exposed from a particular source, including contributions from all exposure pathways.

2.2.2. Estimation of environmental concentrations

2.2.2.1. Air and water

Once the discharge rate has been quantified, the next step in the procedure is to estimate the relevant annual average radionuclide concentration in air or water for the discharge route of concern. If the no dispersion model is applied, the concentration at the point of discharge is needed, while if the generic environmental model is applied the concentration at the location nearest to the facility at which a member of the public will be likely to have access, or from which a member of the public may obtain food or water, is needed. The methods for estimating radionuclide concentrations in air are outlined in Section 3, while the approach for estimating concentrations in surface waters is outlined in Section 4. A screening model for estimating radionuclide transport in sewerage systems and accumulation in sewage sludge is also described in Section 4.

The atmospheric dispersion model of the generic environmental model is designed to estimate annual average radionuclide concentrations in air and the annual average rate of deposition resulting from ground level and elevated sources of release. In locations where air flow patterns are influenced by the presence of large buildings, the model accounts for the effects of buildings on atmospheric dispersion of radionuclides. The surface water model accounts for dispersion in rivers, small and large lakes, estuaries and along the coasts of oceans.

2.2.2.2. Terrestrial and aquatic foods

The methods for assessing radionuclide concentrations in terrestrial and aquatic food products (assuming equilibrium conditions) are described in Section 5. The average concentrations in terrestrial foods representative of the 30th year of operation may be estimated from the annual average rate of deposition (Section 3), taking account of the buildup of radionuclides on surface soil over a 30 year period. The types of terrestrial foods considered in Section 5 are milk, meat and vegetables. The uptake and retention of radionuclides by terrestrial food products can take account of direct deposition from the atmosphere, and irrigation and uptake from soils. The effect of radionuclide intake through inadvertent soil ingestion by humans or grazing animals is implicitly taken into account within the element specific values selected for the soil to plant uptake coefficient.

The uptake and retention of radionuclides by aquatic biota is described in Section 5. The model uses selected element specific bioaccumulation factors that describe an equilibrium state between the concentration of the radionuclide in biota and water. The types of aquatic foods considered are freshwater fish, marine fish and marine shellfish.

The use of surface water as a source of spray irrigation may be taken into account by using the average concentration of the radionuclide in water, determined from Section 4, and appropriate average irrigation rates, from Section 5, to estimate the average deposition rate on to plant surfaces or agricultural land. Irrigation is assumed to occur for a period of 30 years. The contamination of surface water from routine discharges to the atmosphere is considered for both small and large lakes. In the case of a small lake, the estimate of direct deposition from the atmosphere is modified by a term representing runoff from a contaminated watershed.

The process of radioactive decay is taken into account explicitly in the estimation of the retention of deposited radioactive materials on the surfaces of vegetation and on soil, and in the estimation of the losses owing to decay that may occur during the time between harvest and human consumption of a given food item (Section 5).

2.2.3. Estimation of doses

As described in Section 6, calculated average radionuclide concentrations in air, food and water (representative of the 30th year of discharge) are combined with the annual rates of intake to obtain an estimate of the total radionuclide intake during that year. This total intake over the year is then multiplied by the appropriate dose coefficient, given in Section 6, to obtain an estimate of the maximum effective dose in one year from inhalation or ingestion. In a similar manner, the concentrations of radionuclides in shoreline sediments (Section 4) and surface soils (Section 5) in the 30th year of discharge are used with appropriate dose coefficients to estimate the effective dose received during that year from external irradiation.

The effective dose in one year from immersion in a cloud containing radionuclides may be calculated by multiplying the average concentration in air (Section 3) by the appropriate external dose coefficients in Section 6.

To obtain the total maximum effective dose in one year (representative of the 30th year of discharge), the effective doses from all radionuclides and exposure pathways are summed. The equivalent dose estimates for the eyes and skin are summed only for these tissues.

2.2.4. Screening estimates of collective dose

Section 7 provides tables of collective dose per unit activity discharged to the atmosphere and to the aquatic environment for a selection of radionuclides. These tables may be valuable for determining whether further optimization studies are worthwhile. These data are not intended for the purpose of rigorous site specific optimization analyses. The values provided in Section 7 are collective dose commitments, integrated to infinity. These data have uncertainties of the order of a factor of ten, and the data for very long lived radionuclides represent very crude approximations.

3. ATMOSPHERIC DISPERSION

After release to the atmosphere, radionuclides undergo downwind transport (advection) and mixing processes (turbulent diffusion). Radioactive material will also be removed from the atmosphere by both wet and dry deposition on to the ground, and by radioactive decay. The most relevant mechanisms involved [7] are illustrated in Fig. 3. A model that takes account of these processes is needed to assess radionuclide concentrations at locations downwind of the release. This section describes a simple generic atmospheric dispersion model that allows for the above processes, and for the effects of any buildings in the vicinity of the release. Tables of dispersion factors are provided to permit annual average radionuclide concentrations in air, C_A (Bq/m^3), to be estimated on the basis of very limited site specific data.

Before describing this generic model, the relationship between radionuclide concentration in air and the release in the absence of dispersion is given. This provides the basis for the simple pessimistic no dilution approach described earlier and for the data presented in Annex I.

3.1. SCREENING CALCULATIONS

As indicated earlier, the simplest and most pessimistic screening technique is to assume that the radionuclide concentration at the point of interest (often referred to as the receptor location) is equal to the atmospheric radionuclide concentration at the point of release. Thus

$$C_A = \frac{P_p Q_i}{V} \tag{1}$$

where

C_A is the ground level air concentration at downwind distance x (Bq/m^3),
Q_i is the average discharge rate for radionuclide i (Bq/s),
V is the volumetric air flow rate of the vent or stack at the point of release (m^3/s),
P_p is the fraction of the time the wind blows towards the receptor of interest (dimensionless).

A value of P_p = 0.25 has been suggested for screening purposes [8–10]. A value of C_A calculated using Eq. (1) can be used to calculate radionuclide concentrations on the ground (Section 3.9) and subsequent doses to a member of the public located at the receptor point from other potential pathways of exposure (see Annex I). If the doses calculated in this way exceed a reference level, discussed in
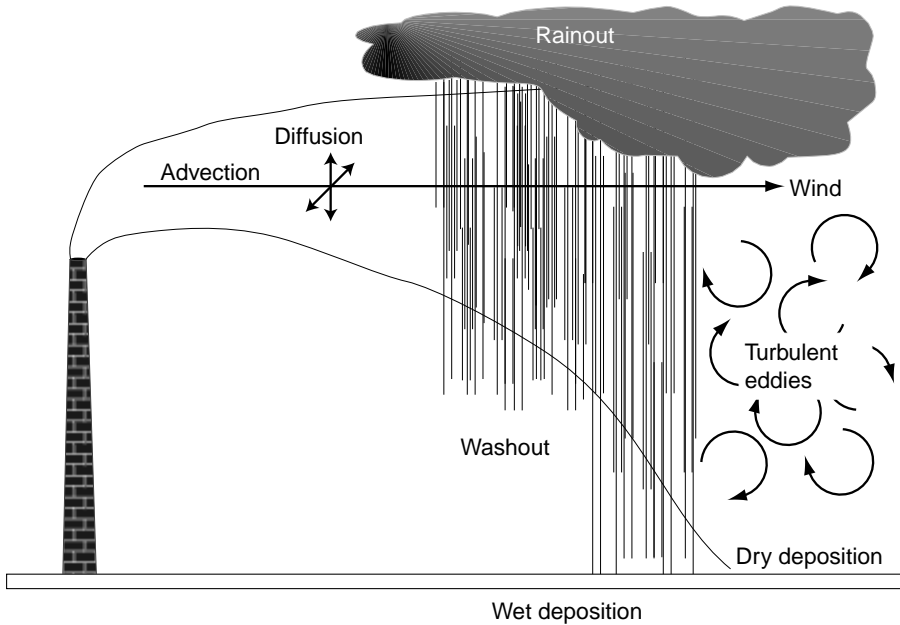
FIG. 3. The most important processes affecting the transport of radionuclides released to the atmosphere.

Section 2, a further assessment that takes account of dispersion is recommended. The remainder of this section provides the information necessary for such an assessment.

3.2.   FEATURES OF THE DISPERSION MODEL

The Gaussian plume model is applied here to assess the dispersion of long term atmospheric releases; this model is widely accepted for use in radiological assessment activities [11]. The model is considered appropriate for representing the dispersion of either continuous or long term intermittent releases within a distance of a few kilometres of the source. For the purposes of this report long term intermittent releases are defined as those for which the short term source strength, released momentarily or continuously per day, does not exceed 1% of the maximum annual source strength, estimated assuming a constant release rate [12]. The methods described here should not be used to calculate radionuclide concentrations in air resulting from short term releases that fail this criterion.

A more detailed discussion of the Gaussian plume model and its limitations is presented in Annex V. References [7, 11, 13, 14] provide a general overview of the use of atmospheric dispersion models in radiological dose assessments. A more detailed explanation of atmospheric transport phenomena is provided in the numerous scientific books and reports that have been published in this field, for example Refs [15–22].

3.3. BUILDING CONSIDERATIONS

The version of the Gaussian plume model that is appropriate depends on the relationship between the height at which the effluent is released H (m) and the height of the buildings that affect airflow near the release point H_B (m). The presence of buildings and other structures, such as cooling towers, will disturb the flow of air. Idealized flow around a simple building is shown schematically in Fig. 4. The three main zones of flow around a building are
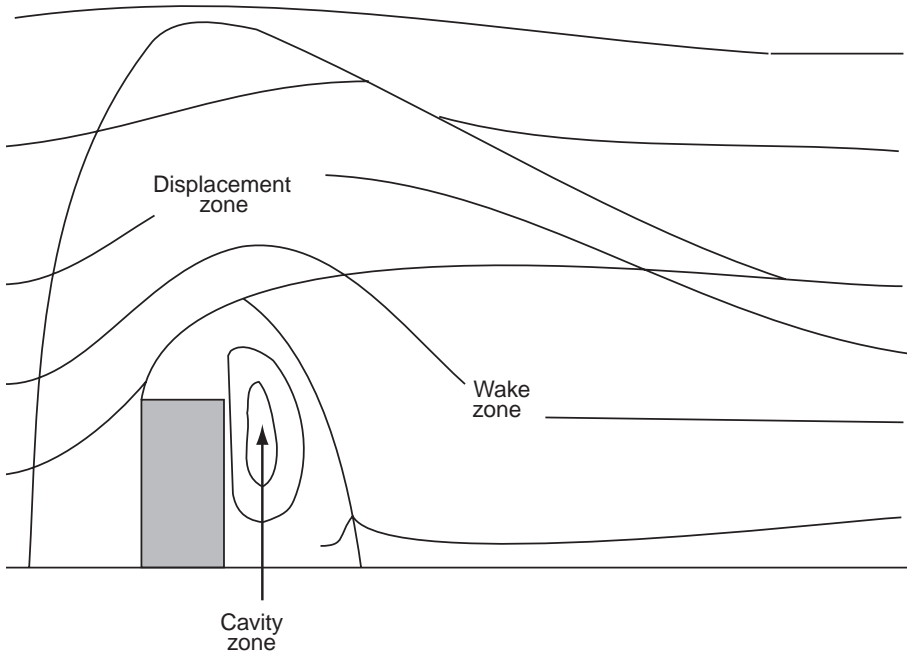
FIG. 4. Air flow around a building, showing the three main zones of flow: displacement zone, wake zone and cavity zone.

14

(a)   The upwind displacement zone, where the approaching air is deflected around the building.

(b)   The relatively isolated cavity zone immediately on the leeward side of the building.

(c)   The highly disturbed wake zone further downwind from the building [23]. The wake zone may extend downwind for some distance (the exact distance depends upon the source configuration and meteorological conditions) [24].

The building from which the release occurs is generally assumed to be the one that most influences the resulting plume dispersion. However, this is not always the case. If the release point is on a building in the immediate vicinity of a much larger building, the larger building is likely to exert more influence on plume dispersion than the smaller one from which the release originated [25].

The prevailing dispersion pattern depends upon both the release height and the receptor location (H and x) relative to the building geometry. For example, if the release height (H) is greater than 2.5 times the building height (H_B), that is

$$H > 2.5H_B$$

then dispersion can be considered to be undisturbed, that is in the displacement zone. If, however

$$H \leq 2.5H_B \text{ and } x > 2.5\sqrt{A_B}$$

where A_B is the projected cross-sectional area of the building most influencing the flow of the plume, then dispersion is considered to be inside the wake zone. (For screening purposes A_B may be assumed to be the surface area of the largest wall of the building nearest the receptor.) Dispersion inside the cavity zone is defined by

$$0 \leq H \leq 2.5H_B \text{ and } 0 \leq x \leq 2.5\sqrt{A_B}$$

Figure 5 illustrates these zones schematically.

Using the model, radionuclide concentrations in the air can be evaluated for the following dispersion situations.

(a)   Dispersion in the lee of an isolated point source, for example for releases from high stacks (displacement zone) — see Section 3.4;

(b)   Dispersion in the lee of, and reasonably distant from, a building, but still under the influence of its wakes, for example for releases from shorter stacks (wake zone) — see Section 3.5;

(c)   Dispersion where the source and receptor are on the same building surface (cavity zone) — see Section 3.6.1;
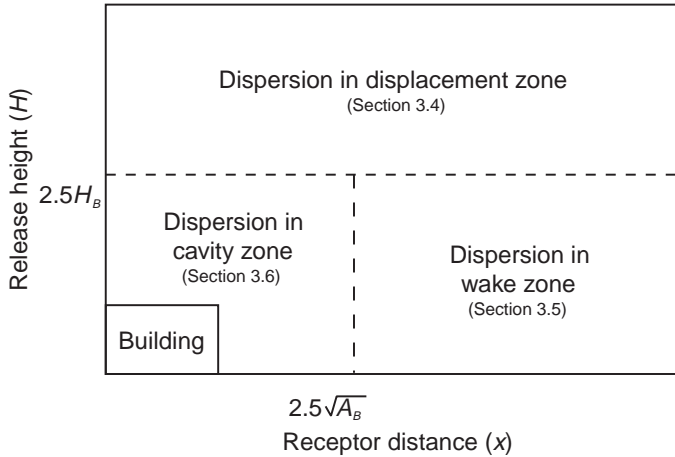
FIG. 5. Relationship between release height and receptor distance for determination of the type of dispersion model to be used.

(d)   Dispersion where the receptor is very close to, but not on, a building, for example releases from a vent on a building (cavity zone) — see Section 3.6.2.

A flow chart showing the choice of appropriate dispersion conditions for these screening calculations is given in Fig. 6. For more detailed information on these procedures see Refs [24, 26, 27].

3.4. DISPERSION IN THE LEE OF AN ISOLATED POINT SOURCE, $H > 2.5H_B$

The methods presented in this section are designed to be used for all cases that do not include building wake effects. This situation is depicted qualitatively in Fig. 7. The condition fulfilled is

$$H > 2.5H_B$$

In this case the sector averaged form of the Gaussian plume model (see Annex V) may be used with the following simplifying assumptions.
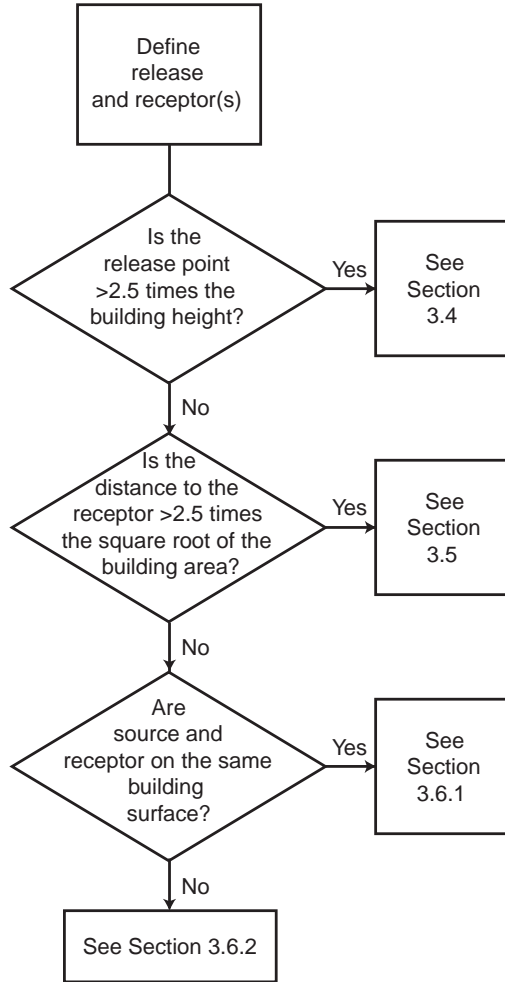
FIG. 6. Selection of an appropriate dispersion model for screening calculations.

(a)   A single wind direction for each air concentration calculation — see Section 3.7,

(b)   A single long term average wind speed,

(c)   A neutral atmospheric stability class (Pasquill–Gifford stability class D) [26].

Based on these assumptions, the screening model for atmospheric dispersion can be represented by
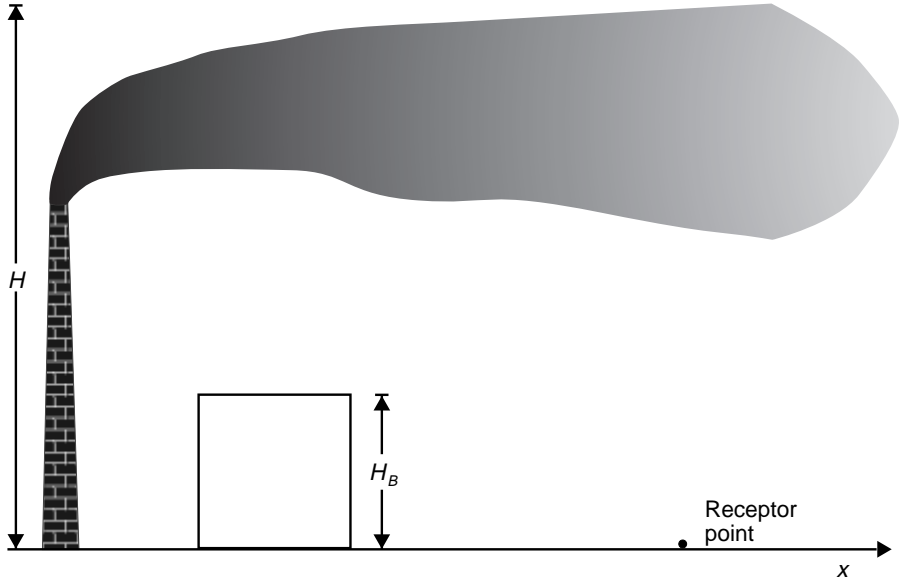
FIG. 7. Air flow in the displacement zone ($H > 2.5H_B$). Building wake effects do not need to be considered.

$$C_A = \frac{P_p F Q_i}{u_a} \qquad (2)$$

where

C_A is the ground level air concentration at downwind distance x in sector p (Bq/m^3),
P_p is the fraction of the time during the year that the wind blows towards the receptor of interest in sector p,
u_a is the geometric mean of the wind speed at the height of release representative of one year (m/s),
F is the the Gaussian diffusion factor appropriate for the height of release H and the downwind distance x being considered (m^{-2}),
Q_i is the annual average discharge rate for radionuclide i (Bq/s).

Values of F as a function of downwind distance x for various values of H are presented in Table I. These values were derived using the 30° sector averaged form of the Gaussian plume model; that is

$$F = \frac{12}{\sqrt{2\pi^3}} \times \frac{\exp\left[-\left(H^2/2\sigma_z^2\right)\right]}{x\sigma_z} \qquad (3)$$

where σ_z is the vertical diffusion parameter (m).

These expressions are appropriate for dispersion over relatively flat terrain without pronounced hills or valleys. The terrain is assumed to be covered with pastures, forests and small villages [27–31]. Three different expressions for the diffusion parameter were used in Eq. (3) to derive Table I. These assumptions are specified in the notes to that table.

The general behaviour of the Gaussian plume model diffusion factor F as a function of downwind distance for an elevated release is shown in Fig. 8. For screening purposes, however, it is assumed in this Safety Report that, for all $H > 0$, F is constant between the point of release and the distance corresponding to the maximum value of F for that value of H — see the dashed line in Fig. 8. This approach clearly overestimates the concentrations near the source, but it is considered

TABLE I. DISPERSION FACTOR (F, m^{-2}) FOR NEUTRAL ATMOSPHERIC STRATIFICATION

Downwind distance, x (m)	Release height, H (m)						
	0–5[a]	6–15[a]	16–25[a]	26–35[a]	36–45[a]	46–80[b]	>80[b]
100	3×10^{-3}	2×10^{-3}	2×10^{-4}	8×10^{-5}	3×10^{-5}	2×10^{-5}	1×10^{-5}
200	7×10^{-4}	6×10^{-4}	2×10^{-4}	8×10^{-5}	3×10^{-5}	2×10^{-5}	1×10^{-5}
400	2×10^{-4}	2×10^{-4}	1×10^{-4}	8×10^{-5}	3×10^{-5}	2×10^{-5}	1×10^{-5}
800	6×10^{-5}	6×10^{-5}	5×10^{-5}	4×10^{-5}	3×10^{-5}	2×10^{-5}	1×10^{-5}
1 000	4×10^{-5}	4×10^{-5}	4×10^{-5}	3×10^{-5}	3×10^{-5}	1×10^{-5}	1×10^{-5}
2 000	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	4×10^{-6}	5×10^{-6}
4 000	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	1×10^{-6}	2×10^{-6}
8 000	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	3×10^{-7}	5×10^{-7}
10 000	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	2×10^{-7}	3×10^{-7}
15 000	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	1×10^{-7}	1×10^{-7}
20 000	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	3×10^{-7}	6×10^{-8}	9×10^{-8}

[a] Calculated on the basis of the following relationship [24]

$$\sigma_z = (0.06)(x)/\sqrt{1+(0.0015)(x)}$$

[b] Calculated on the basis of the following relationship

$$\sigma_z = Ex^G$$

where $E = 0.215$ and $G = 0.885$ for release heights of 46–80 m, and $E = 0.265$ and $G = 0.818$ for release heights greater than 80 m [29–31].
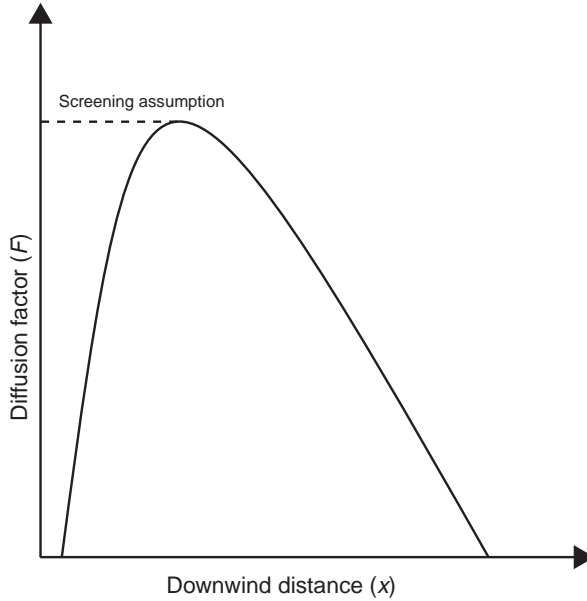
FIG. 8. Relationship between the Gaussian plume diffusion factor (F) and the downwind distance (x) for a given release height (H). In this screening approach, the maximum value of F for a given value of H (indicated by the dashed line) is used for all distances less than or equal to the distance corresponding to the maximum value of F.

appropriate for screening purposes to ensure that actual doses are not underestimated by more than a factor of ten.

3.5. DISPERSION IN THE LEE OF A BUILDING INSIDE THE WAKE ZONE

The methods described in this section are to be applied to all cases characterized by the following criteria

$$H \leq 2.5 H_B \text{ and } x > 2.5 \sqrt{A_B}$$

Such a situation is shown qualitatively in Fig. 9. The concentration of radionuclides in air is estimated using Eq. (2) corrected by a diffusion factor B (m^{-2}) instead of F; that is

$$C_A = \frac{P_p B Q_i}{u_a} \tag{4}$$
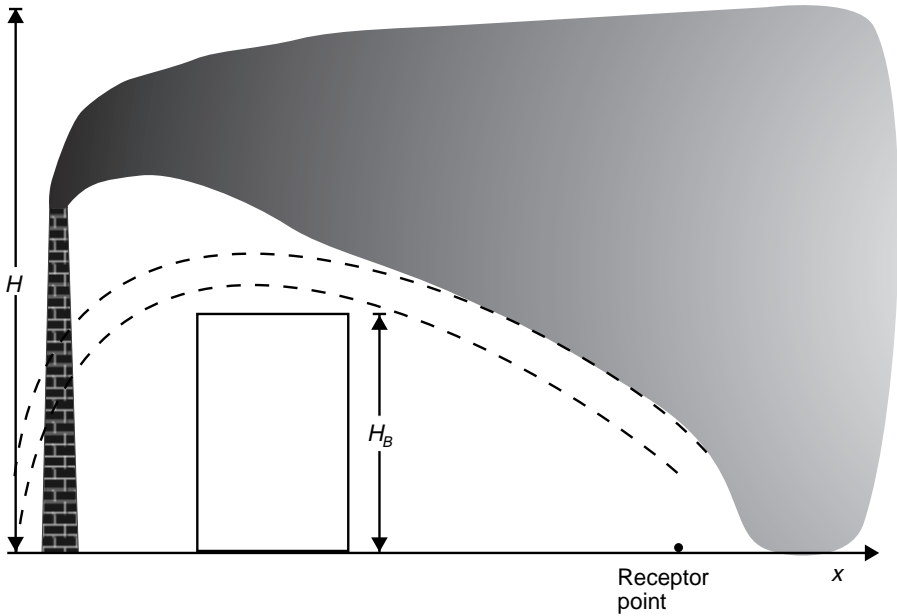
FIG. 9. Air flow in the wake zone ($H \leq 2.5 H_B$ and $x > 2.5 \sqrt{A_B}$).

where P_p, u_a and Q_i are as before (see Section 3.4) and

$$B = \frac{12}{\sqrt{2\pi^3}} \times \frac{1}{x \Sigma_z} \tag{5}$$

where

$$\Sigma_z = \left(\sigma_z^2 + \frac{A_B}{\pi} \right)^{0.5} \text{ for } x \geq 2.5 \sqrt{A_B} \tag{6}$$

where

A_B is the surface area of the appropriate wall of the building of concern (m^2),
σ_z is the vertical diffusion parameter (m) used in Eq. (3).

For long term dispersion, based on an assumed release height $H = 0$, the ground level activity concentration can be calculated by Eq. (4), see Refs [22] or [24]. The respective numerical values of B for various cross-sectional areas of buildings are

TABLE II.   DISPERSION FACTOR WITH BUILDING WAKE CORRECTION (B, m^{-2}) FOR NEUTRAL ATMOSPHERIC STRATIFICATION

Downwind distance, x (m)	Building surface area (m^2)									
	0–100	101–400	401–800	801–1200	1201–1600	1601–2000	2001–3000	3001–4000	4001–6000	>6000
100	3×10^{-3}	2×10^{-3}	1×10^{-3}	9×10^{-4}	8×10^{-4}	7×10^{-4}	6×10^{-4}	5×10^{-4}	4×10^{-4}	3×10^{-4}
200	7×10^{-4}	6×10^{-4}	5×10^{-4}	4×10^{-4}	3×10^{-4}	3×10^{-4}	3×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}
400	2×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}	1×10^{-4}	1×10^{-4}	1×10^{-4}	1×10^{-4}	9×10^{-5}	8×10^{-5}
800	6×10^{-5}	6×10^{-5}	6×10^{-5}	5×10^{-5}	5×10^{-5}	5×10^{-5}	5×10^{-5}	4×10^{-5}	4×10^{-5}	4×10^{-5}
1 000	4×10^{-5}	4×10^{-5}	4×10^{-5}	4×10^{-5}	4×10^{-5}	3×10^{-5}	3×10^{-5}	3×10^{-5}	3×10^{-5}	3×10^{-5}
2 000	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}	1×10^{-5}
4 000	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}	4×10^{-6}
8 000	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}
10 000	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}	1×10^{-6}
15 000	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}	5×10^{-7}
20 000	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}	4×10^{-7}

shown in Table II. These values represent reasonable estimates of turbulent mixing for ground level releases ($H = 0$) only. Application to elevated releases results in rather crude and pessimistic estimates of the dispersion situation, which are nevertheless appropriate for screening purposes.

3.6. DISPERSION IN THE LEE OF A BUILDING INSIDE THE CAVITY ZONE

The methods considered in this section are to be applied when

$$H \leq 2.5H_B \text{ and } x \leq 2.5\sqrt{A_B}$$

Such a situation is depicted qualitatively in Fig. 10. It should be noted that A_B represents the surface area of the largest wall of the building most influencing the plume flow [27]. This is often assumed to be the building from which the release occurs. However, if a release occurs in the midst of a complex of structures, other buildings in the vicinity of the release may influence plume flow to a greater extent.

Receptor points
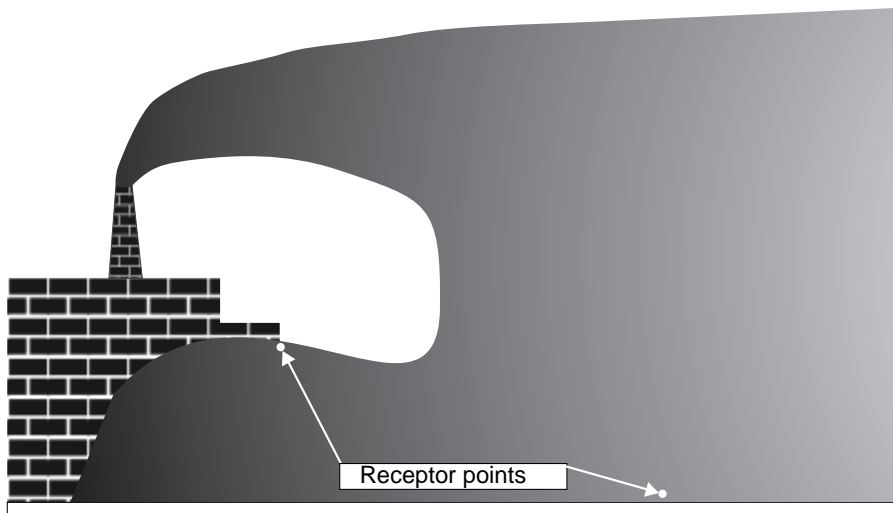
FIG. 10. Air flow in the cavity zone ($H \leq 2.5H_B$ and $x \leq 2.5\sqrt{A_B}$). The approach to be used depends on whether the receptor is or is not located on the same building surface as the release.

It is important to ensure that the correct building is considered when calculations using these procedures are made [25].

3.6.1. Source and receptor on same building surface

This case represents the situation where the receptor point or person of interest is on the same building surface, for example a roof or side wall opening such as a window, or in the building from which the release occurs. To predict the maximum concentration expected at a receptor located x metres from the release point, the following procedure has been adapted from information given in Ref. [27].

(a) If x is less than or equal to three times the diameter of the stack or vent from which the radionuclide is emitted, it may be assumed that no dilution occurs in the atmosphere and, as a result, the air concentration at the receptor point is equal to the concentration of the radionuclide at the point of release (as given in Eq. (1)).
(b) If x is greater than three times the diameter of the stack or vent, use Eq. (7) below to calculate the air concentration with $B_0 = 30$

$$C_A = \frac{B_0 Q_i}{u_a x^2} \tag{7}$$

The unitless constant B_0 accounts for potential increases in the concentration in air along a vertical wall owing to the presence of zones of air stagnation created by building wakes.

3.6.2. Source and receptor not on same building surface

For this situation the following equation is used to calculate the radionuclide concentration in air [32]

$$C_A = \frac{P_p Q_i}{\pi u_a H_B K} \tag{8}$$

where K is a constant of value 1 m. This model is an empirical formulation that yielded conservative predicted concentrations in air when compared with about 40 sets of field data from tracer experiments around nuclear reactor structures. If the width of the building under consideration is less than its height, the width of the building should be used in place of H_B in Eq. (8) [33].

3.7. DEFAULT INPUT DATA

The generic approach described above has been designed to require a minimum input of data by the user. The radionuclide discharge rate and the location of the release point and the receptor (i.e. Q_i, H, H_B, A_B and x), however, must be specified for the particular situation considered. No default values can be given for these parameters, although standardized assumptions have been applied to derive the generic dose calculation factors given in Annex I. The user is cautioned that these parameters should be determined as accurately as possible. The value of H used should ideally include both the physical height of the release point and any additional height resulting from the rise of the effluent plume owing to thermal or mechanical effects [28]. However, neglecting plume rise will tend to result in an overprediction of downwind air concentrations and, therefore, is appropriate for a generic assessment.

The building surface area used, A_B, should be that of the building most influencing the air flow around the source [27]. For most releases this will be the building from which the release occurs. If the release point is surrounded by other buildings and similar structures, such as cooling towers, a building other than the one associated with the release may need to be used for A_B. The downwind distance x used in the screening calculations should be the location of the nearest point of interest for dose calculation purposes. This location may be different for different pathways of exposure. For example, the nearest point of public access to a facility would be appropriate for the inhalation and external exposure pathways, while the nearest location where food could be grown would be appropriate for assessing doses from terrestrial food pathways. This may result in multiple air concentrations being required for a single facility assessment. It should also be noted that the Gaussian plume model is not generally applicable at $x > 20$ km. As a result, it is recommended that any receptors of concern that are beyond 20 km from the release point should be considered to be at $x = 20$ km for generic assessment purposes. In using Tables I and II, if x falls between two values given in the table the smaller distance should be used.

The only meteorological variables required are P_p and u_a. For detailed long term radiological dose assessments the frequency with which the wind blows in each of 12 cardinal wind directions may be obtained from local climatological information. (The sum of the frequencies for all directions is automatically equal to 1.) For generic assessment purposes, however, only one wind direction is considered for each individual air concentration calculation. To help reduce the chance of the predicted values being more than a factor of ten lower than the actual doses, it should be assumed that $P_p = 0.25$ for this single direction. It is preferable to use a site specific value for the wind speed u_a at the height of the release, determined either by measurement or by extrapolation [7]. In the event

that information on u_a appropriate for the release location is not readily available, however, it is suggested that a default value of u_a = 2 m/s be used [8–10].

3.8. PLUME DEPLETION

As noted earlier, and in Fig. 3, a plume of radioactive material may be depleted as it moves downwind. Removal processes include [7]

— Radioactive decay,
— Plume depletion by dry deposition,
— Plume depletion by precipitation scavenging (wet deposition).

Over the relatively short distances of a few kilometres considered here, corrections of the plume activity owing to wet and/or dry deposition can usually be neglected when calculating the annual average concentrations of radionuclides in air. In any case, when plume depletion is not incorporated into the assessment, the result will be conservative.

Depletion of the activity in the plume owing to radioactive decay can also be neglected for most of the radionuclides. This is particularly true if discharges occur after initial retention of short lived radionuclides. Thus radioactive decay may be neglected, except for very short lived nuclides. Where radioactive decay is likely to be significant, calculated air concentrations can be modified by multiplying by a reduction factor f where

$$ f \;=\; \exp\!\left(-\lambda_i\,\frac{x}{u_a}\right) \tag{9} $$

and λ_i is the radioactive decay constant of radionuclide i (s^{-1}). Values for λ_i are provided in Annex II.

3.9. GROUND DEPOSITION

Activity concentrations of radionuclides on the ground can be calculated simply by using ratios representing the amount of activity deposited on the ground per unit time and the ground level air concentration. These ratios are called deposition velocities or, better, deposition coefficients, because their physical meaning is not that of a velocity. Further information on wet and dry deposition coefficients is given in Refs [13, 15, 18, 21, 34, 35].

For generic assessment purposes the following relationship is used

$$\dot{d}_i = (V_d + V_w)\,C_A \qquad\qquad\qquad (10)$$

where

$\dot{d}_i$ is the total daily average deposition rate on the ground of a given radionuclide i from both dry and wet processes, including deposition either on to impervious surfaces or on to both vegetation and soil (Bq·m^{-2}·d^{-1});

V_d is the dry deposition coefficient for a given radionuclide (m/d);

V_w is the wet deposition coefficient for a given radionuclide (m/d).

As indicated in Section 5, the deposition rate $\dot{d}_i$ is used to calculate the radionuclide concentration on vegetation owing to direct contamination ($C_{v,i,1}$ in Eq. (30)) and the concentration of radionuclide in dry soil ($C_{s,i}$ in Eq. (32) or C_{gr} in Eq. (42)).

Measured values of V_d and V_w for radionuclides are quite variable. They depend on such factors as the physical and chemical form of the radionuclide, the nature of the depositing surface, meteorological conditions and, in the case of V_w, the precipitation rate [25]. It has been recommended that a total deposition coefficient, V_T (= $V_d + V_w$), of 1000 m/d be used for screening purposes for deposition of aerosols and reactive gases [8–10]. This value of V_T is consistent with values for ^{131}I and ^{137}Cs fallout resulting from the accident at the Chernobyl nuclear power station in 1986 [36]. It should be noted, however, that the Chernobyl fallout aerosols were well mixed in the atmosphere by the time the measurements were taken. Materials deposited near the point of release are likely to be less well mixed. As a result, it is expected that using $V_T = 1000$ m/d will result in a conservative estimate of total deposition for screening purposes [25]. For ^{3}H, ^{14}C and non-reactive gases such as krypton it should be assumed that $V_T = 0$ [7–10].

Deposition from a plume of material in the atmosphere may result not only in radionuclides on the ground but also in lakes in the vicinity of the release. This may occur by both direct deposition on to the lake surface and from runoff into the lake of material initially deposited on the land surface. As noted earlier, Gaussian plume procedures for calculating air concentrations are designed to be used within 20 km of the release point. If a lake is found within this radius, it is recommended that the deposition rate at the centre of the lake should be calculated, and the methods presented in Section 4 may be used to estimate the resulting water concentration.

3.10. RESUSPENSION OF DEPOSITED RADIONUCLIDES

Radionuclides on the ground can be resuspended into the air by the action of wind and other disturbances. There have been several reviews of the processes involved (see, for example, Refs [37, 38]). Resuspension mechanisms can be broadly

divided into human-made and wind driven disturbances. Resuspension caused by human-made disturbances, such as vehicular traffic, digging and farming activities, is usually localized and of potential importance only for the exposure of a few individuals. Wind driven resuspension leads to more general exposure. Resuspension has been included in previous generic calculations, for example in Ref. [4]. However, it is generally found to be a relatively minor exposure pathway following routine discharges of radionuclides to the atmosphere [39]. It is relatively important only where there is no continuing discharge and, for those radionuclides such as the actinides, where doses owing to external irradiation and food chain transfer are relatively unimportant. Resuspension is therefore not considered further here. Appropriate models are described in Ref. [39]. This pathway is, however, taken into account when assessing doses to workers exposed to sewage sludge, as described in Section 4.

3.11. ESTIMATES FOR AREA SOURCES

The models presented in this section are designed for use with releases from point sources, not area or volume sources such as from uranium mill tailings. It has been suggested, however, that the following procedures could be used to calculate the concentration in air from an area source [25].

(a)   Calculate a pseudo point source release rate by integrating over the entire area of the source,
(b)   Assume the release height to be zero,
(c)   Locate the release point at the edge of the area source nearest the location of concern.

Use of this procedure should result in a conservative estimate of the concentration in air.

3.12. UNCERTAINTY ASSOCIATED WITH THESE PROCEDURES

As indicated earlier, the methodology presented in this Safety Report is designed to minimize the possibility that the calculated doses will underestimate real doses by more than a factor of ten. The concepts presented in this section for calculating the radionuclide concentrations in air are based primarily on recommendations for models presented elsewhere [7–10]. The procedures outlined in one of these reports [8] have been carefully evaluated to determine the degree of conservatism that they represent [25]. The results of that review may be summarized as follows.

(a)   For isolated point sources the Gaussian plume model appears to be capable of predicting long term average air concentrations within a factor of four for flat terrain and within a factor of ten for sites with complex terrain and meteorology [40].

(b)   For releases near buildings and receptors within a distance equal to 2.5 times the square root of the building frontal area the models presented are generally conservative. However, these models should not be used in situations where confinement of the plume is probable, for example for releases into a street canyon.

(c)   For releases near buildings and receptors at distances larger than 2.5 times the square root of the building frontal area predicted concentrations are expected to be within a factor of two for simple building shapes and flat terrain. However, the model presented here begins to underpredict consistently for wind speeds greater than 5 m/s [41].

(d)   The default assumptions of $P_p = 0.25$, $u_a = 2$ m/s, $V_T = 1000$ m/d, plume rise = 0 and plume depletion = 0 are believed to be conservative in nature.

(e)   The bias is unknown for the assumption that class D atmospheric stability prevails 100% of the time.

4. RADIONUCLIDE TRANSPORT IN SURFACE WATERS

This section describes generic methods for estimating radionuclide concentrations in water, C_w (Bq/m^3), and in sediment, C_s (Bq/kg), from routine radionuclide discharges into surface water, Q_i (Bq/s) (Fig. 2). The following types of surface water are considered.

— Rivers,
— Estuaries,
— Coastal waters,
— Small lakes,
— Large lakes.

Radionuclides discharged into surface waters are subject to a series of physical and chemical processes that affect their transport from the source point. These processes include

— Flow processes, such as downcurrent transport (advection) and mixing processes (turbulent dispersion);

— Sediment processes, such as adsorption/desorption on suspended, shore/beach and bottom sediments, and downcurrent transport, deposition and resuspension of sediment, which adsorbs radionuclides;

— Other processes, including radionuclide decay and other mechanisms that will reduce concentrations in water, such as radionuclide volatilization (if any).

These processes are illustrated in Fig. 11 and are described in more detail in Refs [42, 43]. The processes are, in general, three dimensional and transient in nature. However, a generic assessment of annual average radionuclide concentrations from routine discharges can be made on the assumption that a number of processes are at a steady state.

Before describing the generic approach applied to each surface water body, the relationship between the discharge rate and the radionuclide concentration in water is provided for the screening assumption that all water usage occurs at the point of release. Features of the generic modelling procedure that are common to all surface water types are then outlined, followed by the generic model for each of the surface water types in turn. Uncertainty issues are discussed before finally describing a modelling approach for assessing discharges to a sewerage system.

4.1. SCREENING CALCULATIONS

As in Section 2, the simplest and most pessimistic approach for screening purposes is to neglect dilution effects — that is in effect to assume that exposure occurs at the point of discharge. This approach is independent of the type of water body into which the discharge occurs. The concentration in water under these circumstances is given by

$$C_{w,\,tot} \; = \; C_0 \; = \; \frac{Q_i}{F} \tag{11}$$

where

$C_{w,\,tot}$ is the total radionuclide concentration (Bq/m^3),
C_0 is the radionuclide concentration in the effluent discharge outfall (Bq/m^3),
Q_i is the annual average discharge rate for radionuclide i (Bq/s),
F is the flow rate of the liquid effluent (m^3/s).

A value of $C_{w,\,tot}$ calculated using Eq. (11) can be used to calculate radionuclide concentrations in sediment and aquatic foods and subsequent critical group doses (see Annex I). As indicated in Section 2, if the doses calculated in this way exceed the relevant dose criterion, a further assessment that takes account of dilution in the
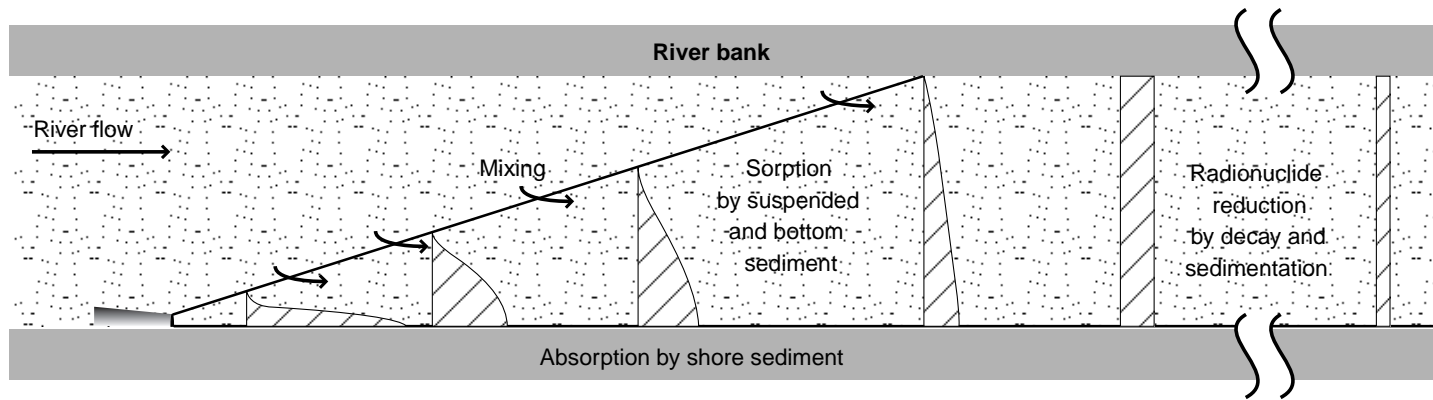
FIG. 11. Processes affecting the movement of radionuclides from the point of discharge into a river.

The figure contains the following labels:

River bank

River flow

Mixing

Sorption by suspended and bottom sediment

Radionuclide reduction by decay and sedimentation

Absorption by shore sediment

appropriate surface water body (i.e. river, estuary, coastal water, small lake or large lake) is recommended. The remainder of Section 4 provides the information necessary for such assessments.

4.2. FEATURES OF MODELS OF DILUTION IN SURFACE WATERS

There are three basic types of model used to estimate radionuclide transport in surface waters [44].

— Numerical models usually transform basic equations describing radionuclide dispersion into finite difference or finite element forms.
— Box type models treat the entire water body or sections of a water body as homogeneous compartments. These models often include some sediment–radionuclide interactions.
— Analytical models solve the basic radionuclide transport equations. Simplifying assumptions are made regarding water body geometry, flow conditions and dispersion processes in order to obtain analytical solutions to the governing equations.

The generic methodology presented in this Safety Report is based on analytical solutions to advection–diffusion equations describing radionuclide transport in surface waters with steady state uniform flow conditions. A box type model has been used for small lakes and reservoirs. The detailed derivation of these solutions and associated assumptions is discussed in Annex VI.

Radionuclide concentrations in water (and sediment) may be calculated for specific locations where members of hypothetical critical groups could use this water for drinking, fishing, irrigation or swimming, and could use the sediment for recreational or agricultural activities. These locations (at a specified distance x from the point of discharge) are selected to represent the nearest point where water usage is conceivable during the projected lifetime of an operating facility. The methodology also includes simple processes to estimate default dispersion coefficients, river flow conditions and the coastal current, if site specific values are unavailable.

If there is more than one source of radionuclide discharge, it is recommended that the radionuclide concentration for each of the discharges be calculated separately. The cumulative radionuclide concentration at location x may then be obtained by summing the results for the individual radionuclides.

4.2.1. Sediment effects

Radionuclides may become adsorbed on to sediments in water, thus reducing the dissolved radionuclide concentration in the water column. However, radionuclides

adsorbed on to suspended sediment may still migrate downstream, deposit to the river/sea bottom and then be resuspended from the bottom and become incorporated in banks and beaches [45]. These processes are highly site specific. Over a period of time, remobilization of radionuclides associated with deposited sediments may become an additional source of exposure to members of critical population groups, the importance of which depends upon the radionuclide involved.

It is possible to calculate radionuclide concentrations in sediment, corresponding to the concentration in water, by using a distribution coefficient K_d (L/kg). However, it is recommended that the effect of sediment adsorption be ignored, for the sake of simplicity, for screening purposes and because this approach will tend to overestimate radionuclide concentrations in water and doses from direct uses of water (e.g. drinking). This approach is followed here, although data required to take account of sediment effects are presented in Section 4.7.

4.2.2. Applicability and limitations of the models

As a result of the simplifying assumptions implicit in its derivation, this generic methodology strictly applies only if the following conditions are satisfied.

(a)   The surface water geometry (e.g. river cross-section, shoreline) does not change greatly with distance;
(b)   The flow characteristics (e.g. flow velocity, water depth) do not change significantly with distance or with time;
(c)   Radionuclides in water and sediment, under the conditions of a routine, long term release, can be considered to be in equilibrium.

This approach is often used to assess the distribution of contaminants in surface waters [44, 46–48]. Annex VI provides more details and general solutions to the equations describing radionuclide transport.

4.2.2.1. Conservatism

A degree of conservatism (i.e. the tendency to overestimate actual concentrations) is necessary for such a generic approach, and in this methodology this is derived from the following conditions.

(1)   The locations where surface waters or sediments are assumed to be utilized by hypothetical critical group members are intentionally selected to limit the potential for underestimating actual exposures.
(2)   Values for flow rates, current velocity and water depth are representative of the lowest annual average conditions occurring over a period of 30 years.

(3)   Radionuclide concentrations are calculated along the plume centreline (except for concentrations along a coastal shoreline).

(4)   Radionuclides are assumed to be released along the bank of a river, estuary or large lake, thus restricting mixing.

(5)   Additional conservatism can be achieved by excluding the effect of sediment adsorption (thus maximizing the estimate of the concentrations of radionuclides in the water for estimation of doses resulting from the direct use of the water).

4.3. RIVERS

The model has been developed to allow the radionuclide concentration in water to be calculated at a location along either the river bank from which the discharge occurs or along the opposite bank (see Fig. 11). It has been designed to require a minimum input of site specific data. As indicated above, an initial estimate of the radionuclide concentration in water can be made by neglecting sediment effects. However, the approach needed to take account of the effects of water filtration and sedimentation is described in Section 4.7.

4.3.1.   Basic river characteristics required for calculations

The following three parameters are required to calculate radionuclide concentrations in a river (corresponding to the lowest annual river flow rates expected in a 30 year period).

(a)   River width B (m),
(b)   Longitudinal distance from the release point to a potential receptor location x (m),
(c)   Radionuclide decay constant λ_i (s^{-1}).

Site specific values for the following are desirable: 30 year low annual river flow rate q_r (m^3/s); flow depth D (m), that corresponds to q_r; river velocity U (m/s), that corresponds to q_r.

If such data are not available, they may be estimated, on the basis of hydrological studies [49], as indicated in Section 4.3.1.1.

The longitudinal distance for complete mixing is based on the assumption that complete lateral and vertical mixing is achieved when the minimum concentration is one half of the maximum concentration along the same lateral and vertical lines. The longitudinal distance required to achieve this vertical complete mixing, L_z (m), is

$$L_z = 7D$$

4.3.1.1. Estimating a default value for the river flow rate

From observation or a map estimate a river width $\bar{B}$ (m) under normal river flow conditions upstream of the estuary, where there is no tidal effect on the river flow.

The mean annual river flow rate $\bar{q}_r$ (m³/s) that corresponds to the river width $\bar{B}$ may be obtained from Table III. For default purposes it may be assumed that the 30 year low annual river flow rate q_r (m³/s) is 1/3 of the mean annual river flow rate $\bar{q}_r$. The river width B (m) and depth D (m) that corresponds to the 30 year low annual river flow rate q_r (m³/s) may then be obtained from Table III.

The net freshwater velocity U (m/s) corresponding to the 30 year low annual river flow rate may be calculated by taking

$$U = \frac{q_r}{BD}$$

4.3.2. Calculation of radionuclide concentrations

The procedure for calculating radionuclide concentrations is illustrated in the form of a flow chart in Fig. 12. Two situations are considered below.

4.3.2.1. Water usage on the river bank opposite to the radionuclide discharge point

In this case a radionuclide must traverse at least half of the river width to reach the bank opposite to the discharge point, and the maximum radionuclide concentration is the cross-sectionally averaged concentration. Thus the radionuclide concentration in unfiltered water at the opposite bank can be obtained by

$$C_{w,\,tot} = \frac{Q_i}{q_r} \exp\left(-\frac{\lambda_i x}{U}\right) = C_t \tag{12}$$

where

$C_{w,\,tot}$ is the total radionuclide concentration in water (Bq/m³),
Q_i is the average discharge rate for radionuclide i (Bq/s),
q_r is the mean river flow rate (m³/s),
λ_i is the radioactive decay constant (s⁻¹),
x is the distance between the discharge point and the receptor (m),
U is the net freshwater velocity (m/s).

Values for radionuclide decay constants (λ_i) are provided in Annex II.

TABLE III. RELATIONSHIPS BETWEEN RIVER FLOW RATE, RIVER WIDTH AND DEPTH[a]

River flow rate, q_r (m³/s)	River width, B (m)	Depth, D (m)	River flow rate, q_r (m³/s)	River width, B (m)	Depth, D (m)
0.1	3.47	0.058	100	83.2	1.28
0.2	4.77	0.079	200	114	1.74
0.3	5.75	0.095	300	138	2.09
0.4	6.56	0.108	400	157	2.37
0.5	7.27	0.120	500	174	2.62
0.6	7.91	0.130	600	190	2.84
0.7	8.49	0.139	700	204	3.05
0.8	9.02	0.148	800	216	3.24
0.9	9.53	0.156	900	229	3.41
1	10.0	0.16	1 000	240	3.57
2	13.8	0.22	2 000	330	4.87
3	16.6	0.27	3 000	398	5.84
4	18.9	0.30	4 000	454	6.64
5	21.0	0.34	5 000	503	7.34
6	22.8	0.36	6 000	547	7.96
7	24.5	0.39	7 000	587	8.53
8	26.0	0.41	8 000	624	9.05
9	27.5	0.44	9 000	659	9.54
10	28.8	0.48	10 000	692	10.0
20	39.7	0.63	20 000	952	13.6
30	47.8	0.75	30 000	1 150	16.3
40	54.6	0.85	40 000	1 310	18.6
50	60.5	0.94	50 000	1 450	20.5
60	65.8	1.02	60 000	1 580	22.3
70	70.6	1.09	70 000	1 690	23.9
80	75.1	1.16	80 000	1 800	25.3
90	79.2	1.22	90 000	1 900	26.7
			100 000	2 000	28.0

[a] Using linear interpolation between values.

4.3.2.2. Water usage on the same river bank as the radionuclide discharge point

If water usage occurs on the same bank as the discharge, and at a location before complete vertical mixing can occur (i.e. $x \leq L_z = 7D$), the radionuclide concentration in water may be assumed to be undiluted; that is it is assumed to be the

```
                    ┌─────────────────┐
                    │    Select x     │
                    └────────┬────────┘
                             │
                         ╱───┴───╲
                        ╱ Do you  ╲
                       ╱ know low   ╲         No
                      ⟨ river flow   ⟩─────────────────┐
                       ╲ conditions? ╱                 │
                        ╲(qr, U, B, D)╱                 │
                         ╲───┬───╱                      │
                             │                          │
                             │                    ┌─────┴──────────────────────┐
                             │                    │ Use six steps:             │
                             │                    │  • Estimate B̄              │
                             │                    │  • Get qr from Table III    │
                             │ Yes                │  • Calculate qr = qr/3      │
                             │                    │  • Get B from Table III     │
                             │                    │  • Get D from Table III     │
                             │                    │  • Calculate U = qr/BD      │
                             │                    └─────┬──────────────────────┘
                             │                          │
                             └──────────────┬───────────┘
                                            │
                                        ╱───┴───╲
                                       ╱  Is x   ╲        No
                                      ⟨ on the same ⟩──────────────┐
                                       ╲ bank side? ╱              │
                                        ╲───┬───╱                  │
                                            │ Yes                  │
                 No                 ┌───────┴───────┐              │
         ┌──────────────────────────│    x > 7D?    │              │
         │                          └───────┬───────┘              │
         │                                  │ Yes                  │
         │                    ┌─────────────┴──────────────┐       │
         │                    │ • Calculate A = 1.5Dx/B²    │       │
         │                    │ • Obtain Pr from Table IV   │       │
         │                    └─────────────┬──────────────┘       │
  ┌──────┴──────┐          ┌────────────────┴────────┐  ┌──────────┴──────────┐
  │  C = C0     │          │ C = Qi/qr exp(-λi x/U)Pr │  │ C = Qi/qr exp(-λi x/U)Pr │
  └─────────────┘          └─────────────────────────┘  └─────────────────────┘
```
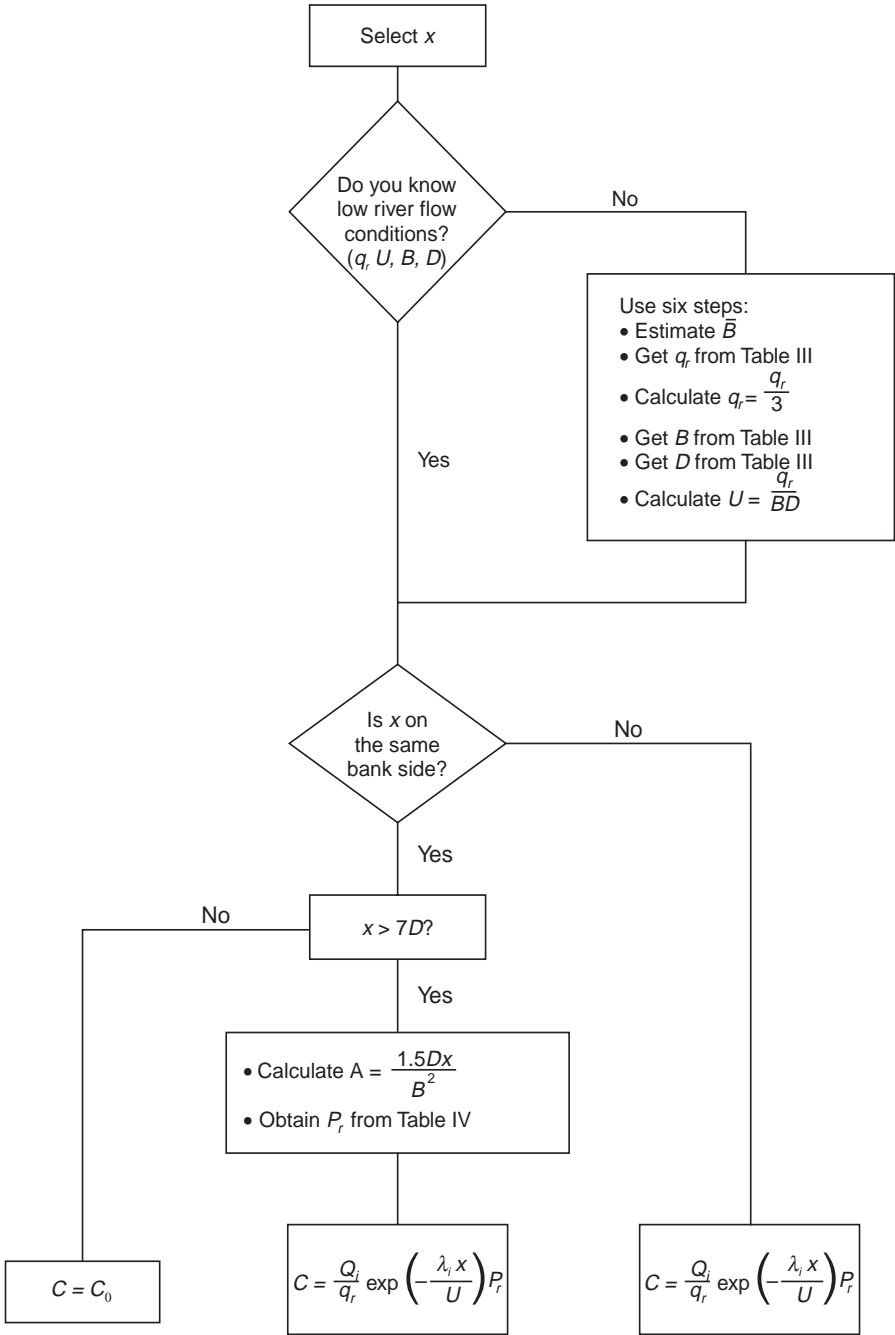
FIG. 12. Procedure for calculating radionuclide concentrations in water resulting from a discharge into a river.

37

same as the concentration at the discharge outlet. Thus $C_{w, tot} = C_0$ from Eq. (11). It is highly unlikely that drinking water would ever be extracted from a location so close to the discharge point. However, concentrations at or close to the discharge point may be valid for calculating concentrations in some aquatic biota (e.g. seaweed and mollusca).

If water usage occurs on the same bank as the discharge and at a location after which complete vertical mixing occurs (i.e. $x > L_z = 7D$), the calculated radionuclide concentration must be modified to take into account the fact that lateral mixing may be incomplete. Table IV provides values for the river partial mixing coefficient P_r for various values of the partial mixing index A, where A is given by

$$A = \frac{1.5Dx}{B^2} \tag{13}$$

The radionuclide concentration at a downstream distance x along the river bank is then

TABLE IV. RIVER PARTIAL MIXING CORRECTION FACTOR $P_r{}^a$

A^b	P_r	A	P_r	A	P_r	A	P_r
1×10^{-6}	31.0	1×10^{-4}	20.9	1×10^{-2}	10.7	1	2.6
2	29.8	2	19.4	2	9.3	2	2.0
3	28.9	3	18.5	3	8.5	3	1.7
4	28.2	4	17.8	4	7.9	4	1.5
5	27.6	5	17.4	5	7.5	5	1.4
6	27.2	6	17.1	6	7.2	6	1.3
7	26.9	7	16.7	7	6.9	7	1.3
8	26.7	8	16.4	8	6.6	8	1.2
9	26.4	9	16.1	9	6.3	9	1.1
1×10^{-5}	26.1	1×10^{-3}	15.9	1×10^{-1}	6.0	10	1.0
2	24.8	2	14.2	2	4.8	20	1.0
3	23.6	3	13.3	3	4.2	30	1.0
4	22.9	4	12.8	4	3.7	40	1.0
5	22.5	5	12.2	5	3.4	50	1.0
6	22.1	6	11.8	6	3.2	60	1.0
7	21.6	7	11.5	7	3.0	70	1.0
8	21.3	8	11.2	8	2.8	80	1.0
9	21.1	9	11.0	9	2.7	90 and greater	1.0

^{a} Using linear interpolation between values.
^{b} Index.

$$C_{w,\,tot} = C_t\,P_r \tag{14}$$

Note that the fully mixed radionuclide concentration C_t is given by Eq. (12).

The variable P_r can be regarded as a correction factor for the partial mixing, and it approaches unity as the downstream distance x increases. (Note that for $x > 3\ B^2/D$, $P_r \approx 1$, as discussed in Annex VI.)

4.4. ESTUARIES

4.4.1. Estuarine regions

An estuary is a water body that is connected at one end to a river and at the other end to the sea. An estuary velocity reverses with the tide, and an estuary can contain fresh or saline water, although it is generally less saline than that of the sea. For the purposes of this generic methodology a radioactive discharge is assumed to occur from one of the estuarine banks. The radionuclide concentration at the banks may be assessed using a methodology that is very similar to that for rivers, but with some adjustments to account for tidal effects. (See Annex VI for a detailed description of the estuarine methodology.)

4.4.2. Basic estuarine characteristics required for calculation

The following variables and parameters are required to calculate the radionuclide concentration in an estuary: estuarine width B (m); estuarine flow depth D (m); river width $\bar{B}$ (m) under a mean annual river flow rate upstream of the tidal flow area tidal period T_p (s); longitudinal distance from the release point to a potential receptor location x (m); and radionuclide decay constant λ_i (s^{-1}).

The longitudinal distances at which complete vertical mixing is achieved is assumed to occur when the minimum concentration is at least half of the maximum concentration along the vertical direction. The longitudinal distance required to achieve this complete vertical mixing L_z (m) is

$$L_z = 7D$$

The radionuclide upstream travel distance L_u (m) is calculated by

$$L_u = 0.32\,\big|U_f\big|\,T_p \tag{15}$$

39

4.4.2.1. Estimating a default value for the river flow rate and tidal velocities

From observation or a map the river width $\bar{B}$ (m) under normal river flow conditions upstream of the estuary, where there is no tidal effect on the river flow, may be estimated.

The mean river flow rate $\bar{q}_r$ (m³/s) that corresponds to the river width $\bar{B}$ may be obtained from Table III. For default purposes it may be assumed that the 30 year low annual river flow rate $\bar{q}_r$ (m³/s) is 1/3 of the mean annual river flow rate q_r. The net freshwater velocity U (m/s) corresponding to the 30 year low annual river flow rate may be calculated by taking

$$U = \frac{q_r}{BD}$$

If the maximum ebb (seaward) velocity U_e (m/s) and the flood (upstream) velocity U_f (m/s) is not available, it is possible to assume

$U_e = 0.5$ m/s
$U_f = 0.5$ m/s

The mean tidal flow speed U_t and the tidal flow rate q_w may then be estimated by taking

$$U_t = 0.32(|U_e| + |U_f|), \quad q_w = DBU_t$$

4.4.3.   Calculation of radionuclide concentrations

The procedure for calculating radionuclide concentrations in water at a location along the banks of an estuary is illustrated in the form of a flow chart in Fig. 13. Four situations are considered as described.

4.4.3.1. Water usage on the bank of the estuary opposite to the radionuclide discharge point

The maximum radionuclide concentration on the opposite bank is expected to be the cross-sectionally averaged concentration; thus the radionuclide concentration in this region is calculated by

$$C_{w,\,tot} = \frac{Q_i}{q_w} \exp\left(-\frac{\lambda_i x}{U}\right) = C_{te} \tag{16}$$
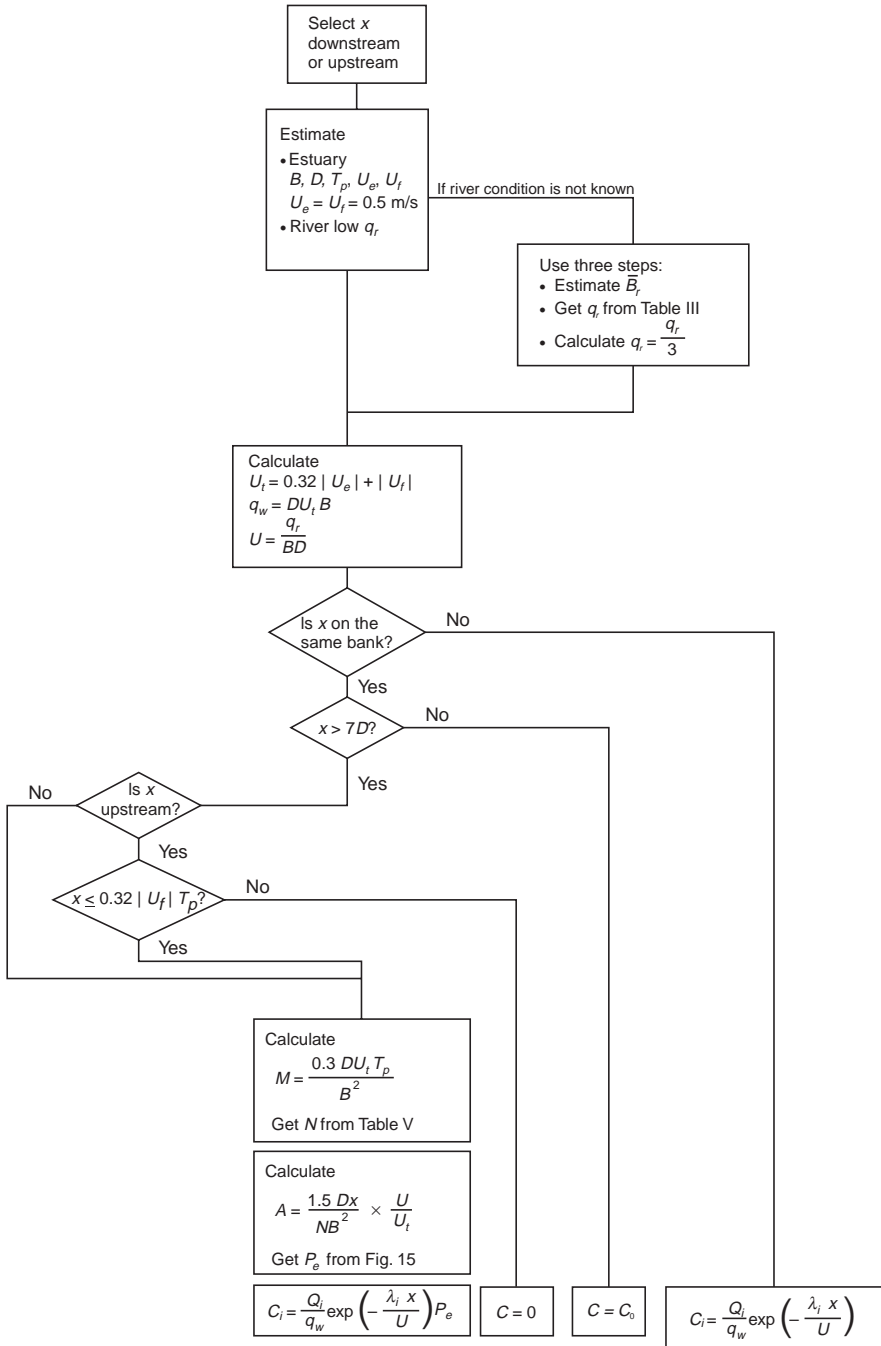
FIG. 13. Procedure for calculating radionuclide concentrations in water resulting from discharge into an estuary.

where

$C_{w,\text{tot}}$ is the total radionuclide concentration in water (Bq/m^3),
Q_i is the average discharge rate for radionuclide i (Bq/s),
U is the net freshwater velocity (m/s).

4.4.3.2. Water usage upstream or downstream prior to complete mixing

If water usage occurs at either upstream or downstream locations prior to complete vertical mixing (i.e. $|x| \leq L_z = 7D$), the radionuclide concentration in this case is assumed to be

$$C_{w,\text{tot}} = C_0$$

where C_0 is the radionuclide concentration at the point of discharge (Eq. (11)) in Bq/m^3.

4.4.3.3. Water usage upstream at a distance greater than L_u

In a case where water usage occurs upstream at a distance greater than L_u (i.e. $|x| > L_u = 0.32 |U_f| T_p$), tidal flow cannot reach the water use location during the flood tide (when the estuarine flow moves landward). Thus

$$C_{w,\text{tot}} = 0$$

4.4.3.4. Water usage upstream at a distance less than L_u or downstream at a distance greater than L_z

If water usage occurs upstream but within the distance L_u (i.e. $7D < |x| \leq L_u = 0.32 |U_f| T_p$) or if water usage occurs downstream beyond complete vertical mixing (i.e. $x > 7D$), the estimated radionuclide concentration should be modified to allow for partial mixing. In this case upstream and downstream dispersions are treated as the same. Annex VI describes the upstream concentration correction, which will enable a reader to improve the accuracy of the radionuclide concentration estimate in the upstream area.

The estuarine calculation procedure adopted here is similar to, although a little more complicated than, that described for rivers because tidal effects must be taken into account. The procedure is illustrated in Fig. 13. The parameters required are as follows.

As indicated in Ref. [50], the ratio M of the tidal period (T_p) to the timescale for cross-sectional mixing is

$$M = \frac{0.3DU_tT_p}{B^2} \tag{17}$$

where T_p is the tidal period in seconds (4.5×10^4 s for tides occurring twice per day or 9×10^4 s for a predominant tide occurring once per day).

The ratio of the longitudinal dispersion coefficient in the estuary to that in a river N corresponding to M may be obtained from Table V. The partial mixing index A is then

$$A = \frac{1.5Dx}{NB^2} \times \frac{U}{U_t} \tag{18}$$

The mixing coefficient P_e corresponding to A may then be determined from Fig. 14. If P_e is less than unity it is recommended that $P_e = 1$ be assumed.

The radionuclide concentration at upstream or downstream distance x along the estuarine bank with the default longitudinal and lateral dispersion coefficients is obtained by

$$C_{w,\,tot} = C_{te}P_e \tag{19}$$

TABLE V. RATIO N OF THE LONGITUDINAL DISPERSION COEFFICIENT BETWEEN AN ESTUARY AND A RIVER[a]

M	Ratio N	M	Ratio N	M	Ratio N
0.01	0.00028	0.1	0.0240	1.0	0.705
0.02	0.00115	0.2	0.0900	2.0	0.910
0.03	0.00237	0.3	0.167	3.0	0.940
0.04	0.00427	0.4	0.267	4.0	0.950
0.05	0.00640	0.5	0.350	5.0	0.970
0.06	0.00930	0.6	0.430	6.0	0.980
0.07	0.0122	0.7	0.544	7.0	0.990
0.08	0.0152	0.8	0.610	8.0	0.995
0.09	0.0205	0.9	0.640	9.0 and greater	1.00

[a] Using linear interpolation between values.
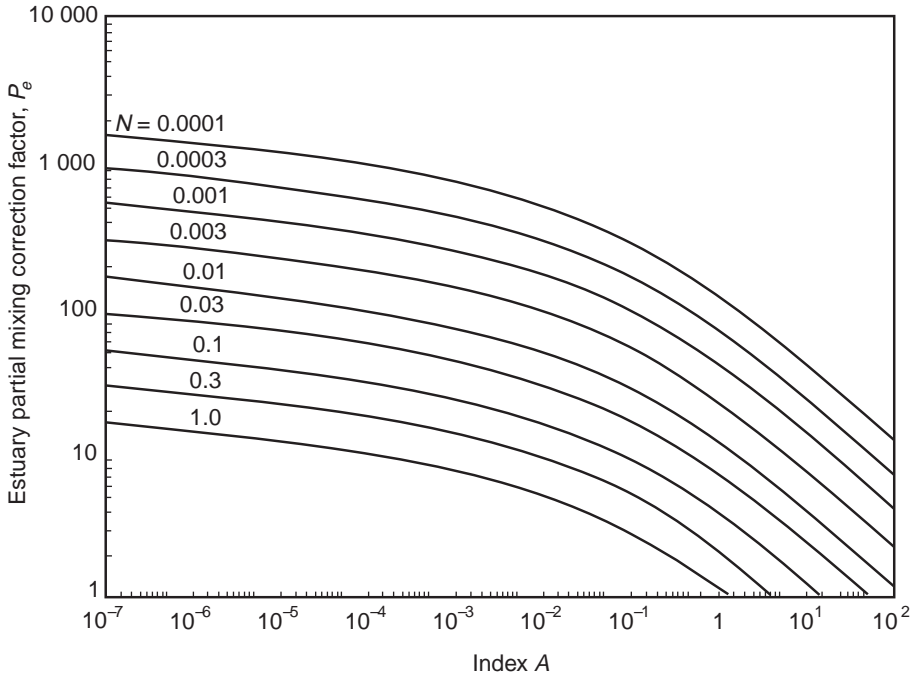
FIG. 14. Relationship between the estuary partial mixing coefficient and index A.

where C_{te} is obtained from Eq. (16).

As for the river, C_{te} is the completely mixed radionuclide concentration over an estuarine cross-section. The variable P_e can be regarded as a correction factor for partial mixing and approaches unity as the downstream distance x increases. (Note that for $x > 0.6B^2/D$, $P_e \approx 1$ as discussed in Annex VI.)

4.5. COASTAL WATERS

4.5.1. Coastal region modelling approach

The mathematical model selected for coastal waters is based on a steady state, vertically averaged advection–diffusion equation. Further details are provided in Annex VI. Note that the scale of the mixing length becomes larger as a plume spreads further in the coastal water [51, 52]; the lateral dispersion coefficient in this methodology changes with downcurrent distance.

4.5.2.   Basic coastal water characteristics

The following model variables and parameters are required for estimation of the radionuclide concentration in coastal waters: water depth D (m) at the radionuclide discharge effluent outfall; distance between the release point and the beach y_0 (m); longitudinal distance (along the coastal current direction) from the release point to a potential receptor location x (m); and radionuclide decay rate λ_i (s^{-1}).

Note that for significantly stratified coastal waters, vertical mixing is inhibited most of the time. Thus for releases above or below the boundary between layers of water that differ significantly in temperature or salinity, the water depth above or below this boundary should ideally be used. Otherwise, the average total water depth may be used.

A default coastal current of $U = 0.1$ m/s may be used when site specific information is not available.

4.5.3.   Radionuclide concentration estimate

The methodology presented here is applicable to downcurrent areas satisfying the following conditions (illustrated in Fig. 15).

$$7D \ < \ x$$

$$\left| \frac{y \ - \ y_0}{x} \right| \ \ll \ 3.7$$

A radionuclide concentration $C_{w,\,tot}$ for fishing may be estimated by

$$C_{w,\,tot} = \ \Big|_{y \,=\, y_0} = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \exp\!\left(-\frac{\lambda_i x}{U}\right) \tag{20}$$

where Q_i is the average discharge rate of radionuclide i (Bq/s).

If a fishing site cannot be determined then, as a default, x may be assumed to be 50 times the water depth.

To calculate a radionuclide concentration along the shoreline, the following equation may be used.

$$C_{w,\,tot} = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \exp\!\left[\frac{(-7.28 \times 10^5) U^{2.34} y_0^2}{x^{2.34}}\right] \exp\!\left(-\frac{\lambda_i x}{U}\right) \tag{21}$$

A flow chart of the screening methodology for coastal waters is shown in Fig. 16.
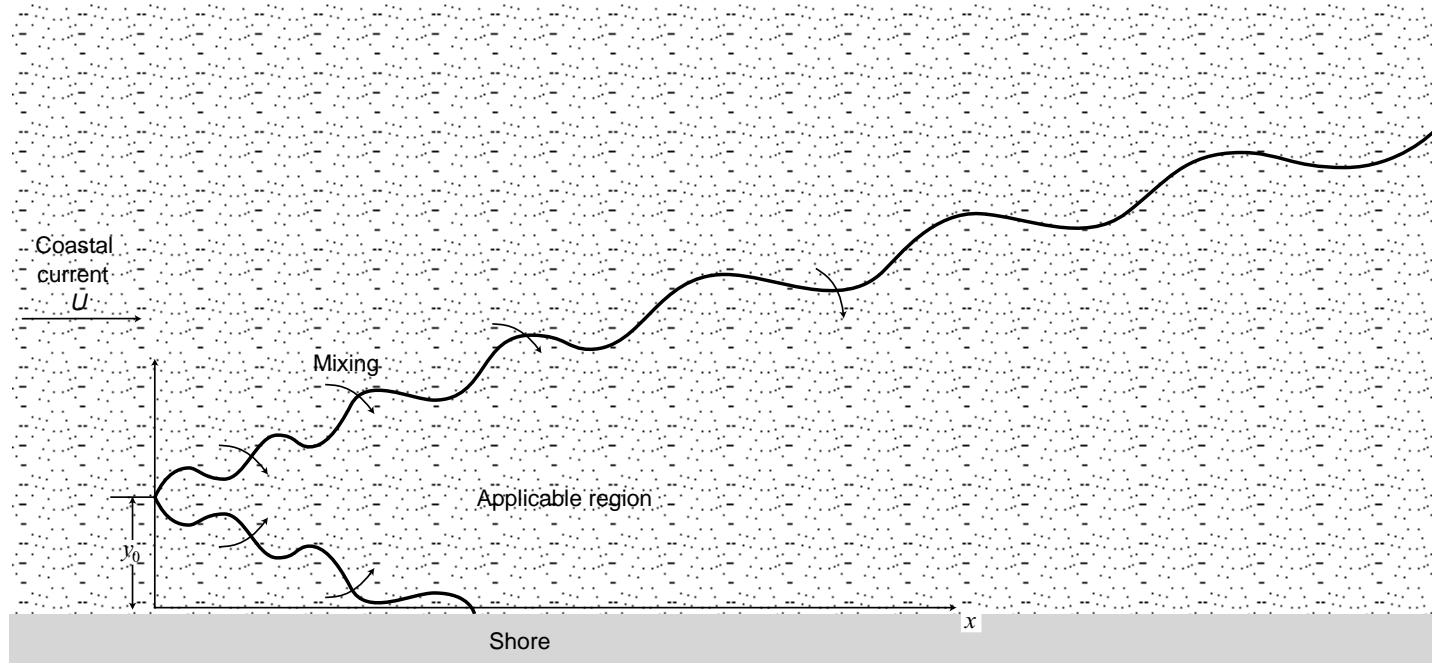
FIG. 15. Conditions under which the coastal waters methodology is applicable.

Estimate
D, U, y_0

Select x
- For fishing
- For shellfish
- For shoreline

Shoreline:
$$C = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \times \exp \frac{(-7.28 \times 10^5)\, U^{2.34}\, y_0^{\,2}}{x^{2.34}} \times \exp\left(-\frac{\lambda_i x}{U}\right)$$

Shellfish and fish:
$$C = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \exp\left(-\frac{\lambda_i x}{U}\right)$$
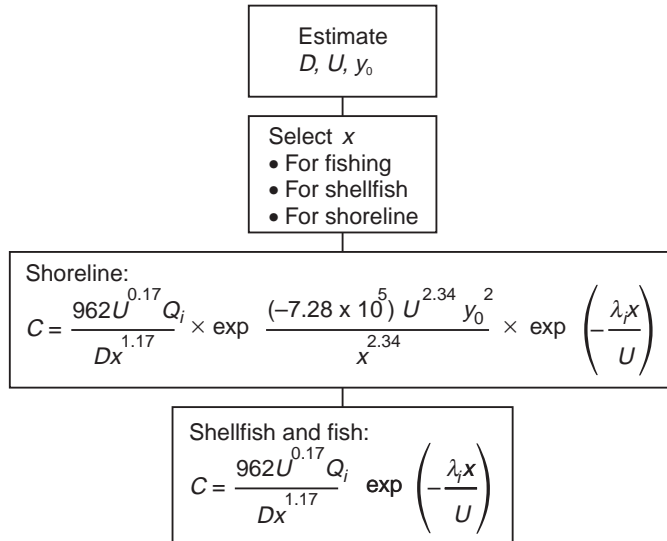
FIG. 16. Procedure for calculating radionuclide concentrations in water resulting from discharges into coastal waters.

4.6. LAKES AND RESERVOIRS

4.6.1. Classification

For assessment purposes lakes and reservoirs are divided into two types: large and small. Large lakes are those that have significant wind induced flow and a water surface area of at least 400 km^2. Since a person might be expected to be able to see across a distance of around 20 km, as a rough rule a lake can be considered to be large when the opposite side of the lake is not visible to a person standing on a 30 m high shore.

4.6.2. Small lakes and reservoirs

For a small lake or reservoir the radionuclide concentration is assumed to be uniform throughout (see Fig. 17).

4.6.2.1. Required parameters

In order to estimate the radionuclide concentration in a small lake or reservoir, the following parameters must be known.
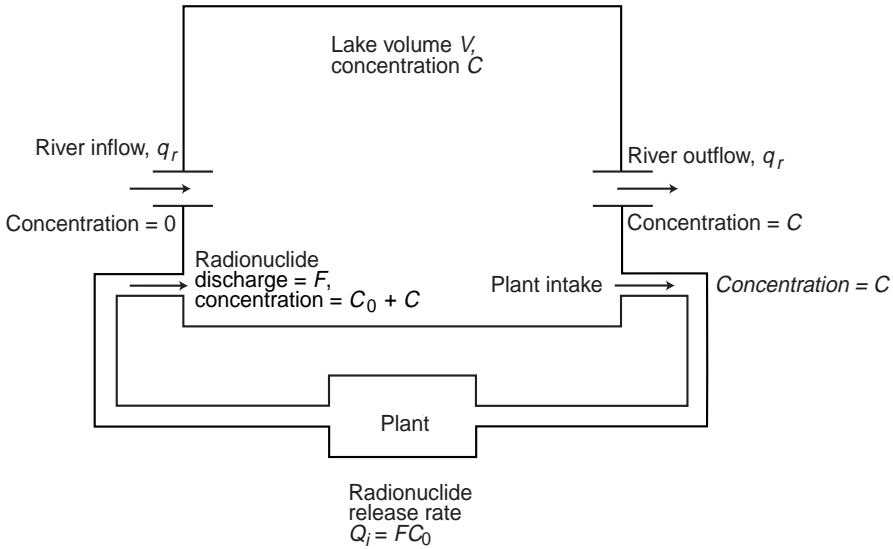
FIG. 17. Schematic diagram showing assumed conditions of release into a small lake or reservoir.

— The 30 year low annual river flow rate into and out of the lake q_r (m³/s) or the river width (m) under a mean annual river flow rate;
— The lake surface area A_l (m²);
— The lake volume V (m³) or average lake depth D (m);
— The deposition rate from the atmosphere $\dot{d}$ (Bq·m⁻²·d⁻¹) as described in Section 3;
— The expected life of the facility discharging the radionuclide effluent into the lake, $t = T$ (e.g. 30 years) (s);
— The radionuclide decay constant λ_i (s⁻¹).

If the 30 year low annual river flow rate into and out of the lake is not known, it may be estimated as described earlier for rivers and estuaries: the river width, under normal river flow conditions, is first estimated by direct observation or from a map. The mean annual river flow rate $\bar{q}_r$ (m³/s) corresponding to that river width may be taken from Table III. The 30 year low annual river flow rate q_r (m³/s) may be estimated as 1/3 of $\bar{q}_r$ (m³/s).

4.6.2.2. Radionuclide concentration estimate

If there is atmospheric deposition of radionuclides on to a small lake and its watershed, the calculation of radionuclide concentrations in the lake should account

48

for this contribution, in addition to the liquid discharge directly to the lake. To account for this contribution, it is assumed that the size of the lake watershed is 100 times the lake surface area, and that 2% of a radionuclide deposited on to the watershed reaches the lake through runoff, surface soil erosion and groundwater seepage. Thus the combined radionuclide release rate Q'_i (Bq/s) is

$$Q'_i = Q_i + \frac{3\dot{d}A_l}{86\,400} \tag{22}$$

where Q_i is the annual average rate of radionuclide discharged directly into the lake. The atmospheric deposition rate can be estimated as the product of the deposition velocity and the radionuclide concentration in air above the lake, as discussed in Section 3.

Assuming that $C_{w,\,\text{tot}} = 0$ at time $t = 0$, the radionuclide concentration in a small lake or reservoir is given by

$$C_{w,\,\text{tot}} = \frac{Q'_i}{q_r + \lambda_i V}\left\{1 - \exp\left[-\left(\frac{q_r}{V} + \lambda_i\right)t\right]\right\} \tag{23}$$

If

$$\left(\frac{q_r}{V} + \lambda_i\right) > 10^{-8}\,\text{s}^{-1}$$

then the radionuclide concentration becomes steady (not time dependent) and is given by

$$C_{w,\,\text{tot}} = \frac{Q'_i}{q_r + \lambda_i V} \tag{24}$$

The flow chart of the screening methodology for a small lake is shown in Fig. 18.

4.6.3. Large lakes

Large lakes are those with a large residence time (V/q_r), and with lake flows dominated by wind induced currents (e.g. the Great Lakes in North America). The radionuclide transport in these large lakes is controlled mainly by wind induced flow, stratification and seasonal turnover, and scale dependent mixing, similar to the situation in coastal waters [48]. However, even in a large lake complete mixing over
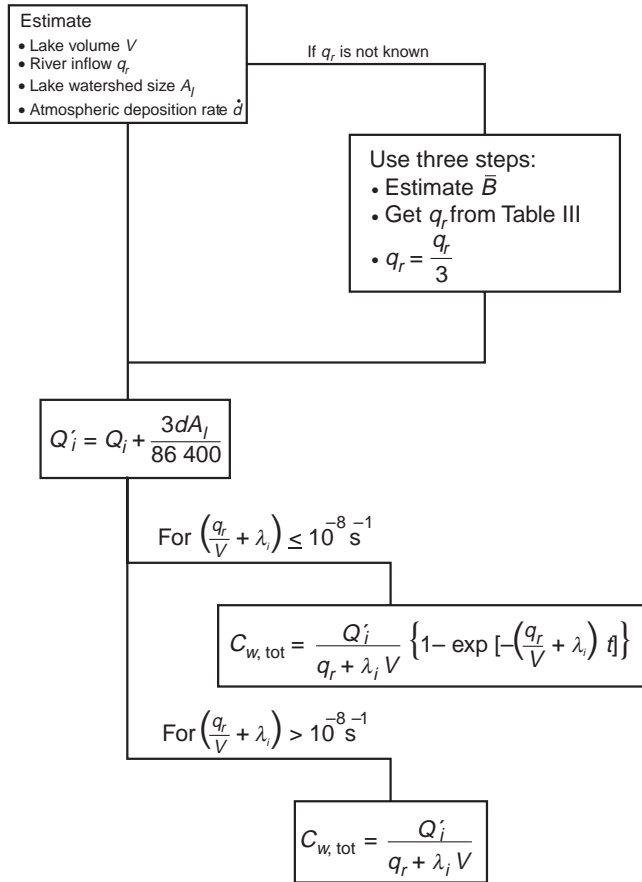
FIG. 18. Procedure for calculating radionuclide concentrations in water resulting from discharge into a small lake or reservoir.

the entire lake can be achieved in a relatively short period of time (much less than one year) owing to a very large dispersion coefficient that changes with distance and seasonal turnover [47]. However, near the release point complete mixing is unlikely to occur. Thus the long term radionuclide concentration is calculated by using the methodology for partial mixing described in Section 4.4.

4.6.3.1. Required parameters

Model parameters required for a large lake include those described in Section 4.4.3.

— The water depth D (m) at the radionuclide discharge effluent outfall,
— The distance between the release point in the lake and the beach y_0 (m),
— The longitudinal distance (along the lake current direction) from the release point to a potential receptor location x (m),
— The radionuclide decay constant λ_i (s^{-1}).

4.6.3.2. Default lake flow velocity

If a site specific lake flow velocity is not available, a default value of $U = 0.1$ m/s may be used.

4.6.3.3. Radionuclide concentration estimates

The radionuclide concentration is calculated using Eqs (20) and (21) for partial mixing in lake water. The unfiltered radionuclide concentration along the radionuclide plume centreline may be obtained by

$$C_{w, \text{tot}} = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \exp\left(-\frac{\lambda_i x}{U} \right)$$

A radionuclide concentration along the shoreline is calculated by

$$C_{w, \text{tot}} = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \exp\left[\frac{(-7.28 \times 10^5) U^{2.34} y_0^2}{x^{2.34}} \right] \exp\left(-\frac{\lambda_i x}{U} \right)$$

As for coastal waters, this methodology is subject to the following conditions being satisfied

$$7D < x$$

$$\left| \frac{y - y_0}{x} \right| << 3.7$$

A flow chart of the screening methodology for a large lake is shown in Fig. 19.

51

Estimate
D, U, y_0

Select x
- For fishing
- For shellfish
- For shoreline exposure
- For drinking water
- For irrigation

Shoreline and drinking and irrigation water:

$$C = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \times \exp \frac{(-7.28 \times 10^5) U^{2.34} y_0^2}{x^{2.34}} \times \exp\left(-\frac{\lambda_i x}{U}\right)$$

Shellfish and fish:

$$C = \frac{962 U^{0.17} Q_i}{D x^{1.17}} \times \exp\left(-\frac{\lambda_i x}{U}\right)$$
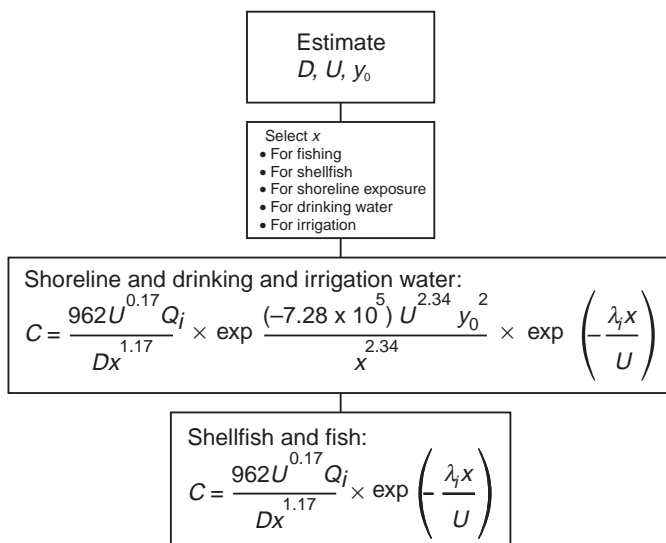
FIG. 19. Procedure for calculating radionuclide concentrations in water resulting from discharge into a large lake.

4.7. SEDIMENT EFFECTS

4.7.1. Sorption and retention

When sediment interacts with radionuclides dissolved in water, the concentration of the radionuclides in the dissolved phase may be decreased owing to radionuclide adsorption on to sediment particles. Consequently, the concentration of radionuclides on suspended sediment and the banks and bed of the water body will be increased owing to adsorption and particle settling. Figure 20 shows the effects of sediment adsorption and suspended sediment concentration on the radionuclide concentration.

The distribution coefficient K_d (L/kg) is used to express the exchange of radionuclides between the dissolved and sediment sorbed phases and is defined for a given radionuclide as

$$K_d = \frac{\text{sediment sorbed radionuclide concentration per unit weight of sediment (Bq/kg)}}{\text{dissolved radionuclide concentration per unit volume of water (Bq/L)}} \tag{24}$$
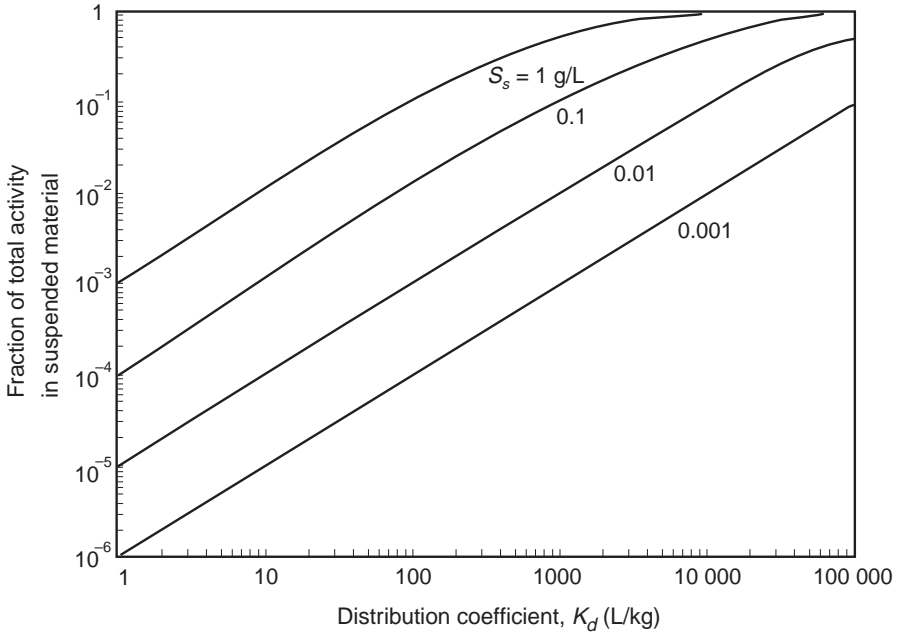
FIG. 20. Relationship between suspended sediment concentrations (S_s), distribution coefficients (K_d) and the fraction of activity adsorbed on to suspended material.

4.7.2. Radionuclide concentration in water

The dissolved (filtered) radionuclide concentration (Bq/m^3) in surface water can be obtained by

$$C_{w,s} = \frac{C_{w,\,tot}}{1 + 0.001\,K_d S_s} \tag{25}$$

where S_s is a suspended sediment concentration (kg/m^3 or g/L).

Note that the 0.001 in the denominator of Eq. (25) is the unit conversion of K_d from L/kg to m^3/kg. It is, however, recommended that calculations for screening assessments of doses from drinking water, fish and shellfish be based on the total concentration in water (including sediments). In this case sediment effects should be taken into account only for the purpose of calculating doses arising from exposure to sediment.

The radionuclide concentration $C_{w,\,tot}$ is estimated as described in Sections 4.3–4.6 for the appropriate type of surface water body.

A selection of K_d values associated with suspended sediment in fresh water and salt water is provided in Table VI [42, 45]. Values of K_d are affected by sediment type, water quality and other conditions [42], and can vary by several orders of magnitude for each radionuclide. It is therefore important to take care to select a value that is appropriate for the specific site under consideration. If a site specific K_d value is not available, default values from Table VI may be used. Usually, the finer the sediment, the higher the K_d value for a given radionuclide under the same water quality conditions [53, 54]. The suspended sediment concentration will vary widely depending on the characteristics of the water body. Ideally, a site specific value for S_s should be used, but in the absence of such data the following default values may be used: estuary, $S_s = 5 \times 10^{-2}$ kg/m³; river/lake, $S_s = 5 \times 10^{-2}$ kg/m³; coastal sea, $S_s = 1 \times 10^{-2}$ kg/m³.

4.7.3. Radionuclide concentration in suspended sediment

The radionuclide concentration $C_{s,w}$ (Bq/kg) adsorbed by suspended sediment can be obtained by

$$C_{s,w} = \frac{0.001 K_d C_{w,\text{tot}}}{1 + 0.001 S_s K_d} = 0.001 K_d C_{w,s} \tag{26}$$

When surface water is used for drinking, suspended sediment is removed by water treatment processes. Thus most of the radionuclide adsorbed on suspended sediment is removed from drinking water, although the efficiency of removal varies with radionuclide and with the specific water treatment process used. The radionuclide concentration in suspended sediment may be important if suspended sediment is used as a source of surface soil.

4.7.4. Radionuclide concentration in bottom sediment

Bottom sediment can contain radionuclides owing to deposition of suspended sediment, on which radionuclides are adsorbed, and to direct adsorption on to bottom sediments of dissolved radionuclides from overlying water [45]. Field data [53, 54] suggest much smaller K_d values associated with bottom sediment than with suspended sediment. This is partly due to the presence of relatively coarser sediment at the bottom and the greater abundance of bed sediment than of suspended sediment. A notable exception to this is in the null zone of an estuary, where sea water and fresh water merge [57]. The apparent K_d value for bottom sediment is generally assumed to be one tenth of the K_d value associated with suspended sediment. This assumption is still likely to overestimate the K_d values of the bottom sediment, with the result that, in the event of this bottom sediment being dredged for use in a landfill, such a K_d

TABLE VI. RECOMMENDED SCREENING VALUES FOR K_d (L/kg) FOR ELEMENTS IN NATURAL FRESHWATER AND MARINE ENVIRONMENTS, WITH EMPHASIS ON OXIDIZING CONDITIONS

Element	Screening values for K_d (L/kg)	
	Fresh water [a]	Salt water [b]
Ac		2×10^6
Ag		1×10^3
Am	5×10^3	2×10^6
As		
At		
Au		
Ba		5×10^3
Bi		
Br		
C	5	2×10^3
Cd		2×10^3
Ce	1×10^4	2×10^6
Cm	5×10^3	2×10^6
Co	5×10^3	2×10^5
Cr	1×10^4 [c]	5×10^4
Cs	1×10^3	3×10^3
Cu		
Eu	5×10^2	5×10^5
Fe	5×10^3	5×10^4
Ga		
H	0 [c]	1
Hg		1×10^4
I	10	2×10^1
In		1×10^5
Mn	1×10^3	2×10^5
Mo		
Na		1
Nb		5×10^5
Ni		1×10^5
Np	10	5×10^3
P	50 [c]	1×10^2 [c]
Pa		1×10^6
Pb		2×10^5
Pd		5×10^4
Pm	5×10^3	2×10^6

TABLE VI. (cont.)

Element	Screening values for K_d (L/kg)	
	Fresh water [a]	Salt water [b]
Po		2×10^7
Pu	1×10^5	1×10^5
Ra	500	5×10^3
Rb		
Rh		
Ru	500^c	3×10^2
S		5×10^{-1}
Sb	50^c	1×10^3
Se		1×10^3
Sr	1×10^3	1×10^3
Tc	5	1×10^2
Te		1×10^3
Th	1×10^4	2×10^6
Tl		2×10^4
U	50	1×10^3
Y		1×10^7
Zn	500	2×10^4
Zr	1×10^3	1×10^6

[a] Values for freshwater sediments were taken from Ref. [6], unless otherwise indicated.
[b] Values for salt water correspond to those for coastal sediment given in Ref. [55], unless otherwise indicated.
[c] Additional values provided by Y. Onishi [56].

value would lead to a relatively pessimistic dose estimation. Radioactive decay during accumulation of the radionuclide on the river bottom is taken into account in calculating the radionuclide concentration in the bottom sediment, $C_{s,b}$ (Bq/kg), as follows

$$C_{s,b} = \frac{(0.1)(0.001)K_d C_{w,\,\mathrm{tot}}}{1 + 0.001 S_s K_d} \times \frac{1 - e^{-\lambda_i T_e}}{\lambda_i T_e} = 0.1 C_{s,w} \times \frac{1 - e^{-\lambda_i T_e}}{\lambda_i T_e} \qquad (27)$$

where T_e is the effective accumulation time (s).

To give a conservative estimate of $C_{s,b}$ a default value of 3.15×10^7 s (1 year) is recommended for the effective accumulation period T_e. This value was selected to account for the typical situation in which the bottom sediments migrate downstream

(with the exception of a small lake situation without much river inflow) or are buried under more recently deposited layers of sediment. In effect, the sediment layer of concern is the most recently deposited material, which has had least opportunity for radioactive decay to occur.

4.7.5. Radionuclide concentration in shore/beach sediment

The surface activity concentration of a radionuclide in shore/beach sediment (Bq/m^2), taking account of radioactive decay occurring while the radionuclide is accumulating in shore or beach sediment, is assumed to be

$$C_{s,s} = \frac{(0.1)(0.001)K_d \times 60 \times C_{w,\text{tot}}}{1 + 0.001 S_s K_d} \times \frac{1 - e^{-\lambda_i T_e}}{\lambda_i T_e} = 60 C_{s,b} \tag{28}$$

where T_e is the effective accumulation time (s).

The factor of 60 (kg/m^2) takes account of the top layer (assumed to be 5 cm) and bulk density of sediment. A default value of T_e of 3.15×10^7 s (1 year) may be used to provide a conservative estimate of $C_{s,s}$.

4.8. UNCERTAINTY

Uncertainty in this methodology arises from possible departures of the real situation from the assumptions outlined above. Although analytical solution based methods such as the current methodology are widely used, rigorous testing to quantify uncertainty under all potential conditions of model application has not been conducted. The following discussion, nevertheless, gives an indication of the uncertainty to be expected in applying the methodology.

For accurate estimates in partially mixed zones of the water body the surface geometry and flow characteristics must be relatively constant. These assumptions are usually satisfied. However, if marked changes in geometry or flow characteristics occur between a radionuclide release point and a receptor location, one can perform bounding calculations to estimate upper and lower values of radionuclide concentrations using extreme values for the changing characteristics of flow and geometry. After complete mixing is achieved, changes in surface geometry and flow characteristics are no longer critical.

The sediment adsorption calculation is a potential source of significant error. The ratio of a sediment sorbed radionuclide to a dissolved radionuclide is the product of a distribution coefficient K_d (L/kg) and one 1000th of the suspended sediment concentration S_s $(kg/m^3$ or $g/L)$. If this product is small, say much less than 1, then the error in estimating the amount of sediment adsorption is not great. However, if this

product is much larger than 1, then site specific values of K_d and S_s should be obtained to reduce the potential error in the radionuclide concentration estimate. Figure 20 shows the effects of suspended sediment concentration and the distribution coefficient on radionuclide concentrations. This figure demonstrates that in most surface water cases the potential for the suspended sediment to reduce the dissolved radionuclide concentration is minor. However, for some radionuclides, such as plutonium and caesium, the sediment effects may be very important [53]. If it is necessary to evaluate radionuclide distributions with the dynamic effects of sediment–radionuclide interactions (e.g. radionuclide adsorption/desorption, and transport, deposition and resuspension of sediment sorbed radionuclides and variable flow conditions), numerical models of these interactions are required [45, 58].

4.9. RADIONUCLIDES DISCHARGED TO SEWERS

In many cases radionuclides are discharged along with other wastes into the sewerage system. In order to estimate the subsequent radiation doses, two extreme possible scenarios may be postulated.

(a)   Assume that no radioactive material is retained in the sewage sludge, but that it is all discharged to the water body in liquid form;
(b)   Assume that all of the radioactive material discharged is retained in the sewage sludge at the sewage treatment plant.

It is recommended that both situations be considered and the scenario giving the higher estimated dose be used for screening purposes.

For case (a) the radionuclide concentrations in water and sediment may be calculated as described earlier in this section for discharges into surface water. Subsequent individual radiation doses may be calculated as outlined in Section 6. Case (b) is considered here.

The concentration of a radionuclide in sewage sludge, assuming complete transfer of discharged activity to the sludge, is given by

$$C_{\text{sludge}} = Q_{\text{sludge}} / S_{\text{sludge}} \qquad (29)$$

where

C_{sludge} is the annual average concentration of the radionuclide in the sludge (Bq/kg),
Q_{sludge} is the annual discharge of the radionuclide (Bq/a),
S_{sludge} is the annual sewage sludge production at the relevant sewage treatment plant (kg/a).

The annual sewage sludge production at the relevant sewage treatment plant will vary depending on the size of the treatment plant and the size of the population it serves. In the United Kingdom and Austria a person produces an average of about 25–30 kg/a (dry weight) of sewage [59]. This estimate includes both domestic and industrial sewage (domestic sewage is about 15 kg/a). For screening purposes a default annual sewage production of 20 kg per person per year (dry weight) is recommended. A default value for S_{sludge} (the annual average) can then be obtained by multiplying this value by the number of people served by the plant. As an example, a typical plant serving 20 000 people would treat about 400 t/a (dry weight). The exposure pathways arising from sewage sludge are outlined in Section 6 of this Safety Report. These exposure pathways generally relate to sewage in its wet state. It is therefore necessary to ensure that sludge concentrations relate to wet sludge. Approximately 5% of sewage sludge is comprised of solid material [60], such that the wet weight concentration is $0.05 \times$ concentration (dry weight).

5. TRANSPORT OF RADIONUCLIDES THROUGH TERRESTRIAL AND AQUATIC FOOD CHAINS

Ingestion of radionuclides in foods can be an important contributor to the total dose received by an individual or population group. An estimate of the radionuclide concentration is needed to assess such doses. This section describes a generic methodology to calculate the concentration of radionuclides in human food crops C_v (Bq/kg) and animal produce, C_m (Bq/L) for milk and C_f (Bq/kg) for meat, resulting from either an air concentration C_A (Bq/m^3) or a ground deposition rate $\dot{d}_i$ (Bq·m^{-2}·d^{-1}) estimated from Section 3, or from a water concentration C_W (Bq/m^3) estimated from Section 4 (Fig. 2). For the purpose of generic assessments the models and default parameters described in this section consider the following processes either explicitly or implicitly.

(a) Deposition by dry or wet processes;
(b) Initial interception and retention by vegetation surfaces;
(c) Translocation to the edible tissues of vegetation;
(d) Post-deposition retention by vegetation and soil surfaces;
(e) Uptake by roots;
(f) Adhesion of soil particles on to vegetation surfaces;
(g) Direct ingestion of surface soil by humans or grazing animals;
(h) Transfer of radionuclides in soil, air, water and vegetation into the milk and meat of grazing animals;

(i)   Transfer of radionuclides in surface water to the terrestrial system by spray irrigation;

(j)   Transfer of radionuclides in surface water to sediment and to aquatic biota.

In the interest of simplifying the model, a number of these processes are taken into account by the use of composite parameters that describe the effect of two or more interacting processes. The default parameter values listed in this section have been selected expressly for the purpose of performing generic calculations of doses for critical groups. As a result, they may differ significantly from those given in other reviews and in earlier IAEA handbooks, which are intended to be 'best estimate' values [4]. The user is referred to IAEA Technical Report No. 364 [6] for a more detailed discussion of the sources of data that support the parameter values selected in this section.

5.1. TERRESTRIAL FOOD CHAIN MODELS

The terrestrial food chain models are designed to accept an input of radionuclides from either the atmosphere (Section 3) or the hydrosphere (Section 4). These models include those transfer processes that are likely to be important for the radiological assessment of routine discharges.

5.1.1. Concentrations in vegetation

Radionuclides intercepted by and retained on vegetation may result from fallout, washout, rainout, irrigation with contaminated water or deposition of resuspended matter. External deposits can be taken up by foliar absorption into plants. Radionuclides may also be incorporated by uptake from the soil through roots, followed by internal redistribution of radionuclides within the plant. Processes that may lead to the reduction of radionuclide concentrations in vegetation include radioactive decay, growth dilution, wash-off of externally deposited radionuclides, leaching and soil fixation. Further removal of radioactive material from vegetation may occur due to grazing, harvesting, etc.

For conditions of prolonged deposition, such as from discharges, the following equation may be used to estimate the concentration $C_{v,i,1}$ due to direct contamination of nuclide i in and on vegetation

$$C_{v,i,1} = \frac{\dot{d}_i \alpha \left[1 - \exp\left(-\lambda_{E_i^v} t_e \right) \right]}{\lambda_{E_i^v}} \qquad (30)$$

where

$C_{v,i,1}$ is measured in Bq/kg dry matter for vegetation consumed by grazing animals and in Bq/kg fresh matter for vegetation consumed by humans.

$\dot{d}_i$ is the deposition rate (from wet and dry processes) of radionuclide i on to the ground (Bq·m^{-2}·d^{-1}, calculated from Eq. (10)).

α is the fraction of deposited activity intercepted by the edible portion of vegetation per unit mass (or mass interception factor, m^2/kg) as the result of both wet and dry deposition processes; for pasture forage the unit of mass is conventionally given in terms of dry weight, and for fresh vegetables the unit is in wet weight.

$\lambda_{E_i^v}$ is the effective rate constant for reduction of the activity concentration of radionuclide i from crops (d^{-1}), where $\lambda_{E_i^v} = \lambda_i + \lambda_w$.

t_e is the time period that crops are exposed to contamination during the growing season (d).

λ_w is the rate constant for reduction of the concentration of material deposited on the plant surfaces owing to processes other than radioactive decay (d^{-1}).

λ_i is the rate constant for radioactive decay of radionuclide i (d^{-1}).

The radionuclide concentration in vegetation arising from indirect processes — from uptake from the soil and from soil adhering to the vegetation — is

$$C_{v,i,\ 2} = F_v \times C_{s,i} \tag{31}$$

where

$C_{v,i,\ 2}$ is measured in Bq/kg dry matter for vegetation consumed by grazing animals and in Bq/kg fresh matter for vegetation consumed by humans.

F_v is the concentration factor for uptake of the radionuclide from soil by edible parts of crops (Bq/kg plant tissue per Bq/kg dry soil). It is conservatively assumed that all activity removed from the atmosphere becomes available for uptake from the soil; in addition, the selected values also implicitly take account of the adhesion of soil to the vegetation (again, for pasture forage the unit of mass is for dry matter; for vegetation consumed by humans the unit is for fresh weight).

$C_{s,i}$ is the concentration of radionuclide i in dry soil (Bq/kg).

$C_{s,i}$ is defined by

$$C_{s,i}\ =\ \frac{\dot{d}_i\left[1 - \exp\left(-\lambda_{E_i^s} t_b\right)\right]}{\rho\lambda_{E_i^s}} \tag{32}$$

where

$\lambda_{E_i^s}$ is the effective rate constant for reduction of the activity concentration in the root zone of soils (d^{-1}), where $\lambda_{E_i^s} = \lambda_i + \lambda_s$;

λ_s is the rate constant for reduction of the concentration of material deposited in the root zone of soils owing to processes other than radioactive decay (d^{-1});

t_b is the duration of the discharge of radioactive material (d);

ρ is a standardized surface density for the effective root zone in soil (kg/m^2, dry soil).

Eq. (32) refers to the total deposit and neglects the amount which is adsorbed to the vegetation. The total concentration of the radionuclide on the vegetation at the time of consumption is

$$C_{v,i} = (C_{v,i,1} + C_{v,i,2}) \exp(-\lambda_i t_h) \tag{33}$$

where

$C_{v,i}$ is measured in Bq/kg dry matter for vegetation consumed by grazing animals and in Bq/kg fresh matter for vegetation consumed by humans,

λ_i is the rate constant for radioactive decay of radionuclide i (d^{-1}),

t_h is a delay (hold-up) time that represents the time interval between harvest and consumption of the food (d).

For sprinkler irrigation of cultivated areas Eqs (30) and (32) may be used again. For Eq. (30) the deposition rate is given as

$$\dot{d}_i = C_{w,i} \times I_w \tag{34}$$

where $C_{w,i}$ is the concentration of nuclide i in the water (Bq/m^3) and I_w is the average irrigation rate (m^3·m^{-2}·d^{-1}) over the period of irrigation. For Eq. (32) the deposition rate $\dot{d}_i$ from Eq. (34) is multiplied by the fraction of the year over which the irrigation takes place, in order to provide an annual average irrigation rate. Thus deposition on plant surfaces is calculated with the average irrigation rate over the irrigation period, while deposition on soil is calculated with the average irrigation rate over the entire year. The actual rate of irrigation could be much higher than an average rate; the use of the appropriate average value will tend to overestimate the activity accumulated on vegetation or on surface soil, since at high irrigation rates foliar retention will be decreased and surface runoff increased [6, 61].

5.1.1.1. Direct deposition on to plant surfaces

Equation (30) relates to the direct deposition of radionuclides on vegetation from the deposition of activity on aerial parts of the plant from either atmospheric or hydrospheric (sprinkler irrigation) sources. The deposition rate from an atmospheric source may be estimated as described in Section 3.9, while the deposition rate from irrigation can be derived from the water concentration and irrigation rate, as described in Section 5.1.1 and Eq. (34). The mass interception factor for forage plants α_1 and the mass interception factor for food crops α_2 are specified separately. This is primarily because vegetation consumed by animals is usually specified on a dry weight basis, while vegetation consumed by humans is most frequently reported as fresh weights; the interception factors reflect this difference. Default values of the mass interception fractions α_1 and α_2 are listed in Table VII. These values implicitly include the effect of translocation of the radionuclides from foliage to the edible tissues of the vegetation.

5.1.1.2. Reduction of radionuclide concentrations from surfaces of vegetation

The radionuclide concentration in vegetation may be reduced by a variety of processes. These include radioactive decay, wash-off of previously intercepted material by rain or irrigation, surface abrasion and leaf bending from the action of the wind, resuspension, tissue ageing, leaf fall or herbivore grazing, addition of new tissue (growth dilution), volatilization or evaporation [6]. Losses other than radioactive decay are normally described using an aggregated parameter in the form of a first order rate constant λ_w. A default value of λ_w is given in Table VII for estimating the removal of radionuclides from vegetation based on a half-life of 14 days. Default values for the crop exposure period (growing season) t_e, the time period between harvest and consumption t_h and the period of soil exposure t_b during the operating lifetime of a nuclear installation are presented in Table VIII.

5.1.1.3. Deposition on soil

Radionuclides can be deposited on soil either by direct deposition from the atmosphere or from the use of surface water for irrigation. When there is a canopy of vegetation radionuclides can reach the soil through leaf fall, leaching, wash-off, cuticular sloughing, herbivore excretion and trampling [6]. For the purpose of generic calculations it is conservatively assumed that the total deposition per unit area reaches the soil surface, regardless of the extent of canopy cover and subsequent harvesting of vegetation. As indicated earlier, the application of Eq. (32) for irrigation may lead to severe overestimates. This applies particularly for irrigation with water containing

TABLE VII.   CONSERVATIVE VALUES FOR MASS INTERCEPTION AND ENVIRONMENTAL REMOVAL RATES FROM PLANT SURFACES

	Parameter	Default value
α_1	Mass interception factor for forage vegetation (dry weight)	3 m²/kg
α_2	Mass interception factor for food crops (wet weight)	0.3 m²/kg
λ_w	Environmental removal rate for all plant surfaces	0.05 d⁻¹

TABLE VIII.   CONSERVATIVE VALUES FOR CROP AND SOIL EXPOSURE PERIODS AND DELAY TIMES

Parameter	$t_e(1)$	$t_e(2)$	$t_h(1)$	$t_h(2)$	$t_h(3)$	t_b	t_m	t_f
Days	30	60	0	90	14	1.1×10^4	1	20

$t_e(1)$ applies to forage grasses.

$t_e(2)$ applies to food crops.

$t_h(1)$ applies to forage.

$t_h(2)$ applies to stored feed for animals.

$t_h(3)$ applies to food crops.

t_b refers to a 30 year exposure period (the assumed operating lifetime of a facility).

t_m is the average time between collection and human consumption of milk.

t_f is the average time between slaughter and human consumption of meat.

anionic radionuclides. Consultation with an expert would be needed for a more precise evaluation.

In this generic model discharges and subsequent deposition on the soil are assumed to continue for a period t_b of 30 years. The extent to which this time period affects the radionuclide concentration in soil depends on the half-life of the radionuclide and the rate at which the radionuclide becomes unavailable for uptake. The estimated radionuclide concentration in soil is also affected by the assumed depth of soil over which the surface deposition is averaged and by the density of the soil. The density of the soil depends on soil type and extent of soil compaction. For generic calculations default values of effective surface soil densities ρ (kg/m²) are given for two categories each of depth and soil type in Table IX. Actual values of ρ may vary depending on the origin, mineral content and classification of the soil [35, 62]. However, uncertainties about soil density are relatively small.

TABLE IX. EFFECTIVE SURFACE SOIL DENSITY FOR SCREENING PURPOSES

Rooting zone depth	Effective surface soil density, ρ (kg/m^2 dry weight soil)	
	Peat soils	Other soils
Pasture: 0–10 cm	50	130
All other crops: 0–20 cm	100	260

From Ref. [62] based on soil densities of 500 kg/m^3 for peat soils and 1300 kg/m^3 for other soils.

5.1.1.4. *Reduction of radionuclide concentration in the soil surface*

Initial concentrations of radionuclides deposited on the soil surface will be reduced by soil erosion, mixing with uncontaminated soil, ploughing, surface runoff and downward migration by leaching, as well as by physical radioactive decay. Another important phenomenon is the incorporation of radionuclides into the soil particle matrix. This is especially important for caesium and strontium. The concentration of radionuclides in soil may also be reduced by uptake through roots and subsequent removal of plants by harvesting or consumption. For generic calculations all of the above processes (with the exception of radioactive decay) are represented by the loss rate constant λ_s (d^{-1}). In practice, the value λ_s is highly dependent on climate, agricultural management practices, soil type, vegetative cover and the chemical form of the radionuclide.

For the purpose of generic assessment a distinction is made between anionic radionuclides, isotopes of strontium and caesium, and all other radionuclides. Anions such as TcO_4^-, Cl^- and I^- are leached quickly; the default value for λ_s is 0.5 a^{-1} (0.0014 d^{-1}). For strontium and caesium the default value is 0.05 a^{-1} (0.00014 d^{-1}). For all other nuclides (also non-anionic Tc) the default value is zero (Table X).

5.1.1.5. *Uptake from soil by edible portions of vegetation and the implicit assumption of inadvertent soil ingestion*

In most radiological assessment models the radionuclide concentration in vegetation resulting from radionuclides in soil is described by the process of root uptake [63, 64], which is typically represented by the plant/soil concentration ratio B_v [6, 64]. In the present procedure, soil adhesion is combined with the uptake, resulting in a concentration ratio F_v (Bq/kg vegetation per Bq/kg dry weight soil). A distinction is made between pasture forage and human food crops (Table XI). $F_{v,1}$ is for pasture

TABLE X. LOSS RATE CONSTANT VALUES FOR SCREENING PURPOSES

Nuclide	λ_S (d^{-1})
Anions such as TcO_4^-, Cl^- and I^-	0.0014[a]
Sr and Cs	0.00014
All other nuclides (including non-anionic forms of Tc and I)	0

[a] These values apply only if the anions remain in this form. Both TcO_4^- and I^- are easily reduced in soil. Cl^- is never reduced and remains anionic.

vegetation or animal feed (Bq/kg dry weight plant per Bq/kg dry weight soil), while $F_{v,2}$ is for fresh food crops consumed by humans (Bq/kg fresh weight plant per Bq/kg dry weight soil). To account implicitly for soil adhesion, a minimum value of 0.1 is assigned to $F_{v,1}$ and 0.001 to $F_{v,2}$. These minimum values are derived as explained in Section 5.1.1.6.

Explicit calculation of soil ingestion by humans and grazing animals will depend on numerous site specific assumptions that are beyond the scope of a generic assessment. For this, the user is referred to Ref. [6].

5.1.1.6. Derivation of minimum values for $F_{v,1}$ and $F_{v,2}$

$F_{v,1}$ (min) was obtained by assuming (1) that the concentration on the soil surface of undisturbed pasture is five times higher than the concentration averaged over the 10 cm effective root zone; (2) that 6% of a grazing animal's diet is composed of surface soil; and (3) that radionuclides ingested with soil are one third as available for assimilation in the animal as are radionuclides incorporated into plant tissue [65–67]. Thus

$$F_{v,1} \text{ (min)} = 5 \times 0.06 \times 0.33 \approx 0.1$$

$F_{v,2}$ (min) was obtained by assuming that 0.1% of the human consumption of fresh vegetables is composed of soil (2×10^{-4} kg/d dry soil/ 0.2 kg/d = 1×10^{-3}). The radionuclide concentration at the soil surface may be as much as three times higher than the average calculated for the 20 cm effective root zone for food crops. This high surface concentration is offset by the assumption that assimilation of radionuclides in soil by humans is one third that for radionuclides in vegetation. Thus

$$F_{v,2} \text{ (min)} = (10^{-3} \times 3)/ 3 = 1 \times 10^{-3}$$

TABLE XI. ELEMENT SPECIFIC TRANSFER FACTORS FOR TERRESTRIAL
FOODS FOR SCREENING PURPOSES

Element	$F_{v, 1}$ forage (dry weight)[a]	$F_{v, 2}$ crops (fresh weight)[b]	F_m milk (d/L)	F_f meat (d/kg)
Ac	0.1	0.001	2×10^{-6}	2×10^{-5}
Ag	0.1	0.01	1×10^{-4}	0.006
Am	0.1	0.002	2×10^{-5}	1×10^{-4}
Ar	0	0	0	0
As	0.2	0.08	1×10^{-4}	0.02
At	0.9	0.2	0.01	0.01
Au	0.4	0.1	1×10^{-5}	0.005
Ba	0.1	0.05	0.005	0.002
Bi	0.5	0.1	0.001	0.002
Br	2	0.4	0.02	0.05
C[c]				
Cd	5	0.5	0.02	0.001
Ce	0.1	0.05	3×10^{-4}	2×10^{-4}
Cm	0.1	0.001	2×10^{-6}	2×10^{-5}
Co	2	0.08	0.01	0.07
Cr	0.1	0.001	2×10^{-4}	0.09
Cs	1[d]	0.04[d]	0.01[e]	0.05[e]
Cu	2	0.5	0.002	0.01
Eu	0.1	0.002	6×10^{-5}	0.002
Fe	0.1	0.001	3×10^{-4}	0.05
Ga	0.1	0.003	1×10^{-5}	3×10^{-4}
H[c]				
Hg	3	0.3	5×10^{-4}	0.01
I	0.1	0.02	0.01[e]	0.05
In	0.1	0.003	2×10^{-4}	0.004
Kr	0	0	0	0
Mn	10	0.3	3×10^{-4}	7×10^{-4}
Mo	1	0.2	0.005	0.01
Na	0.6	0.05	0.25	0.8
Nb	0.2	0.01	4×10^{-6}	3×10^{-6}
Ni	1	0.3	0.2	0.05
Np	0.5	0.04	5×10^{-5}	0.01
P	10	1	0.02	0.05
Pa	0.1	0.01	5×10^{-6}	5×10^{-6}
Pb	0.1	0.02	3×10^{-4}	7×10^{-4}
Pd	0.5	0.1	1×10^{-4}	2×10^{-4}
Pm	0.1	0.002	6×10^{-5}	0.002

TABLE XI. (cont.)

Element	$F_{v, 1}$ forage (dry weight)[a]	$F_{v, 2}$ crops (fresh weight)[b]	F_m milk (d/L)	F_f meat (d/kg)
Po	0.1	0.002	0.003	0.005
Pu	0.1	0.001	3×10^{-6}	2×10^{-4}
Ra	0.4	0.04	0.001	0.005
Rb	2	0.2	0.1	0.03
Rh	2	0.2	5×10^{-4}	0.002
Ru	0.2	0.05	3×10^{-5}	0.05
S	6	0.6	0.02	0.2
Sb	0.1	0.001	2.5×10^{-4}	0.005
Se	1	0.1	0.001	0.1
Sn	1	0.3	0.001	0.01
Sr	10	0.3	0.003	0.01
Tc	80	5	0.001	0.001
Te	10	1	0.005	0.07
Th	0.1	0.001	5×10^{-6}	1×10^{-4}
Tl	2	2	0.003	0.02
U	0.2	0.01	6×10^{-4}	0.003
Xe	0	0	0	0
Y	0.1	0.003	6×10^{-5}	0.01
Zn	2	2	0.01	0.2
Zr	0.1	0.001	6×10^{-6}	1×10^{-5}

[a] $F_{v, 1} = $ (Bq/kg plant dry weight)/(Bq/kg soil dry weight).
[b] $F_{v, 2} = $ (Bq/kg plant fresh weight)/(Bq/kg soil dry weight).
[c]   Models for tritium and carbon are covered separately in Annex III.
[d]   The Cs soil to plant transfer is distinctly influenced by the type of ecosystem. For soils with a pH below 4, tundras, taigas, uplands and tropical systems, the following values should be used: $F_{v, 1} = 20$, $F_{v, 2} = 0.3$.
[e]   Values for small animals (goat, sheep) are higher than those for cattle. F_m (Cs) = 0.1, F_f (Cs) = 0.3, F_m(I) = 0.5.

5.1.2.   Concentrations in animal feed

The concentration of radionuclide i in animal feed is calculated by

$$C_{a,i} = f_p C_{v,i} + (1 - f_p) C_{p,i} \tag{35}$$

where

$C_{a,i}$ is the concentration of radionuclide i in the animal feed (Bq/kg, dry matter);

$C_{v,i}$ is the concentration of radionuclide i for pasture, calculated using Eqs (30)–(33) with $t_h = 0$ (Bq/kg, dry matter);

$C_{p,i}$ is the concentration of radionuclide in stored feeds (Bq/kg, dry weight), calculated using Eqs (30)–(33), and substituting $C_{p,i}$ for $C_{v,i}$, with $t_h = 90$ d;

f_p is the fraction of the year that animals consume fresh pasture vegetation (dimensionless).

5.1.3. Intake of radionuclides by animals and transfer to milk and meat

The intake of radionuclides by animals depends on animal species, mass, age and growth rate of the animal, the digestibility of feed and, in the case of lactating animals, the milk yield [6]. For generic calculations grazing animals are assumed to be cattle which, during the grazing season, are on a diet of fresh pasture only. The grazing season depends on latitude, and ranges from a few months to the whole year.

Sources of radionuclides considered for intake by animals are fresh or stored forage and drinking water. The intake of radionuclides with soil has been implicitly included in the soil–plant uptake factor F_v. Default values of pasture and water consumption for generic milk and meat producing animals are given in Table XII.

The default values for animal intake have been based on values for dairy and beef cattle. They are conservative, however, and would not be expected substantially to underestimate concentrations of radionuclides in milk or meat of other animals. Uptake factors from feed to milk and meat (F_m and F_f, respectively) are provided in Table XI. For caesium and iodine special uptake factors are supplied for small animals such as sheep and goats. If the latter factors are used, the food supply quantities Q (see Table XII) also must be adapted. (Note: it is incorrect to change only the food supply quantities Q. Whenever this is done, the values for F_m and F_f must also be adapted [6].)

5.1.3.1. Concentration in milk

The concentration of a radionuclide in milk depends directly on the radioactivity concentration of the feed consumed by the lactating animal. With the value of $C_{a,i}$ calculated with Eq. (35), and the concentration in drinking water $C_{w,i}$ calculated in Section 4, the concentration of radionuclide i in milk is estimated as

$$C_{m,i} = F_m \left(C_{a,i} Q_m + C_{w,i} Q_w \right) \exp \left(-\lambda_i t_m \right) \tag{36}$$

TABLE XII. ANIMAL INTAKES OF WATER AND DRY MATTER AND THE FRACTION OF THE YEAR THAT ANIMALS CONSUME FRESH PASTURE

Parameter	Default value	Unit	Ref.
Q_w (milk, large animal)[a]	0.06	m^3/d	[6]
Q_w (meat, large animal)[a]	0.04	m^3/d	[6]
Q_m (milk, large animal)	16	kg/d (dry weight)	[63]
Q_f (meat, large animal)	12	kg/d (dry weight)	[63]
f_p	0.7	Unitless	—

[a] For small animals such as goats and sheep one tenth of these quantities should be used as a default.

where

$C_{m,i}$ is the concentration in milk of radionuclide i (Bq/L),

F_m is the fraction of the animal's daily intake of the radionuclide that appears in each litre of milk at equilibrium (d/L) (see Table XI),

$C_{a,i}$ is the concentration of radionuclide i in the animal feed (Bq/kg, dry matter),

$C_{w,i}$ is the concentration of radionuclide i in water (Bq/m^3),

Q_m is the amount of feed (in dry matter) consumed by the animal per day (kg/d) (see Table XII),

Q_w is the amount of water consumed by the animal per day (m^3/d) (see Table XII),

λ_i is the rate constant for radioactive decay of radionuclide i (d^{-1}),

t_m is the average time between collection and human consumption of milk (assumed to be one day for fresh milk) (see Table VIII).

Default values for F_m are given in Table XI. These values are based on reviews considered in Ref. [6] and values recommended in Ref. [9]. The values in Table XI are specific to dairy cows; however, they are applicable to other lactating animals without producing a substantial underestimate of the concentration in the milk of these animals. For small animals such as goats and sheep special values are supplied for caesium and iodine.

5.1.3.2. Concentration in meat

The radionuclide concentration in meat is calculated in the same way as the concentration in milk. The same constraints exist.

$$C_{f,i} = F_m (C_{a,i}Q_f + C_{w,i}Q_w) \exp(-\lambda_i t_f) \tag{37}$$

where

$C_{f,i}$ is the concentration of radionuclide i in animal flesh (Bq/kg),

F_f is the fraction of the animal's daily intake of a radionuclide that appears in each kg of flesh at equilibrium or at the time of slaughter (d/kg) (see Table XI),

$C_{a,i}$ is the concentration of radionuclide i in the animal's feed (Bq/kg, dry matter),

$C_{w,i}$ is the concentration of radionuclide i in water (Bq/m^3),

Q_f is the amount of feed (in dry matter) consumed by the animal per day (kg/d) (see Table XII),

Q_w is the amount of water consumed by the animal per day (m^3/d) (see Table XII),

λ_i is the rate constant for radioactive decay of radionuclide i (d^{-1}),

t_f is the average time between slaughter and human consumption of meat — a default value is 20 days (see Table VIII).

The default values for F_f in Table XI are based on data summarized in Ref. [6] and on recommendations in Ref. [9]. Because these values for F_m and F_f are intended for the purposes of a conservative generic assessment, they may deviate substantially from the 'expected' values published in other reports.

For generic calculations cattle have been selected as the reference herbivore. Although uptake can be expected to vary among species, the values for cattle can be assumed to apply to other meat producing animals. For small animals such as goats and sheep specific values are supplied for caesium and iodine. Values of F_f, Q_f and Q_w for other animal species are summarized in Ref. [6].

5.1.4.   (Semi-)natural terrestrial ecosystems

There is evidence that in some (semi-)natural terrestrial ecosystems the behaviour of radionuclides may differ from their behaviour in agricultural ecosystems. For example, it appears that caesium remains much more available in (semi-)natural ecosystems at high latitudes. Such caesium may be taken up by mushrooms that are consumed by game. Another example is the accumulation of caesium, from atmospheric sources, by lichens that are consumed by reindeer. For particular areas higher soil to plant uptake values for caesium are recommended (see Table XI). These values are sufficiently conservative to account for the increased uptake in (semi-)natural ecosystems in uplands. A complete treatment of these phenomena is outside the scope of generic assessment studies; the user is referred to Refs [6, 61].

5.2.   AQUATIC FOOD CHAIN TRANSPORT

Radionuclides discharged into the aquatic environment are also assimilated by living organisms. Some of the assimilated radionuclides are passed along the aquatic

food chains and may eventually reach humans. Models are used for dose assessments to simulate the transport of radionuclides in aquatic environments.

5.2.1. Basic model

Models that describe the transport of radionuclides from liquid discharges to aquatic foods generally take the form

$$C_{af,i} = C_{w,i} B_p / 1000 \tag{38}$$

where

$C_{af,i}$ is the concentration of radionuclide i in aquatic food p (Bq/kg);
$C_{w,i}$ is the concentration of dissolved radionuclide i in water (Bq/m^3);
B_p is the equilibrium ratio of the concentration of radionuclide i in aquatic food p to its dissolved concentration in water (Bq·kg^{-1}/Bq·L^{-1}, or L/kg), known as the bioaccumulation factor;
1000 is the conversion factor from m^3 to L.

Methods for the generic estimation of $C_{w,i}$ were considered in Section 4.

5.2.2. Bioaccumulation factor B_p

The transfer of radionuclides from water, through various trophic levels of aquatic life, to those organisms consumed by humans is condensed into one parameter — the bioaccumulation factor B_p. This parameter is quite variable, with values ranging in some cases over several orders of magnitude for a given radionuclide and organism [6].

The most important factor governing the value of B_p is the trophic level of the organism. Other factors are

(a) Suspended sediment concentration,
(c) Chemical composition of the water body,
(d) Chemical state of the released radionuclide,
(e) Characteristics specific to the aquatic organism.

For the purpose of these generic calculations default values of B_p (Table XIII) have been selected to ensure that the transfer of dissolved radionuclides from water to aquatic organisms is conservatively estimated, and to avoid the possibility of substantial underestimation occurring for any specific application. As a result, the values given in Table XIII may differ from listings of B_p given in other reports.

TABLE XIII. ELEMENT SPECIFIC BIOACCUMULATION FACTOR B_p[a]

Element	Freshwater fish (Bq·kg^{-1}/Bq·L^{-1})	Marine fish (Bq·kg^{-1}/Bq·L^{-1})	Marine shellfish (Bq·kg^{-1}/Bq·L^{-1})
Ac	15	50	1 000
Ag	5	500	10 000
Am	30	50	20 000
As	500	1 000	2 000
At	15	10	50
Au	35	100	1 000
Ba	4	10	1
Bi	10	20	1 000
Br	400	3	10
C[b]			
Cd	200	1 000	20 000
Ce	30	50	5 000
Cm	30	50	30 000
Co	300	1 000	5 000
Cr	200	200	800
Cs	2 000–10 000[c]	100	30
Cu	200	700	2 000
Eu	50	300	7 000
Fe	200	3 000	30 000
Ga	400	700	700
H[b]			
Hg	1 000	20 000	20 000
I	40	10	10
In	10 000	1 000	10 000
Mn	400	400	5 000
Mo	10	10	100
Na	20	0.1	0.3
Nb	300	30	1 000
Ni	100	1 000	2 000
Np	30	10	400
P	50 000	30 000	20 000
Pa	10	50	500
Pb	300	200	1 000
Pd	10	300	300
Pm	30	500	5 000
Po	50	2 000	50 000
Pu	30	40	3 000
Ra	50	500	1 000
Rb	2 000	100	20

TABLE XIII. (cont.)

Element	Freshwater fish $(Bq \cdot kg^{-1}/Bq \cdot L^{-1})$	Marine fish $(Bq \cdot kg^{-1}/Bq \cdot L^{-1})$	Marine shellfish $(Bq \cdot kg^{-1}/Bq \cdot L^{-1})$
Rh	10	100	1 000
Ru	10	2	2 000
S	800	2	4
Sb	100	400	400
Se	200	6 000	6 000
Sr	15–75[c]	2	2
Tc	20	30	1 000
Te	400	1 000	1 000
Th	100	600	1 000
Tl	1 000	5 000	5 000
U	10	1	30
Y	30	20	1 000
Zn	1 000	1 000	50 000
Zr	300	20	5 000

[a] Values derived from Refs [6, 55, 68–74].

[b] Models for tritium and carbon are covered separately in Annex III.

[c] See Section 5.2.4 for explanation.

5.2.3. Adjustment of B_p for the effect of suspended sediment

For generic assessment purposes it is often simpler to use the total concentrations of radionuclides ($C_{w, tot}$) estimated by the equations in Sections 4.3 to 4.6 rather than the dissolved radionuclide concentrations estimated in Section 4.7. This introduces conservatism for particle reactive radionuclides because some fraction of these will be adsorbed on to suspended particulate matter and thus will be unavailable for biological uptake. If this pathway is important and better estimates are needed, then the dissolved concentrations may be estimated (see Section 4.7).

5.2.4. Adjustment of B_p for caesium and strontium in freshwater fish

In Table XIII ranges are given for the values of B_p for strontium and caesium in freshwater fish. For regions with sedimentary bedrock, clay rich soil and hard water the lower values should be selected. For regions with igneous bedrock, sandy or organic soils, and soft water the higher values are appropriate. If dissolved potassium and suspended sediment concentrations are known, the site specific B_p value for

caesium can be estimated as described in Ref. [6]. Similarly, if dissolved calcium concentrations are known, the site specific B_p for strontium can be estimated [6, 68].

5.2.5. Biota not included in this Safety Report

Although the categories of aquatic biota presented in Table XIII encompass those most frequently consumed, it is recognized that in some regions freshwater crustaceans and marine macroalgae are also consumed. B_p values for freshwater crustaceans are generally unavailable, but may be assumed to be ten times greater than those for freshwater fish, with the exception of caesium, whose B_p value is three times lower. For marine macroalgae B_p values are presented in Ref. [55].

5.3. UNCERTAINTY ASSOCIATED WITH THESE PROCEDURES

As discussed earlier, the models and default parameter values presented here are intended for use in calculating doses for screening purposes, such that they are generally likely to overestimate doses received and are unlikely to underestimate real doses by more than a factor of ten under any circumstances. They have been based partly on recommendations for similar models presented elsewhere (see Ref. [9]), where further discussion on uncertainties is given. Particular points are also made here.

(a) The models for terrestrial food chain transfer are thought to be generally conservative. In particular, they do not include any allowance for the reduction in radionuclide concentrations owing to food preparation and processing, which can be significant.

(b) The models for transfer of radionuclides to milk and meat are based on information for cattle. However, it is expected that their use for other animals should not lead to substantial underestimation. In particular, the predicted concentrations in milk should not be more than a factor of three less than the actual concentration, even for milk from other species.

(c) For predicting the transfer of radionuclides to aquatic foods, a simple concentration factor approach is adopted through the use of bioaccumulation factors. These factors have been chosen specifically for screening procedures and thus are thought to be conservative.

6. DOSIMETRIC, HABIT AND OTHER DATA FOR DOSE ESTIMATION

The previous sections describe methods for calculating concentrations in various environmental materials, and other intermediate quantities needed to evaluate radiation doses. In order to complete the dose calculations outlined in Section 2, various dosimetric factors are required. For example, additional data are needed to estimate the annual inhalation dose E_{inh} (Sv/a) and the annual plume immersion dose E_{imm} (Sv/a) from the annual average radionuclide concentration in air C_A (Bq/m^3), estimated in Section 3, and to estimate the external dose E_{gr} (Sv/a) from radionuclides deposited on the ground over a period of 30 years (Fig. 2). Similarly, data are needed to determine doses from the ingestion of food and drinking water, and from external radiation from radionuclides in aquatic sediments. Dose factors are also needed to estimate doses from sewage sludge from the external pathway and from inhalation of resuspended material from activity on the surface of a sewage sludge container C_{sludge} (Bq/m^2), estimated in Section 4 (Fig. 2). These dosimetric data are discussed in this section, and default values are given for the various cases. In addition, the relevant habit information and other data required to estimate doses are summarized. In each case, data are provided to evaluate the effective dose as defined by the ICRP [2] and the BSS [1]. Information is also provided for calculation of doses to skin and to the lens of the eye.

In some cases, the ingestion or inhalation dose conversion factor associated with a particular radionuclide will depend on the chemical or physical form of the radionuclide. For example, the ingestion dose conversion factor for the radionuclide ^{35}S depends on whether the material is in the organic or inorganic form. The internal dose conversion factors presented in this section for screening purposes are the highest reasonable values listed in Ref. [1] for each radionuclide. The dose coefficient values given are those recommended in the BSS for all unspecified compounds for the purpose of calculating doses to workers.

The process involved in calculating doses from atmospheric discharges and from discharges to the hydrosphere and to sewerage systems is summarized in Fig 2.

6.1. ESTIMATION OF TOTAL INDIVIDUAL DOSES FROM A SOURCE

The following sections describe how individual doses are calculated for the transfer of particular radionuclides through particular pathways. For generic assessment purposes the total hypothetical critical group dose due to a particular source is estimated by summing the doses from all pathways and all radionuclides. Thus the hypothetical critical group is assumed to represent those members of the

public most exposed from the source from all possible pathways. In practice this is unlikely to occur, although it is a reasonable assumption for generic purposes.

As explained earlier, the dose of concern is the maximum dose received during the 30th year of discharge. The calculations are based either on the assumption that equilibrium is reached (e.g. air or water pathways) or on the assumption of a continuous buildup of long lived radionuclides in the environment (e.g. soil or sediment pathways), as appropriate.

6.2. CALCULATION OF EXTERNAL DOSES FROM AIRBORNE RADIONUCLIDES

The annual effective dose from immersion in the atmospheric discharge plume E_{im} (Sv/a) is given by

$$E_{im} = C_A DF_{im} O_f \tag{39}$$

where

C_A is the annual average concentration of nuclide i in air (Bq/m^3) calculated in Section 3,

DF_{im} is the effective dose coefficient for immersion (Sv/a per Bq/m),

O_f is the fraction of the year for which the hypothetical critical group member is exposed to this particular pathway.

The value of O_f will vary depending on the particular circumstances of the exposure. Values of O_f applicable for screening purposes are given in Table XIV.

Values of dose coefficients DF_{im} for selected radionuclides, obtained using the semi-infinite cloud model described below, are given in Table XV. These values are based on those given in Ref. [75] with additional account of the ingrowth of daughters with a half-life of less than 30 min. This half-life cut-off was chosen to ensure that only those progeny likely to be present relatively close to the site, and therefore to expose potential critical group members, are taken into account. These values can be combined with the nuclide concentrations in air C_A, obtained as described in Section 3, to estimate annual hypothetical critical group doses arising from external exposure to the material in the cloud.

The semi-infinite cloud model assumes that radiation from the cloud is in radiative equilibrium, so that the energy absorbed by a given volume element of the cloud is equal to that emitted by the same element. However, near to the ground the radiation source represents only one half of the total solid angle subtended, so that the energy absorbed in a given volume element is only one half of the energy emitted by

the same element. This model has been widely used, and several compilations of dose conversion factors relating dose rate with the nuclide concentration in air have been published (see, for example, Refs [76, 77]).

The degree of shielding in a particular building depends on a number of factors, including the structure and the composition of the building materials. Shielding factors

TABLE XIV. DEFAULT VALUES OF HABIT AND OTHER DATA FOR EXTERNAL EXPOSURE, INHALATION AND INGESTION DOSE ESTIMATION FOR A CRITICAL GROUP IN EUROPE

Type of exposure	Adult		Infant (1 a)	
	Occupancy (h/a)	Fraction O_f	Occupancy (h/a)	Fraction O_f
External exposure				
Surface contaminated owing to air deposition	8760	1	8760	1
Working/playing over contaminated sediments	1600	0.18	1000	0.12
Submersion in air	8760	1	8760	1
Garden and ground exposure from irrigation	500	0.06	500	0.06
Inhalation				
Breathing rate (m^3/a)	8400		1400	
Ingestion	Intake per person			
Freshwater fish (kg/a)	30		15	
Marine fish (kg/a)	50		25	
Marine shellfish (kg/a)	15		0	
Water and beverages (m^3/a)	0.600		0.260	
Fruit, vegetables and grain, including potatoes (kg/a)	410		150	
Milk (L/a)	250		300	
Meat (kg/a)	100		40	

TABLE XV. EFFECTIVE EXTERNAL DOSE COEFFICIENTS FOR VARIOUS RADIONUCLIDES

Nuclide	Effective dose from cloud immersion[a] (Sv/a per Bq/m^3)	Skin dose from cloud immersion[b] (Sv/a per Bq/m^3)	Effective dose from surface deposits[c] (Sv/a per Bq/m^2)
Ac-228	1.5×10^{-6}	2.5×10^{-6}	3.1×10^{-8}
Ag-110m	4.3×10^{-6}	5.0×10^{-6}	8.5×10^{-8}
Am-241	2.6×10^{-8}	4.0×10^{-8}	8.9×10^{-10}
As-76	7.0×10^{-7}	3.0×10^{-6}	1.7×10^{-8}
At-211[d]	5.1×10^{-8}	6.2×10^{-8}	1.3×10^{-9}
Au-198	6.2×10^{-7}	1.3×10^{-6}	1.3×10^{-8}
Bi-206	5.1×10^{-6}	6.0×10^{-6}	1.0×10^{-7}
Bi-210[d]	8.3×10^{-9}	7.3×10^{-7}	1.1×10^{-9}
Bi-212[d]	2.3×10^{-6}	3.9×10^{-6}	4.2×10^{-8}
Br-82	4.1×10^{-6}	4.9×10^{-6}	8.1×10^{-8}
Cd-109	9.6×10^{-9}	3.1×10^{-8}	7.4×10^{-10}
Ce-141	1.1×10^{-7}	3.2×10^{-7}	2.4×10^{-9}
Ce-144	1.1×10^{-7}	2.7×10^{-6}	5.7×10^{-9}
Cm-242	1.9×10^{-10}	1.4×10^{-9}	3.3×10^{-11}
Cm-244	1.7×10^{-10}	1.2×10^{-9}	3.0×10^{-11}
Co-58	1.5×10^{-6}	1.8×10^{-6}	3.0×10^{-8}
Co-60	4.0×10^{-6}	4.6×10^{-6}	7.5×10^{-8}
Cr-51	4.8×10^{-8}	5.5×10^{-8}	9.8×10^{-10}
Cs-134	2.4×10^{-6}	3.0×10^{-6}	4.9×10^{-8}
Cs-135	3.0×10^{-10}	2.9×10^{-8}	1.1×10^{-12}
Cs-136	3.4×10^{-6}	3.9×10^{-6}	6.7×10^{-8}
Cs-137[d]	8.7×10^{-7}	1.4×10^{-6}	1.8×10^{-8}
Cu-64	2.9×10^{-7}	5.2×10^{-7}	6.0×10^{-9}
Eu-154	2.0×10^{-6}	2.6×10^{-6}	3.8×10^{-8}
Eu-155	8.0×10^{-8}	1.1×10^{-7}	1.9×10^{-9}
Fe-55	0	0	0
Fe-59	1.9×10^{-6}	2.2×10^{-6}	3.6×10^{-8}
Ga-67	2.3×10^{-7}	2.7×10^{-7}	4.8×10^{-9}
Hg-197	8.5×10^{-8}	1.1×10^{-7}	2.1×10^{-9}
Hg-197m	1.3×10^{-7}	3.2×10^{-7}	2.8×10^{-9}
Hg-203	3.6×10^{-7}	4.9×10^{-7}	7.4×10^{-9}
I-123	2.3×10^{-7}	3.0×10^{-7}	5.3×10^{-9}
I-125	1.7×10^{-8}	4.4×10^{-8}	1.4×10^{-9}
I-129	1.2×10^{-8}	3.5×10^{-8}	8.3×10^{-10}
I-131	5.8×10^{-7}	9.4×10^{-7}	1.2×10^{-8}
I-132	3.6×10^{-6}	5.0×10^{-6}	7.2×10^{-8}
I-133	9.5×10^{-7}	1.8×10^{-6}	2.0×10^{-8}
I-134	4.2×10^{-6}	5.9×10^{-6}	8.3×10^{-8}

TABLE XV. (cont.)

Nuclide	Effective dose from cloud immersion[a] (Sv/a per Bq/m^3)	Skin dose from cloud immersion[b] (Sv/a per Bq/m^3)	Effective dose from surface deposits[c] (Sv/a per Bq/m^2)
I-135	2.6×10^{-6}	3.5×10^{-6}	4.8×10^{-8}
In-111	5.9×10^{-7}	7.2×10^{-7}	1.2×10^{-8}
In-113m	3.9×10^{-7}	6.9×10^{-7}	8.1×10^{-9}
Mn-54	1.3×10^{-6}	1.5×10^{-6}	2.6×10^{-8}
Mo-99[d]	4.0×10^{-7}	1.2×10^{-6}	9.2×10^{-9}
Na-22	3.4×10^{-6}	4.2×10^{-6}	6.7×10^{-8}
Na-24	7.0×10^{-6}	8.7×10^{-6}	1.2×10^{-7}
Nb-95	1.2×10^{-6}	1.4×10^{-6}	2.4×10^{-8}
Ni-59	0	0	0
Ni-63	0	0	0
Np-237[d]	3.3×10^{-8}	4.9×10^{-8}	7.2×10^{-9}
Np-239	2.5×10^{-7}	5.0×10^{-7}	5.2×10^{-9}
P-32	1.7×10^{-8}	1.4×10^{-6}	2.7×10^{-9}
Pa-231	5.5×10^{-8}	7.7×10^{-8}	1.3×10^{-9}
Pa-233	3.0×10^{-7}	5.2×10^{-7}	6.2×10^{-9}
Pb-210[d]	1.8×10^{-9}	4.0×10^{-9}	1.2×10^{-9}
Pd-103[d]	2.5×10^{-9}	1.2×10^{-8}	4.0×10^{-10}
Pd-107	0	0	0
Pd-109	1.5×10^{-8}	6.8×10^{-7}	1.3×10^{-9}
Pm-147	2.8×10^{-10}	2.6×10^{-8}	1.1×10^{-12}
Po-210	1.3×10^{-11}	1.5×10^{-11}	2.6×10^{-13}
Pu-238	1.7×10^{-10}	1.3×10^{-9}	2.9×10^{-11}
Pu-239	1.4×10^{-10}	5.9×10^{-10}	1.3×10^{-11}
Pu-240	1.6×10^{-10}	1.2×10^{-9}	2.8×10^{-11}
Pu-241[d]	2.3×10^{-12}	3.7×10^{-12}	3.0×10^{-11}
Pu-242	1.4×10^{-10}	1.0×10^{-9}	2.3×10^{-11}
Ra-224[d]	1.6×10^{-8}	2.1×10^{-8}	4.7×10^{-8}
Ra-225[d]	9.7×10^{-9}	9.5×10^{-8}	8.8×10^{-9}
Ra-226[d]	1.0×10^{-8}	1.5×10^{-8}	5.7×10^{-8}
Rb-86	1.7×10^{-7}	1.5×10^{-6}	5.4×10^{-9}
Rh-105	1.2×10^{-7}	3.4×10^{-7}	2.5×10^{-9}
Rh-107	4.9×10^{-7}	1.4×10^{-6}	1.1×10^{-8}
Ru-103	7.2×10^{-7}	8.7×10^{-7}	1.5×10^{-8}
Ru-106[d]	3.6×10^{-7}	3.4×10^{-6}	1.1×10^{-8}
S-35	1.0×10^{-10}	9.2×10^{-9}	5.5×10^{-13}
Sb-124	2.9×10^{-6}	4.0×10^{-6}	5.6×10^{-8}
Sb-125[d]	6.5×10^{-7}	8.4×10^{-7}	1.4×10^{-8}
Se-75	5.9×10^{-7}	6.8×10^{-7}	1.2×10^{-8}
Sn-113[d]	1.2×10^{-8}	2.6×10^{-8}	8.8×10^{-9}

TABLE XV. (cont.)

Nuclide	Effective dose from cloud immersion[a] (Sv/a per Bq/m^3)	Skin dose from cloud immersion[b] (Sv/a per Bq/m^3)	Effective dose from surface deposits[c] (Sv/a per Bq/m^2)
Sr-85	7.7×10^{-7}	8.9×10^{-7}	1.6×10^{-8}
Sr-87m	4.9×10^{-7}	6.8×10^{-7}	1.0×10^{-8}
Sr-89	1.4×10^{-8}	1.2×10^{-6}	2.2×10^{-9}
Sr-90[d]	3.1×10^{-9}	2.9×10^{-7}	3.5×10^{-9}
Tc-99	9.2×10^{-10}	8.6×10^{-8}	2.5×10^{-12}
Tc-99m	1.9×10^{-7}	2.3×10^{-7}	3.9×10^{-9}
Te-125m	1.5×10^{-8}	6.1×10^{-8}	1.2×10^{-9}
Te-127m	4.9×10^{-9}	2.7×10^{-8}	7.0×10^{-10}
Te-129m	5.4×10^{-8}	4.7×10^{-7}	1.9×10^{-9}
Te-131m[d]	2.5×10^{-6}	3.4×10^{-6}	4.9×10^{-8}
Te-132[d]	3.3×10^{-7}	4.4×10^{-7}	7.9×10^{-8}
Th-228[d]	2.9×10^{-9}	4.7×10^{-9}	4.7×10^{-8}
Th-230[d]	5.6×10^{-10}	1.4×10^{-9}	5.7×10^{-8}
Th-232[d]	2.9×10^{-10}	1.1×10^{-9}	7.8×10^{-8}
Tl-201	1.2×10^{-7}	1.5×10^{-7}	2.8×10^{-9}
Tl-202	7.0×10^{-7}	8.3×10^{-7}	1.5×10^{-8}
U-232[d]	4.7×10^{-10}	1.9×10^{-9}	4.7×10^{-8}
U-234[d]	2.5×10^{-10}	1.3×10^{-9}	5.7×10^{-8}
U-235[d]	2.3×10^{-7}	2.7×10^{-7}	5.3×10^{-9}
U-238[d]	1.2×10^{-10}	9.2×10^{-10}	6.0×10^{-8}
Y-87[d]	6.9×10^{-7}	7.9×10^{-7}	2.4×10^{-8}
Y-90	2.6×10^{-8}	2.0×10^{-6}	3.5×10^{-9}
Y-91	2.0×10^{-8}	1.2×10^{-6}	2.4×10^{-9}
Zn-65	9.2×10^{-7}	1.0×10^{-6}	1.8×10^{-8}
Zr-95[d]	1.1×10^{-6}	1.4×10^{-6}	4.7×10^{-8}

[a] The effective dose values given here have been estimated from the effective dose equivalent values for immersion in the cloud, given in Ref. [75], plus the corresponding weighted skin dose component, to provide an approximation to effective dose. Additional account of the contribution from radioactive progeny with half-lives less than 30 min has also been taken where appropriate.

[b] Skin doses from immersion in the cloud are taken from Ref. [75]. Additional account has been taken of radioactive progeny with half-lives less than 30 min, where appropriate. These data are provided to allow comparison with the dose limit for skin. The skin dose component has been included in the effective dose coefficients, which are likely to be more useful for other purposes.

[c] The effective dose values given here have been estimated from the effective dose equivalent values for surface deposits, given in Ref. [75], plus the corresponding weighted skin dose component. Additional account has been taken of the contribution from radioactive progeny likely to be important over a period of 30 years (the assumed period of operation and deposition).

Radioactive progeny considered in external dose coefficients are listed below. Where relevant, branching ratios are shown in brackets.

Nuclide	Progeny	
At-211	Po-211	
Bi-210	Po-210	
Bi-212	Tl-208 (0.3593), Po-212 (0.6407)	
Ce-144	Pr-144	
Cs-137	Ca-137m	
Mo-99	Tc-99m, Tc-99	
Np-237	Pa-233	
Pb-210	Bi-210	
Pd-103	Rh-103m	(deposit only)
Pu-241	Am-241	(deposit only)
Ra-224	Rn-220, Po-210, Pb-212, Bi-212, Tl-208 (0.3593), Po-212 (0.6407)	(deposit only)
Ra-225	Ac-225, Fr-221, At-217, Bi-213, Po-213 (0.978), Tl-209 (0.0216)	(deposit only)
Ra-226	Rn-222, Po-210, Pb-214, Bi-214, Po-210, Bi-210, Po-214, Po-210	
Ru-106	Rh-106	
Sb-125	Re-125m	
Sn-113	In-113m	
Sr-90	Y-90	
Te-127m	Te-127	
Te-131m	Te-131, I-131	
Te-132	I-132	
Th-228	Ra-224, Rn-220, Po-210, Pb-212, Bi-212, Tl-208 (0.3593), Po-212 (0.6407)	(deposit only)
Th-230	Ra-226, Rn-222, Po-210, Pb-214, Bi-214, Po-210, Bi-210, Po-214, Po-210	
Th-232	Ra-228, Ac-228, Th-228, Ra-224, Rn-220, Po-210, Pb-212, Bi-212, Tl-208 (0.3593), Po-212 (0.6407)	(deposit only)
U-232	Th-228, Ra-224, Rn-220, Po-210, Pb-212, Bi-212, Tl-208 (0.3593), Po-212 (0.6407)	(deposit only)
U-234	Th-230, Ra-226, Rn-222, Po-210, Pb-214, Bi-214, Po-210, Bi-210, Po-214, Po-210	
U-235	Th-231	(deposit only)
U-238	Th-234, Pa-234m, U-234, Th-230, Ra-226, Rn-222, Po-210, Pb-214, Bi-214, Po-210, Bi-210, Po-214, Po-210	
Y-87	Sr-87m	(deposit only)
Zr-95	Nb-95	(deposit only)

have been found to range from 0.01–0.4 or more [78], with higher values applying to wooden houses and lower values applying to multistorey buildings and offices. The time an individual will spend indoors depends on a number of factors, such as climate, occupation and individual habits. On the global scale, the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has suggested that, on average, 80% of time is spent indoors [78].

For estimation of external exposure the shielding effects of buildings and time spent indoors could be considered. However, for the purposes of this simple generic assessment methodology these considerations do not need to be taken into account.

For evaluation of the effective dose from external exposure, γ radiation from airborne radionuclides is the main exposure pathway. However, some radionuclides give rise to β radiation, which can lead to exposure to the skin. The equivalent dose to skin can be calculated from the annual average concentration of a given radionuclide in air (see Section 3.2.1) and the dose coefficients for the skin given in Table XV, using Eq. (40)

$$E_{\mathrm{im},s} = C_A DF_s O_f \tag{40}$$

where

$E_{\mathrm{im},s}$ is the annual skin dose from β irradiation (Sv/a),
DF_s is the skin dose owing to β irradiation per unit air concentration (Sv/a per Bq/m^3).

The coefficients given in Table XV were calculated as described above from data in Ref. [75]. The effective dose from skin exposure may be obtained by multiplying the skin dose by the weighting factor of 0.01 [1, 2].

Some regulatory instruments require the dose to the lens of the eye to be calculated. For purposes of an initial generic assessment the dose to the lens of the eye is assumed to be equal to the dose calculated for the skin.

6.3. CALCULATION OF EXTERNAL DOSES FROM DEPOSITED ACTIVITY

The annual effective dose from ground deposition E_{gr} (Sv/a) is given by

$$E_{\mathrm{gr}} = C_{\mathrm{gr}} DF_{\mathrm{gr}} O_f \tag{41}$$

where

DF_{gr} is the dose coefficient for exposure to ground deposits (Sv/a per Bq/m^2),

C_{gr} is the deposition density of radionuclide i (Bq/m^2).

C_{gr} is obtained from the ground deposition rate $\dot{d}_i$ calculated in Section 3.9 according to the equation

$$C_{gr} = \frac{\dot{d}_i \left[1 - \exp(-\lambda_{E_i^s} t_b)\right]}{\lambda_{E_i^s}} \tag{42}$$

where

$\dot{d}_i$ is the total ground deposition rate (Bq·m^{-2}·d^{-1}) (Section 3.9);

$\lambda_{E_i^s}$ is the effective rate constant for reduction of the activity in the top 10 to 20 cm of soil (d^{-1}), where $\lambda_{E_i^s} = \lambda_i + \lambda_s$;

λ_s is the rate constant for reduction of soil activity owing to processes other than radioactive decay;

t_b is the duration of the discharge of radioactive material (d).

Default values of t_b and λ_s are provided in Section 5 (Tables VIII and X). Values for λ_i are provided in Annex II.

Table XV gives dose coefficients to estimate the effective dose arising from unit deposition on the ground for a number of radionuclides. These values have been calculated on the basis of data given in Ref. [75] for external doses from surface deposits. The skin component has been included in the effective dose coefficients, and additional account has been taken of the ingrowth of progeny likely to make a significant contribution to external dose over the assumed operational period of 30 years. Other compilations of data are also available (e.g. Refs [76, 77]).

For a more realistic estimation of individual doses from this exposure route, particularly for longer lived radionuclides, the effects of migration down the soil column and the shielding effect of overlying layers of soil would need to be taken into account. In addition, shielding afforded by buildings during time spent indoors should also be considered, as discussed in Section 6.2.

6.3.1. Estimating external doses from deposits

The estimation of external dose from material deposited on to the ground is a relatively straightforward procedure, and a number of models exist for this purpose. The simplest way to calculate the dose at a reference height of 1 m above a contaminated surface is to assume that the surface is an infinite plane source on which

activity is uniformly distributed. This method is strictly appropriate only for radionuclides with short radioactive half-lives that are deposited on smooth surfaces. For longer lived radionuclides it is necessary to model the migration down through the soil column. A number of models have been developed to predict this downward movement, but the experimental data to support the models are limited to a few radionuclides and a few soil types. However, this process may be of considerable significance in determining the external exposure over the longer term from the deposition of long lived, γ emitting radionuclides, and it should be taken into account for this purpose. The time variation of the vertical profile of material can then be used to estimate the exposure above the soil surface, using standard methods [77]. The shielding provided by overlying layers of soil as a consequence of this downward migration of radionuclides through soil is not taken into account in the estimation of the dose conversion factors given in Table XV.

6.4. CALCULATION OF EXTERNAL DOSES FROM ACTIVITY IN SEDIMENTS

Section 4 describes how to calculate the radionuclide concentrations in sediments and water following discharges of radionuclides into rivers, estuaries, coastal waters or lakes, or from deposition from atmospheric discharges on to lakes. That section also gives guidance on the locations for which the concentrations should be calculated, if these are not already known.

External exposure can occur from sediments on tidal flats and along river banks containing radionuclides discharged to the water body. Other possible external exposure pathways may occur, for example as a result of swimming, boating or handling fishing gear. However, the external dose from time spent on sediment is likely to be much greater than that from the other external aquatic pathways, and therefore this pathway alone is considered in this simple generic approach.

The annual effective dose due to external exposure E_m (Sv/a) from sediment is given by

$$E_m = C_{s,s} DF_{gr} O_f \qquad (43)$$

where

$C_{s,s}$ is the surface activity concentration in shore/beach sediments (Bq/m^2) as described in Section 4.7.5,

DF_{gr} is the dose coefficient for exposure to ground deposits (Sv/a per Bq/m^2),

O_f is the fraction of the year for which a hypothetical critical group member is exposed to this particular pathway.

Occupancy times characteristic of the potential critical group members at the site of interest may be available, but default values representing a hypothetical critical group for use in a generic assessment are given in Table XIV.

6.5. CALCULATION OF INTERNAL DOSES DUE TO INTAKE BY INHALATION AND INGESTION

The internal dose following an intake of radioactive material into the body by inhalation or ingestion is protracted in time after the intake. The effective dose coefficient from intake will depend on the metabolism, age and life expectancy of the individual as well as the physicochemical behaviour of the radionuclide concerned. Dose coefficients are usually evaluated using representative 'reference' values for the various factors, such as those related to metabolism, and are therefore averages either for complete populations or for particular subgroups in a population.

In assessing the exposure of the general public, account has to be taken of the age of the exposed population and of the likely chemical form of radioactive material in the environment. Partly as a result of the smaller body size of children, dose coefficients for children are generally greater than those for adults. Although this tends to be counteracted by lower inhalation and ingestion rates, infants and children may receive higher individual doses from a source of exposure than would adults. In assessing the exposure of hypothetical critical groups, the intakes and effective dose coefficients appropriate to different ages should ideally be used. Models have been developed to predict the exposure of members of the public of different ages owing to the intake of radionuclides in forms likely to be encountered in the environment (e.g. Ref. [79, 80]). Databases have been compiled of the dose coefficients for intake by inhalation and ingestion of a large number of radionuclides in a wide range of forms as a function of the age of the individual at the time of intake. The BSS [1] contain a compilation of dose coefficient data for children and adults.

Values of effective dose coefficients for intake by inhalation and ingestion taken from Ref. [1] are given in Tables XVI and XVII, respectively. Values are presented for infants (1–2 years) and adults. A variety of radionuclides are considered and, for each, data are presented for the chemical forms recommended for 'unspecified compounds' in the BSS (a generally conservative approach). For adults the doses per unit intake are integrated for 50 years. For infants the doses are integrated to age 70 from the age at intake.

6.5.1. Irradiation from inhaled radionuclides

The annual effective dose from inhalation E_{inh} (Sv/a) is

TABLE XVI. COMMITTED EFFECTIVE DOSE COEFFICIENTS FOR INHALATION (Sv/Bq)

Nuclide	Lung class	Infant (1–2 a)	Adult
Ac-228	F	1.6×10^{-7}	2.5×10^{-8}
Ag-110m	S	4.1×10^{-8}	1.2×10^{-8}
Am-241	M	6.9×10^{-5}	4.2×10^{-5}
As-76	M	4.6×10^{-9}	7.4×10^{-10}
At-211	S	3.7×10^{-7}	1.1×10^{-7}
Au-198	S	4.4×10^{-9}	8.6×10^{-10}
Bi-206	S	8.0×10^{-9}	1.7×10^{-9}
Bi-210	S	3.0×10^{-7}	9.3×10^{-8}
Bi-212	S	1.1×10^{-7}	3.1×10^{-8}
Br-82	M	3.0×10^{-9}	6.3×10^{-10}
Cd-109	F	3.7×10^{-8}	8.1×10^{-9}
Ce-141	S	1.2×10^{-8}	3.8×10^{-9}
Ce-144	S	1.8×10^{-7}	5.3×10^{-8}
Cm-242	M	1.8×10^{-5}	5.2×10^{-6}
Cm-244	M	5.7×10^{-5}	2.7×10^{-5}
Co-58	S	7.5×10^{-9}	2.1×10^{-9}
Co-60	S	8.6×10^{-8}	3.1×10^{-8}
Cr-51	S	2.1×10^{-10}	3.7×10^{-11}
Cs-134	F	7.3×10^{-9}	6.6×10^{-9}
Cs-135	F	9.9×10^{-10}	6.9×10^{-10}
Cs-136	F	5.2×10^{-9}	1.2×10^{-9}
Cs-137	F	5.4×10^{-9}	4.6×10^{-9}
Cu-64	S	5.7×10^{-10}	1.2×10^{-10}
Eu-154	M	1.5×10^{-7}	5.3×10^{-8}
Eu-155	M	2.3×10^{-8}	6.9×10^{-9}
Fe-55	F	3.2×10^{-9}	7.7×10^{-10}
Fe-59	S	1.3×10^{-8}	4.0×10^{-9}
Ga-67	M	1.0×10^{-9}	2.4×10^{-10}
Hg-197	M (inorganic)	1.2×10^{-9}	3.0×10^{-10}
Hg-197m	M (inorganic)	2.5×10^{-9}	5.3×10^{-10}
Hg-203	M (inorganic)	7.9×10^{-9}	2.4×10^{-9}
I-123	F	7.9×10^{-10}	7.4×10^{-11}
I-125	F	2.3×10^{-8}	5.1×10^{-9}
I-129	F	8.6×10^{-8}	3.6×10^{-8}
I-131	F	7.2×10^{-8}	7.4×10^{-9}
I-132	F	9.6×10^{-10}	1.1×10^{-10}
I-133	F	1.8×10^{-8}	1.5×10^{-9}
I-134	F	3.7×10^{-10}	4.5×10^{-11}
I-135	F	3.7×10^{-9}	3.2×10^{-10}
In-111	M	1.2×10^{-9}	2.3×10^{-10}

TABLE XVI. (cont.)

Nuclide	Lung class	Infant (1–2 a)	Adult
In-113m	M	1.1×10^{-10}	2.0×10^{-11}
Mn-54	M	6.2×10^{-9}	1.5×10^{-9}
Mo-99	S	4.8×10^{-9}	9.9×10^{-10}
Na-22	F	7.3×10^{-9}	1.3×10^{-9}
Na-24	F	1.8×10^{-9}	2.7×10^{-10}
Nb-95	S	5.9×10^{-9}	1.8×10^{-9}
Ni-59	F	8.1×10^{-10}	1.8×10^{-10}
Ni-63	F	2.0×10^{-9}	4.4×10^{-10}
Np-237	M	4.0×10^{-5}	2.3×10^{-5}
Np-239	M	4.2×10^{-9}	9.3×10^{-10}
P-32	M	1.5×10^{-8}	3.4×10^{-9}
Pa-231	M	2.3×10^{-4}	1.4×10^{-4}
Pa-233	S	1.3×10^{-8}	3.9×10^{-9}
Pb-210	M	3.7×10^{-6}	1.1×10^{-6}
Pd-103	S	1.8×10^{-9}	4.5×10^{-10}
Pd-107	S	2.0×10^{-9}	5.9×10^{-10}
Pd-109	S	1.9×10^{-9}	3.7×10^{-10}
Pm-147	M	1.8×10^{-8}	5.0×10^{-9}
Po-210	F	4.8×10^{-6}	6.1×10^{-7}
Pu-238	M	7.4×10^{-5}	4.6×10^{-5}
Pu-239	M	7.7×10^{-5}	5.0×10^{-5}
Pu-240	M	7.7×10^{-5}	5.0×10^{-5}
Pu-241	M	9.7×10^{-7}	9.0×10^{-7}
Pu-242	M	7.3×10^{-5}	4.8×10^{-5}
Ra-224	S	9.2×10^{-6}	3.4×10^{-6}
Ra-225	S	2.2×10^{-5}	7.7×10^{-6}
Ra-226	S	2.9×10^{-5}	9.5×10^{-6}
Rb-86	F	7.7×10^{-9}	9.3×10^{-10}
Rh-105	S	1.7×10^{-9}	3.5×10^{-10}
Rh-107	S	9.7×10^{-11}	1.7×10^{-11}
Ru-103	S	1.0×10^{-8}	3.0×10^{-9}
Ru-106	S	2.3×10^{-7}	6.6×10^{-8}
S-35	M	4.5×10^{-9}	1.4×10^{-9}
Sb-124	M	2.4×10^{-8}	6.4×10^{-9}
Sb-125	M	1.6×10^{-8}	4.8×10^{-9}
Se-75	F	6.0×10^{-9}	1.3×10^{-9}
Sn-113	M	1.0×10^{-8}	2.7×10^{-9}
Sr-85	S	3.7×10^{-9}	8.1×10^{-10}
Sr-87m	S	1.2×10^{-10}	2.1×10^{-11}
Sr-89	S	3.0×10^{-8}	7.9×10^{-9}
Sr-90	S	4.0×10^{-7}	1.6×10^{-7}

TABLE XVI. (cont.)

Nuclide	Lung class	Infant (1–2 a)	Adult
Tc-99	F	2.5×10^{-9}	2.9×10^{-10}
Tc-99m	F	8.7×10^{-11}	1.2×10^{-11}
Te-125m	F	4.2×10^{-9}	5.1×10^{-10}
Te-127m	F	1.4×10^{-8}	1.5×10^{-9}
Te-129m	F	1.3×10^{-8}	1.3×10^{-9}
Te-131m	F	7.6×10^{-9}	8.6×10^{-10}
Te-132	F	1.8×10^{-8}	1.8×10^{-9}
Th-228	M	1.1×10^{-4}	3.2×10^{-5}
Th-230	M	7.4×10^{-5}	4.3×10^{-5}
Th-232	M	8.1×10^{-5}	4.5×10^{-5}
Tl-201	F	3.3×10^{-10}	4.4×10^{-11}
Tl-202	F	1.2×10^{-9}	1.9×10^{-10}
U-232	S	9.7×10^{-5}	3.7×10^{-5}
U-234	S	2.9×10^{-5}	9.4×10^{-6}
U-235	S	2.6×10^{-5}	8.5×10^{-6}
U-238	S	2.5×10^{-5}	8.0×10^{-6}
Y-87	S	2.2×10^{-9}	3.9×10^{-10}
Y-90	S	8.8×10^{-9}	1.5×10^{-9}
Y-91	S	3.4×10^{-8}	8.9×10^{-9}
Zn-65	F	1.0×10^{-8}	2.2×10^{-9}
Zr-95	S	1.9×10^{-8}	5.9×10^{-9}

TABLE XVII. COMMITTED EFFECTIVE DOSE COEFFICIENTS FOR INGESTION (Sv/Bq)

Nuclide	Gut transfer factor, f_1	Infant (1 a)	Adult
Ac-228	5.0×10^{-4}	2.8×10^{-9}	4.3×10^{-10}
Ag-110m	0.050	1.4×10^{-8}	2.8×10^{-9}
Am-241	5.0×10^{-4}	3.7×10^{-7}	2.0×10^{-7}
As-76	0.500	1.1×10^{-8}	1.6×10^{-9}
At-211	1.000	7.8×10^{-8}	1.1×10^{-8}
Au-198	0.100	7.2×10^{-9}	1.0×10^{-9}
Bi-206	0.050	1.0×10^{-8}	1.9×10^{-9}
Bi-210	0.050	9.7×10^{-9}	1.3×10^{-9}
Bi-212	0.050	1.8×10^{-9}	2.6×10^{-10}
Br-82	1.000	2.6×10^{-9}	5.4×10^{-10}
Cd-109	0.050	9.5×10^{-9}	2.0×10^{-9}
Ce-141	5.0×10^{-4}	5.1×10^{-9}	7.1×10^{-10}
Ce-144	5.0×10^{-4}	3.9×10^{-8}	5.2×10^{-9}
Cm-242	5.0×10^{-4}	7.6×10^{-8}	1.2×10^{-8}
Cm-244	5.0×10^{-4}	2.9×10^{-7}	1.2×10^{-7}
Co-58	0.100	4.4×10^{-9}	7.4×10^{-10}
Co-60	0.100	2.7×10^{-8}	3.4×10^{-9}
Cr-51	0.100	2.3×10^{-10}	3.8×10^{-11}
Cs-134	1.000	1.6×10^{-8}	1.9×10^{-8}
Cs-135	1.000	2.3×10^{-9}	2.0×10^{-9}
Cs-136	1.000	9.5×10^{-9}	3.0×10^{-9}
Cs-137	1.000	1.2×10^{-8}	1.3×10^{-8}
Cu-64	0.500	8.3×10^{-10}	1.2×10^{-10}
Eu-154	5.0×10^{-4}	1.2×10^{-8}	2.0×10^{-9}
Eu-155	5.0×10^{-4}	2.2×10^{-9}	3.2×10^{-10}
Fe-55	0.100	2.4×10^{-9}	3.3×10^{-10}
Fe-59	0.100	1.3×10^{-8}	1.8×10^{-9}
Ga-67	0.001	1.2×10^{-9}	1.9×10^{-10}
Hg-197	0.020	1.6×10^{-9}	2.3×10^{-10}
Hg-197m	0.020	3.4×10^{-9}	4.7×10^{-10}
Hg-203	1.000	1.1×10^{-8}	1.9×10^{-9}
I-123	1.000	1.9×10^{-9}	2.1×10^{-10}
I-125	1.000	5.7×10^{-8}	1.5×10^{-8}
I-129	1.000	2.2×10^{-7}	1.1×10^{-7}
I-131	1.000	1.8×10^{-7}	2.2×10^{-8}
I-132	1.000	2.4×10^{-9}	2.9×10^{-10}
I-133	1.000	4.4×10^{-8}	4.3×10^{-9}
I-134	1.000	7.5×10^{-10}	1.1×10^{-10}
I-135	1.000	8.9×10^{-9}	9.3×10^{-10}
In-111	0.020	1.7×10^{-9}	2.9×10^{-10}

TABLE XVII. (cont.)

Nuclide	Gut transfer factor, f_1	Infant (1 a)	Adult
In-113m	0.020	1.8×10^{-10}	2.8×10^{-11}
Mn-54	0.100	3.1×10^{-9}	7.1×10^{-10}
Mo-99	1.000	3.5×10^{-9}	6.0×10^{-10}
Na-22	1.000	1.5×10^{-8}	3.2×10^{-9}
Na-24	1.000	2.3×10^{-9}	4.3×10^{-10}
Nb-95	0.010	3.2×10^{-9}	5.8×10^{-10}
Ni-59	0.050	3.4×10^{-10}	6.3×10^{-11}
Ni-63	0.050	8.4×10^{-10}	1.5×10^{-10}
Np-237	5.0×10^{-4}	2.1×10^{-7}	1.1×10^{-7}
Np-239	5.0×10^{-4}	5.7×10^{-9}	8.0×10^{-10}
P-32	0.800	1.9×10^{-8}	2.4×10^{-9}
Pa-231	5.0×10^{-4}	1.3×10^{-6}	7.1×10^{-7}
Pa-233	5.0×10^{-4}	6.2×10^{-9}	8.7×10^{-10}
Pb-210	0.200	3.6×10^{-6}	6.9×10^{-7}
Pd-103	0.005	1.4×10^{-9}	1.9×10^{-10}
Pd-107	0.005	2.8×10^{-10}	3.7×10^{-11}
Pd-109	0.005	4.1×10^{-9}	5.5×10^{-10}
Pm-147	5.0×10^{-4}	1.9×10^{-9}	2.6×10^{-10}
Po-210	0.500	8.8×10^{-6}	1.2×10^{-6}
Pu-238	5.0×10^{-4}	4.0×10^{-7}	2.3×10^{-7}
Pu-239	5.0×10^{-4}	4.2×10^{-7}	2.5×10^{-7}
Pu-240	5.0×10^{-4}	4.2×10^{-7}	2.5×10^{-7}
Pu-241	5.0×10^{-4}	5.7×10^{-9}	4.8×10^{-9}
Pu-242	5.0×10^{-4}	4.0×10^{-7}	2.4×10^{-7}
Ra-224	0.200	6.6×10^{-7}	6.5×10^{-8}
Ra-225	0.200	1.2×10^{-6}	9.9×10^{-8}
Ra-226	0.200	9.6×10^{-7}	2.8×10^{-7}
Rb-86	1.000	2.0×10^{-8}	2.8×10^{-9}
Rh-105	0.050	2.7×10^{-9}	3.7×10^{-10}
Rh-107	0.050	1.6×10^{-10}	2.4×10^{-11}
Ru-103	0.050	4.6×10^{-9}	7.3×10^{-10}
Ru-106	0.050	4.9×10^{-8}	7.0×10^{-9}
S-35	1.000	5.4×10^{-9}	7.7×10^{-10}
Sb-124	0.100	1.6×10^{-8}	2.5×10^{-9}
Sb-125	0.100	6.1×10^{-9}	1.1×10^{-9}
Se-75	0.800	1.3×10^{-8}	2.6×10^{-9}
Sn-113	0.020	5.0×10^{-9}	7.3×10^{-10}
Sr-85	0.300	3.1×10^{-9}	5.6×10^{-10}
Sr-87m	0.300	1.7×10^{-10}	3.0×10^{-11}
Sr-89	0.300	1.8×10^{-8}	2.6×10^{-9}
Sr-90	0.300	7.3×10^{-8}	2.8×10^{-8}

TABLE XVII. (cont.)

Nuclide	Gut transfer factor, f_1	Infant (1 a)	Adult
Tc-99	0.500	4.8×10^{-9}	6.4×10^{-10}
Tc-99m	0.500	1.3×10^{-10}	2.2×10^{-11}
Te-125m	0.300	6.3×10^{-9}	8.7×10^{-10}
Te-127m	0.300	1.8×10^{-8}	2.3×10^{-9}
Te-129m	0.300	2.4×10^{-8}	3.0×10^{-9}
Te-131m	0.300	1.4×10^{-8}	1.9×10^{-9}
Te-132	0.300	3.0×10^{-8}	3.8×10^{-9}
Th-228	5.0×10^{-4}	3.7×10^{-7}	7.2×10^{-8}
Th-230	5.0×10^{-4}	4.1×10^{-7}	2.1×10^{-7}
Th-232	5.0×10^{-4}	4.5×10^{-7}	2.3×10^{-7}
Tl-201	1.000	5.5×10^{-10}	9.5×10^{-11}
Tl-202	1.000	2.1×10^{-9}	4.5×10^{-10}
U-232	0.020	8.2×10^{-7}	3.3×10^{-7}
U-234	0.020	1.3×10^{-7}	4.9×10^{-8}
U-235	0.020	1.3×10^{-7}	4.7×10^{-8}
U-238	0.020	1.2×10^{-7}	4.5×10^{-8}
Y-87	1.0×10^{-4}	3.2×10^{-9}	5.5×10^{-10}
Y-90	1.0×10^{-4}	2.0×10^{-8}	2.7×10^{-9}
Y-91	1.0×10^{-4}	1.8×10^{-8}	2.4×10^{-9}
Zn-65	0.500	1.6×10^{-8}	3.9×10^{-9}
Zr-95	0.010	5.6×10^{-9}	9.5×10^{-10}

$$E_{\mathrm{inh}} = C_A R_{\mathrm{inh}} DF_{\mathrm{inh}} \qquad\qquad (44)$$

where

C_A is the radionuclide concentration in air (Bq/m^3),
R_{inh} is the inhalation rate (m^3/a),
DF_{inh} is the inhalation dose coefficient (Sv/Bq) (see Table XVI).

Table XIV gives default inhalation rates for adults and for 1 to 2 year old infants.

6.5.2.   Ingestion of radionuclides

Section 5 describes how to calculate the concentration of radionuclides in terrestrial and aquatic foodstuffs. The ingestion doses for infants and adults are then calculated using the following general equation

$$E_{\text{ing},\,p} = C_{p,i}H_p DF_{\text{ing}} \qquad\qquad (45)$$

where

$E_{\text{ing},\,p}$ is the annual effective dose from consumption of nuclide i in foodstuff p (Sv/a),
$C_{p,i}$ is the concentration of radionuclide i in foodstuff p at the time of consumption (Bq/kg),
H_p is the consumption rate for foodstuff p (kg/a),
DF_{ing} is the dose coefficient for ingestion of radionuclide i (Sv/Bq).

The default intake rates and dosimetric data are given in Tables XIV, XVII and XVIII.

Equation (45) can also be used to estimate the doses arising from drinking water, in which case H_p would be the drinking water intake rate and C_p would be the concentration of radionuclides in drinking water.

TABLE XVIII. DEFAULT VALUES OF INTAKE PER PERSON FOR VARIOUS CRITICAL GROUPS IN THE WORLD (ADULTS)

	Far East	Near East	Africa	South America	Central America	North America	Europe	Oceania
Water (m³/a)	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
Milk (L/a)	65	140	80	135	155	325	250	410
Meat (kg/a)	40	55	35	90	75	205	100	200
Grain, root crops, vegetables and fruits (kg/a)	510	600	380	470	445	535	410	500
Freshwater fish (kg/a)	35	10	15	20	25	25	30	15
Marine fish (kg/a)	60	20	30	35	45	40	50	30
Shellfish (kg/a)	20	5	10	10	15	15	15	10

6.6. RADIATION DOSES FROM RADIONUCLIDES IN SEWAGE SLUDGE

Two potential exposure pathways from radionuclides in sewage sludge are considered in Fig. 2. These pathways are external irradiation and inhalation of resuspended material in the sewage plant itself. Other exposure pathways, such as inadvertent ingestion of sludge and those pathways resulting from the disposal of the sludge on to agricultural land, are generally likely to be lower than the predicted doses to workers in a sewage plant, and thus can be ignored in simple generic dose calculations. Hypothetical individual doses from each of these pathways are calculated as follows.

6.6.1. External irradiation exposure

The external irradiation dose from radionuclides in sewage sludge can be estimated in the same way as the dose from material deposited on the ground (Section 6.4) and that from activity in sediments (Section 6.5); that is

$$E_{s,i} = C'_{\text{sludge}} DF_{\text{gr}} O_f \tag{46}$$

where

$E_{s,i}$ is the external radiation dose from sewage sludge for radionuclide i (Sv/a),
C'_{sludge} is the surface concentration of radionuclide i in sewage sludge (Bq/m^2),
DF_{gr} is the dose conversion factor for ground contamination (Sv/a per Bq/m^2),
O_f is the fraction of time of exposure per year.

C'_{sludge} is the surface concentration of the radionuclide per unit area of the sewage sludge. It can be obtained from the calculated concentration of radionuclides per unit mass of the sludge, calculated as described in Section 4, assuming that the density of sewage sludge is 1000 kg/m^3 and the depth of the sewage sludge container is 1 m. Therefore

$$\begin{aligned} C'_{\text{sludge}} &= C_{\text{sludge}} \times \rho \times \text{depth} \\ &= C_{\text{sludge}} \times 1000 \end{aligned} \tag{47}$$

where C_{sludge} is the concentration of radionuclide i in sewage sludge in Bq/kg (wet weight). As explained in Section 4.8, the radionuclide concentration in wet sewage sludge is required. This may be calculated from a dry weight value by multiplying by a conversion factor of 0.05.

Table XV gives values of the ground dose conversion factor for a variety of radionuclides as discussed in Section 6.3. The fraction of time of exposure per year can be taken as 0.228, based on 2000 h/a.

6.6.2. Inhalation of resuspended material

$$E_{\text{res}} = C_{\text{sludge}} R_{\text{inh}} DF_{\text{inh}} O_f DL \qquad (48)$$

where

E_{res} is the annual dose from inhaling resuspended sewage sludge (Sv/a),
C_{sludge} is the concentration of the radionuclide in sewage sludge (Bq/kg wet weight),
R_{inh} is the annual inhalation rate (m^3/a),
DF_{inh} is the dose coefficient for intake by inhalation (Sv/Bq),
O_f is the fraction of the year during which exposure occurs,
DL is the resuspension factor (kg/m^3).

The concentration in sewage sludge is obtained as described in Section 4. However, it should be adjusted to wet weight. The annual inhalation rate is given in Table XIV, and the doses per unit intake by inhalation for various radionuclides are given in Table XVI. The default fraction of the year for which workers are exposed is 0.228, as above for external exposure to sewage sludge. A default value of 0.1 mg/m³, corresponding to 1×10^{-7} kg/m³, is suggested for the resuspension factor, based on measurements made around a sewage plant [81].

7. ESTIMATION OF COLLECTIVE DOSE FOR SCREENING PURPOSES

The collective dose may be considered as the product of the number of individuals exposed to a source and their average radiation dose. The definition of collective dose, together with the aims and methods of assessment, is given in more detail in Ref. [1]. In this Safety Report the term collective dose is used to refer to the collective effective dose integrated to infinity, unless otherwise indicated.

Simple collective dose assessments have two main roles in the control of radioactive discharges to the environment. Firstly, they may form part of the screening process to demonstrate compliance with a dose limiting criterion. In this case an assessment of collective dose is used in conjunction with the assessment of critical group doses as discussed in Section 8. Secondly, collective doses may be used in a simple optimization exercise to compare protection options in a semi-quantitative way. Modelling procedures for estimating collective doses are similar to those used to estimate individual doses, except that concentrations of radionuclides in environmental compartments extending over large regions, or even globally, are used.

In principle, the estimation of collective dose commitments should include all individual doses, regardless of their magnitude and of when and where they may occur. In practice, however, collective doses are calculated for different subgroups of the population, and in many cases a distinction is made between local, regional and global components. The dispersion of most radionuclides in the environment is limited owing to their short physical half-lives or to removal processes such as sedimentation or migration into the deep layers of the soil. Collective doses are therefore estimated for most radionuclides only in the local and regional zones, which may extend from the point of release to distances varying from about 100 km to several thousand kilometres. Global assessments are generally performed only for four radionuclides: ^{3}H, ^{14}C, ^{85}Kr and ^{129}I, which are characterized by relatively long radioactive half-lives and/or a high environmental mobility. Other long lived radionuclides, such as ^{99}Tc or ^{237}Np, may also become globally dispersed following discharge. However, they have been little studied and are not considered further here.

To assess the collective dose from an atmospheric discharge, the main modelling method involves dividing the region of concern into a number of smaller areas that may be regarded as reasonably homogeneous with respect to population, agricultural practice, etc. The collective dose is then estimated separately for each area, and the total is determined by summation. For discharges to the hydrosphere compartment models are generally used to represent dispersion and transfer through the environment to humans. The collective doses are then evaluated by summing over pathways and radionuclides. Complex models are available for use in realistic collective dose estimates or for more detailed assessments such as estimation of the temporal or geographical distribution of the collective doses [82].

7.1. GENERIC ESTIMATES OF COLLECTIVE DOSE

Models for precise assessments of collective doses are normally relatively complex and require the use of computers [82]. UNSCEAR [83] has developed simple methods for estimating collective doses using generic parameters. These models must be used with caution and can only provide order of magnitude estimates.

Estimates of collective dose have been made for unit discharges of radionuclides to the atmosphere and to water bodies. These are intended for screening or simple generic assessment purposes only. These estimates are based on the results of more complex models applied to locations in western Europe [84] and on the results of simple models based on UNSCEAR methods and using generic, global parameters. From these results order of magnitude estimates of the collective dose per unit discharge have been obtained, and are given in Table XIX for discharges of radionuclides to the atmosphere, in Table XX for discharges of radionuclides into

marine waters and in Table XXI for discharges of radionuclides into fresh waters. Further information on the origin of these values is given in Annex VII.

It is emphasized that the collective doses given in Tables XIX–XXI are order of magnitude estimates only and should be used with caution. They should be used only as part of a screening or generic assessment procedure, for example to ensure compliance with dose limiting criteria or as input to an optimization exercise to compare options as part of an intuitive, semi-quantitative analysis. They should not be used for more rigorous optimization analyses, such as cost–benefit analyses, nor for other purposes. The values provided in Tables XIX–XXI are collective dose commitments, integrated to infinity. For collective doses delivered in the far future there exist considerable uncertainties about the sizes, locations and characteristics of populations. The estimates for long lived radionuclides, particularly for those that become globally distributed, should therefore be treated with particular caution.

8. PROCEDURES TO FOLLOW WHEN ESTIMATED DOSES EXCEED THE SPECIFIED REFERENCE LEVEL

This Safety Report provides an iterative screening approach for assessing critical group doses resulting from radioactive discharges to the environment. Two types of model are provided, the first is a no dilution method that effectively assumes that members of the public receive exposures at the point of discharge. This approach provides a quick and easy way to determine whether a further, more refined approach is warranted. Dose calculation factors based on this approach are given in Annex I. If the critical group dose estimated in this way exceeds the relevant dose criterion (e.g. dose constraint), an assessment using a simple generic environmental model that takes account of dispersion and dilution of discharges in the environment is recommended. A simple generic environmental model is provided in this report which represents the second stage of the iterative critical group assessment process, illustrated in Fig. 1. This model is described in detail in Sections 3 to 6. Dose calculation factors based on this model are also presented in Annex I. These factors were calculated using the data and models presented in this report, together with standardized assumptions regarding the discharge conditions and the habits and location of a hypothetical critical group. The objective of this section is to explain the iterative dose assessment procedure in more detail.

The models presented in this Safety Report represent the first two stages in the iterative assessment approach illustrated in Fig. 21. They are applicable across a wide range of sites and release conditions, and have been designed not to require extensive collection of site specific data, calculational complexity or expertise in environmental

TABLE XIX. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED TO THE ATMOSPHERE, FOR SCREENING PURPOSES

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
Ac-228	1×10^{-12}
Ag-110m	1×10^{-12}
Am-241	1×10^{-9}
Ar-41	1×10^{-16}
As-76	1×10^{-12}
At-211	1×10^{-13}
Au-198	1×10^{-13}
Bi-206	1×10^{-12}
Bi-210	1×10^{-13}
Bi-212	1×10^{-12}
Br-82	1×10^{-12}
C-14	1×10^{-11}
Cd-109	1×10^{-12}
Ce-141	1×10^{-13}
Ce-144	1×10^{-12}
Cm-242	1×10^{-11}
Cm-244	1×10^{-10}
Co-58	1×10^{-12}
Co-60	1×10^{-11}
Cr-51	1×10^{-14}
Cs-134	1×10^{-11}
Cs-135	1×10^{-13}
Cs-136	1×10^{-12}
Cs-137	1×10^{-11}
Cu-64	1×10^{-13}
Eu-154	1×10^{-11}
Eu-155	1×10^{-13}
Fe-55	1×10^{-14}
Fe-59	1×10^{-12}
Ga-67	1×10^{-13}
H-3	1×10^{-15}
Hg-197	1×10^{-13}
Hg-197m	1×10^{-13}
Hg-203	1×10^{-13}
I-123	1×10^{-13}
I-125	1×10^{-12}
I-129	1×10^{-10}
I-131	1×10^{-12}

TABLE XIX. (cont.)

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
I-132	1×10^{-12}
I-133	1×10^{-12}
I-134	1×10^{-12}
I-135	1×10^{-12}
In-111	1×10^{-13}
In-113m	1×10^{-13}
Kr-85	1×10^{-17}
Kr-89	1×10^{-17}
Mn-54	1×10^{-12}
Mo-99	1×10^{-13}
Na-22	1×10^{-11}
Na-24	1×10^{-12}
Nb-95	1×10^{-12}
Ni-59	1×10^{-13}
Ni-63	1×10^{-12}
Np-237	1×10^{-10}
Np-239	1×10^{-13}
P-32	1×10^{-12}
Pa-231	1×10^{-10}
Pa-233	1×10^{-13}
Pb-210	1×10^{-10}
Pd-103	1×10^{-14}
Pd-107	1×10^{-14}
Pd-109	1×10^{-14}
Pm-147	1×10^{-14}
Po-210	1×10^{-10}
Pu-238	1×10^{-9}
Pu-239	1×10^{-9}
Pu-240	1×10^{-10}
Pu-241	1×10^{-11}
Pu-242	1×10^{-10}
Ra-224	1×10^{-11}
Ra-225	1×10^{-11}
Ra-226	1×10^{-10}
Rb-86	1×10^{-12}
Rh-105	1×10^{-13}
Rh-107	1×10^{-13}
Ru-103	1×10^{-12}
Ru-106	1×10^{-12}
S-35	1×10^{-12}

TABLE XIX. (cont.)

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
Sb-124	1×10^{-12}
Sb-125	1×10^{-12}
Se-75	1×10^{-11}
Sn-113	1×10^{-13}
Sr-85	1×10^{-12}
Sr-87m	1×10^{-13}
Sr-89	1×10^{-13}
Sr-90	1×10^{-11}
Tc-99	1×10^{-13}
Tc-99m	1×10^{-13}
Te-125m	1×10^{-13}
Te-127m	1×10^{-13}
Te-129m	1×10^{-13}
Te-131m	1×10^{-12}
Te-132	1×10^{-12}
Th-228	1×10^{-10}
Th-230	1×10^{-10}
Th-232	1×10^{-9}
Tl-201	1×10^{-13}
Tl-202	1×10^{-12}
U-232	1×10^{-10}
U-234	1×10^{-9}
U-235	1×10^{-10}
U-238	1×10^{-10}
Xe-133	1×10^{-16}
Xe-135	1×10^{-16}
Y-87	1×10^{-12}
Y-90	1×10^{-13}
Y-91	1×10^{-13}
Zn-65	1×10^{-11}
Zr-95	1×10^{-12}

TABLE XX. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED INTO MARINE WATERS, FOR SCREENING PURPOSES

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
Ac-228	1×10^{-18}
Ag-110m	1×10^{-13}
Am-241	1×10^{-13}
As-76	1×10^{-16}
At-211	1×10^{-17}
Au-198	1×10^{-16}
Bi-206	1×10^{-16}
Bi-210	1×10^{-16}
Bi-212	1×10^{-19}
Br-82	1×10^{-19}
C-14	1×10^{-11}
Cd-109	1×10^{-13}
Ce-141	1×10^{-16}
Ce-144	1×10^{-14}
Cm-242	1×10^{-13}
Cm-244	1×10^{-12}
Co-58	1×10^{-15}
Co-60	1×10^{-13}
Cr-51	1×10^{-17}
Cs-134	1×10^{-13}
Cs-135	1×10^{-14}
Cs-136	1×10^{-16}
Cs-137	1×10^{-13}
Cu-64	1×10^{-17}
Eu-154	1×10^{-14}
Eu-155	1×10^{-15}
Fe-55	1×10^{-13}
Fe-59	1×10^{-13}
Ga-67	1×10^{-16}
H-3	1×10^{-18}
Hg-197	1×10^{-15}
Hg-197m	1×10^{-15}
Hg-203	1×10^{-13}
I-123	1×10^{-19}
I-125	1×10^{-16}
I-129	1×10^{-11}
I-131	1×10^{-16}
I-132	1×10^{-20}

TABLE XX. (cont.)

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
I-133	1×10^{-18}
I-134	1×10^{-21}
I-135	1×10^{-19}
In-111	1×10^{-16}
In-113m	1×10^{-18}
Mn-54	1×10^{-14}
Mo-99	1×10^{-18}
Na-22	1×10^{-17}
Na-24	1×10^{-21}
Nb-95	1×10^{-16}
Ni-59	1×10^{-14}
Ni-63	1×10^{-15}
Np-237	1×10^{-12}
Np-239	1×10^{-17}
P-32	1×10^{-13}
Pa-231	1×10^{-11}
Pa-233	1×10^{-16}
Pb-210	1×10^{-11}
Pd-103	1×10^{-16}
Pd-107	1×10^{-15}
Pd-109	1×10^{-17}
Pm-147	1×10^{-15}
Po-210	1×10^{-10}
Pu-238	1×10^{-12}
Pu-239	1×10^{-12}
Pu-240	1×10^{-11}
Pu-241	1×10^{-13}
Pu-242	1×10^{-12}
Ra-224	1×10^{-14}
Ra-225	1×10^{-13}
Ra-226	1×10^{-11}
Rb-86	1×10^{-15}
Rh-105	1×10^{-17}
Rh-107	1×10^{-20}
Ru-103	1×10^{-15}
Ru-106	1×10^{-13}
S-35	1×10^{-17}
Sb-124	1×10^{-14}
Sb-125	1×10^{-14}
Se-75	1×10^{-13}

TABLE XX. (cont.)

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
Sr-85	1×10^{-17}
Sr-87m	1×10^{-21}
Sr-89	1×10^{-17}
Sr-90	1×10^{-15}
Tc-99	1×10^{-14}
Tc-99m	1×10^{-19}
Te-125m	1×10^{-14}
Te-127m	1×10^{-14}
Te-129m	1×10^{-14}
Te-131m	1×10^{-16}
Te-132	1×10^{-15}
Th-228	1×10^{-14}
Th-230	1×10^{-14}
Th-232	1×10^{-14}
Tl-201	1×10^{-16}
Tl-202	1×10^{-15}
U-232	1×10^{-13}
U-234	1×10^{-13}
U-235	1×10^{-13}
U-238	1×10^{-13}
Y-87	1×10^{-17}
Y-90	1×10^{-17}
Y-91	1×10^{-16}
Zn-65	1×10^{-12}
Zr-95	1×10^{-15}

TABLE XXI. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED INTO FRESHWATER BODIES, FOR SCREENING PURPOSES

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
Ac-228	1×10^{-17}
Ag-110m	1×10^{-12}
Am-241	1×10^{-11}
As-76	1×10^{-16}
At-211	1×10^{-16}
Au-198	1×10^{-16}
Bi-206	1×10^{-15}
Bi-210	1×10^{-15}
Bi-212	1×10^{-18}
Br-82	1×10^{-16}
C-14	1×10^{-12}
Cd-109	1×10^{-13}
Ce-141	1×10^{-15}
Ce-144	1×10^{-13}
Cm-242	1×10^{-13}
Cm-244	1×10^{-11}
Co-58	1×10^{-13}
Co-60	1×10^{-12}
Cr-51	1×10^{-16}
Cs-134	1×10^{-11}
Cs-135	1×10^{-12}
Cs-136	1×10^{-13}
Cs-137	1×10^{-11}
Cu-64	1×10^{-17}
Eu-154	1×10^{-12}
Eu-155	1×10^{-13}
Fe-55	1×10^{-14}
Fe-59	1×10^{-14}
Ga-67	1×10^{-16}
H-3	1×10^{-14}
Hg-197	1×10^{-16}
Hg-197m	1×10^{-16}
Hg-203	1×10^{-14}
I-123	1×10^{-17}
I-125	1×10^{-11}
I-129	1×10^{-11}
I-131	1×10^{-11}
I-132	1×10^{-18}

TABLE XXI. (cont.)

Radionuclide	Collective effective dose commitment per unit discharge (man·Sv/Bq)
I-133	1×10^{-16}
I-134	1×10^{-19}
I-135	1×10^{-17}
In-111	1×10^{-15}
In-113m	1×10^{-18}
Mn-54	1×10^{-14}
Mo-99	1×10^{-16}
Na-22	1×10^{-13}
Na-24	1×10^{-17}
Nb-95	1×10^{-13}
Ni-59	1×10^{-15}
Ni-63	1×10^{-14}
Np-237	1×10^{-12}
Np-239	1×10^{-16}
P-32	1×10^{-12}
Pa-231	1×10^{-11}
Pa-233	1×10^{-15}
Pb-210	1×10^{-10}
Pd-103	1×10^{-16}
Pd-107	1×10^{-15}
Pd-109	1×10^{-17}
Pm-147	1×10^{-14}
Po-210	1×10^{-11}
Pu-238	1×10^{-12}
Pu-239	1×10^{-12}
Pu-240	1×10^{-12}
Pu-241	1×10^{-13}
Pu-242	1×10^{-12}
Ra-224	1×10^{-14}
Ra-225	1×10^{-13}
Ra-226	1×10^{-10}
Rb-86	1×10^{-14}
Rh-105	1×10^{-17}
Rh-107	1×10^{-20}
Ru-103	1×10^{-13}
Ru-106	1×10^{-11}
S-35	1×10^{-12}
Sb-124	1×10^{-14}
Sb-125	1×10^{-12}
Se-75	1×10^{-14}

TABLE XXI. (cont.)

Radionuclide	Collective effective dose commitmentper unit discharge (man·Sv/Bq)
Sr-85	1×10^{-15}
Sr-87m	1×10^{-19}
Sr-89	1×10^{-12}
Sr-90	1×10^{-11}
Tc-99	1×10^{-13}
Tc-99m	1×10^{-18}
Te-125m	1×10^{-14}
Te-127m	1×10^{-14}
Te-129m	1×10^{-14}
Te-131m	1×10^{-16}
Te-132	1×10^{-15}
Th-228	1×10^{-11}
Th-230	1×10^{-11}
Th-232	1×10^{-11}
Tl-201	1×10^{-16}
Tl-202	1×10^{-15}
U-232	1×10^{-11}
U-234	1×10^{-10}
U-235	1×10^{-12}
U-238	1×10^{-10}
Xe-133	1×10^{-16}
Xe-135	1×10^{-16}
Y-87	1×10^{-16}
Y-90	1×10^{-15}
Y-91	1×10^{-14}
Zn-65	1×10^{-12}
Zr-95	1×10^{-14}

radiological assessment. Under most circumstances these models are likely to result in overestimation of the actual doses experienced by members of a critical group. Under some conditions, however, underestimation may occur, but in specifying the parameters used in the model care has been taken to ensure that the extent of underestimation should not exceed a factor of ten, even under extreme conditions.

8.1. AN ITERATIVE APPROACH TO EVALUATION

The step-wise iterative procedure recommended in this report to assess critical group doses is illustrated in Fig. 21. This approach has the advantage that it represents an efficient use of assessment resources in that, if applied correctly, it allows resources to be targeted at those aspects of the assessment that give the highest doses. To apply this method a suitable reference level of dose is needed to indicate when a more accurate dose assessment, and the use of a more complex model, is required. It is recommended that this reference level be set to take account of both the relevant dose limiting criterion and the uncertainty associated with the dose assessment model, as discussed in Section 2.1.1.

8.1.1. Initial assessment steps

The first step in this iterative process is to assess doses on the basis of the no dilution model. If doses calculated using this approach are less than the relevant dose criterion (e.g. dose constraint), no further model complexity is needed, since this approach is very conservative and actual doses received would be expected to be much lower than those predicted by this method. If doses exceed the dose criterion a greater level of accuracy in the model predictions is warranted and the generic environmental model is recommended.

The second step is to apply the generic environmental model. If the critical group doses predicted using this model are less than a reference level (e.g. one tenth of a dose constraint specified by a Regulatory Authority), no further model complexity is needed. However, if doses exceed the reference level a re-evaluation of the assumed input data is recommended, as indicated in Fig. 21.

8.1.2. Re-evaluation of input data

The objective of this stage in the process is to review the input data applied in making the generic assessment discussed in the previous section, in order to determine whether they are unduly pessimistic in relation to the conditions at the site considered. Examples of the data that may be considered are discussed below.

8.1.2.1. Estimated discharge rate and conditions

In this process the first step is to re-evaluate the estimated discharge rate and the conditions of discharge to confirm that they are not overestimates. If the discharge rate has been overestimated the generic dose calculations may be repeated using a revised estimate.
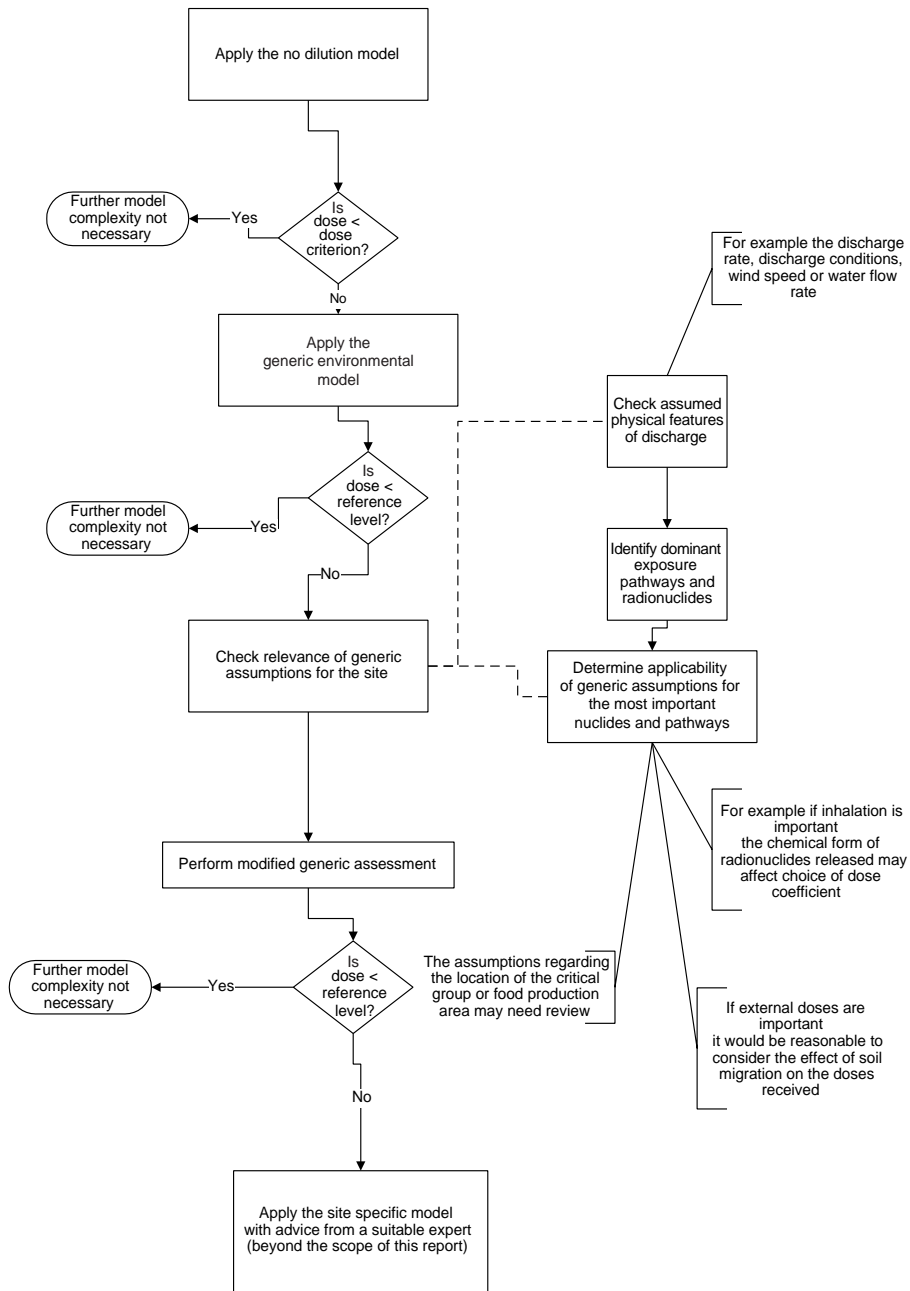
FIG. 21. An iterative approach for assessing critical group doses giving examples of factors that may be considered when checking the relevance of generic assumptions to a site.

8.1.2.2. Exposure conditions

If, after review of the discharge rate and discharge conditions, the revised calculated doses are still greater than the reference level, it is recommended that the exposure conditions, assumed as part of the generic methodology, should be re-evaluated and revised to reflect more closely the conditions actually prevailing at the site. This re-evaluation process should begin with those conditions that apply to the radionuclides and exposure pathways that dominate the dose estimate. Any changes in the exposure assumptions that lead to a revised estimate of the hypothetical critical group dose should be accompanied by a detailed justification of these modifications. The variables in the generic methodology that are likely to be most amenable to site specific re-evaluation and modification are shown below.

(a)   The location of members of the hypothetical critical group and of food production;
(b)   Dietary habits and residence times of members of the hypothetical critical group;
(c)   Annual average flow rate of a surface water body;
(d)   Annual average wind speed;
(e)   Specific exposure pathways;
(f) K_d values for the specific site and the specific chemical forms of radionuclides at the site;
(g)   If external doses are important, it may be necessary to consider migration of radionuclides in soil.

After the initial estimates of dose are completed, the dominant radionuclides and exposure pathways should be identified and evaluated. This evaluation may reveal that some of the pathways assumed in the generic models are absent at the site. Before these pathways are eliminated from a revised generic calculation, care must be taken to account for the presence of other food types not explicitly considered in the generic methodology. The generic food types included in this methodology are intended to serve as surrogates for any food type that might actually be present. Thus the generic meat pathway is intended to be sufficiently robust to account for the harvesting of wild game. The generic fish pathway is also intended to account for the consumption of other aquatic biota that are not specifically identified in the model, including waterfowl. The combined use of the generic pathways of milk, meat and vegetables (including the default values for consumption rates) is intended to account for hypothetical critical group consumers of forest mushrooms.

Some of the generic pathways included will, however, always be present at a given site. For example, if it is possible that people occupy the lands surrounding an atmospheric discharge location, inhalation will always have to be taken into account.

Similarly, external exposure to radionuclides deposited on the ground will always have to be considered if an atmospheric discharge contains γ emitting particulates. Intakes through local food products will also be of general relevance, but the distances between the farm or garden where local foods are produced and the discharge location vary.

8.1.3.   Final revised generic dose calculations

Once the adjustments for site specific conditions are made, the generic calculations should be repeated. If the dose estimates still exceed the reference level, it is recommended that the following be considered.

— Reducing the amount discharged,
— Modifying the conditions of the discharge,
— Consulting professionals in environmental radiological assessments to perform a detailed site specific assessment.

8.2.   REALISTIC DOSE ASSESSMENTS IN CONSULTATION WITH QUALIFIED PROFESSIONALS USING MORE ACCURATE MODELS

If the revised calculations remain above the reference level, and it is not reasonable either to reduce the amount discharged or modify the conditions of the discharge, a more detailed site specific assessment of the dose to members of the critical group should be considered. Such an assessment should be performed by qualified professionals who are able to calculate hypothetical doses using both more accurate models and more realistic estimates of parameter values. The models, assumptions and parameter values used in the assessment should be documented and made available for review by competent authorities. Recommendations for qualified professionals and for more accurate models can be obtained by contacting the IAEA.

When realistic dose estimates are made it is advisable to quantify sources of uncertainty and to propagate these through the calculations to produce a statement of confidence (e.g. a 95% subjective confidence interval) about the dose estimate [5]. The dose estimate and its confidence bounds can then be compared with the relevant dose limiting criterion. If the upper confidence bound of the dose estimate is above this criterion, efforts should be made either to reduce the uncertainty in the site specific dose estimate or to reduce the amount discharged.

Uncertainty is most effectively reduced by collecting additional data on those components of the model that dominate the uncertainty in the dose estimate. The dominant components can be identified through a quantitative uncertainty analysis performed in conjunction with a sensitivity analysis. The procedures of quantitative uncertainty and sensitivity analysis are described in detail in Ref. [5].

If the lower confidence bound of the dose estimated using the more accurate methods exceeds the relevant dose limiting criterion, and if this effect is confirmed through external review by qualified individuals, then there should be a high degree of confidence that actual doses will exceed the criterion. Further investigation is not likely to be justified. Although additional studies should lead to improved confidence in the dose estimate, the fact that the dose limiting criteria are violated will merely be confirmed. Under these circumstances, the only remaining options are to

— Reduce the amount discharged and repeat the assessment,
— Modify the conditions under which the discharge occurs and repeat the assessment,
— Abandon the practice or operation,
— Apply for a special case variance from the relevant dose limiting criterion.

These issues are discussed in more detail in Ref. [3].

REFERENCES

[1] FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL LABOUR ORGANISATION, OECD NUCLEAR ENERGY AGENCY, PAN AMERICAN HEALTH ORGANIZATION, WORLD HEALTH ORGANIZATION, International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources, Safety Series No. 115, IAEA, Vienna (1996).

[2] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, 1990 Recommendations of the International Commission on Radiological Protection, Publication 60, Pergamon Press, Oxford and New York (1991).

[3] INTERNATIONAL ATOMIC ENERGY AGENCY, Regulatory Control of Radioactive Discharges to the Environment, Safety Standards No. WS-G-2.3, IAEA, Vienna (2000).

[4] INTERNATIONAL ATOMIC ENERGY AGENCY, Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases: Exposures of Critical Groups, Safety Series No. 57, IAEA, Vienna (1982).

[5] INTERNATIONAL ATOMIC ENERGY AGENCY, Evaluating the Reliability of Predictions Made Using Environmental Transfer Models, Safety Series No. 100, IAEA, Vienna (1989).

[6] INTERNATIONAL ATOMIC ENERGY AGENCY, Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments, Technical Reports Series No. 364, IAEA, Vienna (1994).

[7] TILL, J.E., MEYER, H.R. (Eds), Radiological Assessment. A Textbook on Environmental Dose Analysis, Rep. NUREG/CR-3332, ORNL-5968, United States Nuclear Regulatory Commission, Washington, DC (1983).

[8] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Screening Techniques for Determining Compliance with Environmental Standards. Releases of Radionuclides to the Atmosphere, NCRP Commentary No. 3, Revision plus Addendum, NCRP, Bethesda, MD (1989).

[9] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Screening Models for Releases of Radionuclides to Atmosphere, Surface Water, and Ground, Rep. 123 I, NCRP, Bethesda, MD (1996).

[10] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Screening Models for Releases of Radionuclides to Atmosphere, Surface Water, and Ground — Work Sheets, Rep. 123 II, NCRP, Bethesda, MD (1996).

[11] INTERNATIONAL ATOMIC ENERGY AGENCY, Atmospheric Dispersion in Nuclear Power Plant Siting: A Safety Guide, Safety Series No. 50-SG-S3, IAEA, Vienna (1980).

[12] HEINEMANN, K., VOGT, K.J., "Statistical studies on the limitation of short-time releases from nuclear facilities", Congress of the International Radiation Protection Association (Proc. 5th Congr. Jerusalem, 1980), Vol. 2, IRPA, Washington, DC (1980) 67–70.

[13] GIFFORD, F.A., "Diffusion in the lower layers of the atmosphere", Meteorology and Atomic Energy (SLADE, D.H., Ed.), Rep. TID-24 190, United States Atomic Energy Commission, Washington, DC (1968) 65.

[14] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment, Rep. 76, NCRP, Bethesda, MD (1984).

[15] PASQUILL, F., SMITH, F.B., Atmospheric Diffusion, 3rd edn, Ellis Horwood, Chichester (1983).

[16] VOGT, K.J., "Atmosphärische Ausbreitungen und Ablagerungen radioaktiver Stoffe", Radioökologie, Berichtsband der Fachtagung Radioökologie des Deutschen Atomforums e.V., Bonn, 1979, Vulkan-Verlag, Essen (1979).

[17] CLARKE, R.H., A Model for Short and Medium Range Dispersion of Radionuclides Released to the Atmosphere. The First Report of a Working Group on Atmospheric Dispersion, Rep. NRPB-R91, National Radiological Protection Board, Chilton (1979).

[18] BRENK, H.D., VOGT, V.J., The calculation of wet deposition from radioactive plumes, Nucl. Safety **22** (1981) 362–371.

[19] HEINEMANN, K., VOGT, K.J., Measurements of the deposition of iodine onto vegetation and of the biological halflife of iodine on vegetation, Health Phys. **39** (1980) 463–474.

[20] HORBERT, M., VOGT, K.J., ANGELETTI, L., Untersuchungen zur Ablagerung von Aerosolen auf Vegetation und anderen Grenzflächen, Rep. Jül-1288, Kernforschungsanlage, Jülich (1976).

[21] JONES, J.A., The Estimation of Long Range Dispersion and Deposition of Continuous Releases of Radionuclides to the Atmosphere, Rep. NRPB-R123, National Radiological Protection Board, Chilton (1981).

[22] BAYER, A., Die Radiologische Belastung der Bevölkerung der Rhein-Maas-Region, Rep. V/1647/77D, Commission of the European Communities, Luxembourg (1978).

[23] THUILLIERE, R.H., MANCUSO, R.L., Building Effects on Effluent Dispersion from Roof Vents at Nuclear Power Plants, Rep. EPRI NP-1380, Electric Power Research Institute, Palo Alto, CA (1980).

[24] HOSKER, R.P., "Flow and diffusion near obstacles", Atmospheric Sciences and Power Production (RANDERSON, D., Ed.), Rep. DOE/TIC-27601, Department of Energy, Washington, DC (1984).

[25] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Uncertainty in NCRP Screening Models Relating to Atmospheric Transport, Deposition and Uptake by Humans, NCRP Commentary No. 8, NCRP, Bethesda, MD (1993).

[26] GIFFORD, F.A., Jr., HANNA, S.R., "Dispersion of sulfur dioxide emission from area sources", Power Generation: Air Pollution Monitoring and Control (NOLL, K.E., DAVIS, W.T., Eds), Ann Arbor Science Publishers, Inc., Ann Arbor, MI (1975).

[27] WILSON, D.J., BRITTER, R.E., Estimates of building surface concentrations from nearby point sources, Atmos. Environ. **16** (1982) 2631–2646.

[28] BRIGGS, G.A., Diffusion Estimation for Small Emissions, 1973 Ann. Rep., Environmental Research Laboratories, Oak Ridge, TN (1974).

[29] GEISS, H., NESTER, K., THOMAS, P., VOGT, K.J., in der Bundesrepublik Deutschland experimentell ermittelte Ausbreitungsparameter für 100 m Emissionshöhe, Reps Jül-1707, KfK-3095, Kernforschungsanlage Jülich/Kernforschungszentrum, Karlsruhe (1981).

[30] VOGT, K.J., GEISS, H., Neue Ausbreitungskoeffizienten für 50 und 100 m Emissionshöhe, Internal Rep., Kernforschungsanlage, Jülich (1980).

[31] HÜBSCHMANN, W., NESTER, K., THOMAS, P., Ausbreitungsparameter für Emissionshöhen von 160 m und 195 m, Rep. KfK-2939, Kernforschungszentrum, Karlsruhe (1980) 182.

[32] MILLER, C.W., YILDIRAN, M., Estimating radionuclide air concentrations near buildings: A screening approach, Trans. Am. Nucl. Soc. **46** (1984) 55–57.

[33] HUBER, A.H., Evaluation of a method for estimating pollution concentrations downwind of influencing buildings, Atmos. Environ. **18** (1984) 2313–2338.

[34] SCHWARZ, G., BRENK, H.D., Untersuchungen zum Depositions- und Postdepositionsverhalten von radioaktiven Freisetzungen auf Oberflächen. Teilstudie der Deutschen Risikostudie Phase B, BS/8304/2-1, Aachen (1985). An English version has been published in Proc. Workshop on Methods for Assessing the Off-site Radiological Consequences of Nuclear Accidents, Rep. EUR 10397 EN, Commission of the European Communities, Luxembourg (1985).

[35] HOFFMAN, F.O., BAES, C.F., III (Eds), A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides, Rep. ORNL/NUREG/TM-282, Oak Ridge Natl Lab., TN (1979).

[36] KÖHLER, H., PETERSON, S.-R., HOFFMAN, F.O. (Eds), Multiple Model Testing Using Chernobyl Fallout Data of ^{131}I in Forage and Milk and ^{137}Cs in Forage, Milk,

Beef and Grain, BIOMOVS Tech. Rep. No. 13, Scenario A4, Natl Inst. for Radiation Protection, Stockholm (1991).

[37] GARLAND, J.A., PATTENDEN, N.J., PLAYFORD, K., "Resuspension following Chernobyl", Modelling of Resuspension, Seasonality and Losses during Food Processing, IAEA-TECDOC-647, Vienna (1992).

[38] SEHMEL, G.A., "Deposition and resuspension", Atmospheric Sciences and Power Production (RANDERSON, D., Ed.), Rep. DOE/TIC-27601, United States Department of Energy, Washington, DC (1984).

[39] SIMMONDS, J.R., LAWSON, G., MAYALL, A., Methodology for Assessing the Radiological Consequences of Routine Releases of Radionuclides to the Environment, Rep. DOC XI-5026/94, Commission of the European Communities, Luxembourg (1994).

[40] MILLER, C.W., HIVELY, L.M., A review of validation studies for the Gaussian plume atmospheric dispersion model, Nucl. Safety **28** (1987) 522–531.

[41] RAMSDELL, J.V., Jr., Diffusion in building wakes for ground-level releases, Atmos. Environ. **24** (1990) 377–388.

[42] ONISHI, Y., SERNE, R.J., ARNOLD, E.M, COWAN, C.E., THOMPSON, F.L., Critical Review: Radionuclide Transport, Sediment Transport and Water Quality Mathematical Modelling, and Radionuclide Sorption/Desorption Mechanisms, Rep. NUREG/CR-13220, PNL-2901, Pacific Northwest Natl Lab., Richland, WA (1980).

[43] JIRKA, G.H., FINDIKAKIS, A.N., ONISHI, Y., RYAN, P.J., "Transport of radionuclides in surface waters", Radiological Assessment — A Text Book on Environmental Dose Analysis (TILL, J.E., MEYER, H.R., Eds), Rep. NUREG/CR-3332, ORNL-5968, United States Nuclear Regulatory Commission, Washington, DC (1983).

[44] INTERNATIONAL ATOMIC ENERGY AGENCY, Hydrological Dispersion of Radionuclide Material in Relation to Nuclear Power Plant Siting, Safety Series No. 50-SG-S6, IAEA, Vienna (1985).

[45] ONISHI, Y., "Contaminant transport modeling in surface waters", Computer Modeling of Free-surface and Pressurized Flows (CHAUDHRY, M.H., MAYS, L.W., Eds), NATO ASI Series E, Applied Sciences, Vol. 274, Kluwer, Dordrecht (1994) 313–341.

[46] SAYRE, N.W., "Natural mixing processes in rivers", Environmental Impact on Rivers (River Mechanics II), H.W. Shen, Fort Collins, CO (1973).

[47] UNITED STATES NUCLEAR REGULATORY COMMISSION, Liquid Pathway Generic Study, Impacts of Accidental Radionuclide Releases to the Hydrosphere from Floating and Land-based Nuclear Power Plants, Rep. NUREG-0440, USNRC, Washington, DC (1978).

[48] UNITED STATES NUCLEAR REGULATORY COMMISSION, Estimating Aquatic Dispersion in Effluents from Accidental and Routine Releases for the Purpose of Implementing Appendix I, USNRC Regulatory Guide I.113, Revision I, Office of Standards Development, USNRC, Washington, DC (1977).

[49] LEOPOLD, L.B., WOLMAN, M.G., MILLER, J.P., Fluvial Processes in Geomorphology, W.H. Froeman, San Francisco, CA (1964).

[50] FISHER, H.G., LIST, E.J., KOH, R.C.Y., IMBERGER, J., BROOKS, N.H., Mixing in Inland and Coastal Waters, Academic Press, New York (1979).

[51] OKUBO, A., Oceanic diffusion diagrams, Deep-Sea Res. **18** (1971) 789–802.

[52] BOWIE, G.L., et al., Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling, 2nd edn, Prepared for the US Environmental Protection Agency by Tetra Tech, Lafayette, CA, and Humbolt State University, Arcata, CA (1985).

[53] VOITSEKHOVITCH, O.V., "Hydrologic processes and their influence on radionuclide behaviour and transport by surface water pathways as applied to water protection after the Chernobyl Accident", Hydrological Impact of Nuclear Power Plant Systems (Proc. UNESCO Workshop Paris, 1992), UNESCO Press, Paris (1993) 83–105.

[54] WALTERS, W.H., ECBER, R.M., ONISHI, Y., Sediment and Radionuclide Transport in Rivers — Summary, Field Sampling Program for Catarangus and Buttermilk Creeks, New York, Rep. NUREG/CR-1030, PNL-3117, Vol. 4, Pacific Northwest Natl Lab., Richland, WA (1982).

[55] INTERNATIONAL ATOMIC ENERGY AGENCY, Sediment K_ds and Concentration Factors for Radionuclides in the Marine Environment, Technical Reports Series No. 247, IAEA, Vienna (1985).

[56] ONISHI, Y., Pacific Northwest Natl Lab., Richland, WA, personal communication, 1999.

[57] ONISHI, Y., TRENT, D.S., "Three-dimensional simulation of flow, salinity, sediment, and radionuclide movements in the Hudson River Estuary", Proc. 1985 Specialty Conf. of the Hydraulics Division, American Society of Civil Engineers, Lake Buena Vista, FL (1985), ASCE, New York, 1095–1100.

[58] ONISHI, Y., "Transport and fate models", Principles of Health Risk Assessment (RICCI, P.F., Ed.), Prentice-Hall, Englewood Cliffs, NJ (1985) 155–234.

[59] DICKSON, D.M.J., A Study of the Radiological Implications of Releases of Radionuclides to Sewer Systems, Rep. DoE/HMIP/RR/94/002, United Kingdom Department of the Environment, London (1994).

[60] ATWOOD, C.A., TITLEY, J.G., SIMMONDS, J.R., ROBINSON, C.A., "Revised generalised derived limits for radioisotopes of strontium, ruthenium, iodine, caesium, plutonium, americium and curium", Docs. of the Nuclear Radiological Protection Board, Vol. 9, No. 1, HMSO, London (1998).

[61] INTERNATIONAL ATOMIC ENERGY AGENCY, Modelling of Radionuclide Interception and Loss Processes in Vegetation and of Transfer in Semi-natural Ecosystems, IAEA-TECDOC-857, Vienna (1996).

[62] HEINONEN, R., Das Volumgewicht als Kennzeichen der normalen Bodenstruktur, J. Sci. Agric. Soc. Finland **32** (1960) 81.

[63] GROGAN, H.A., (Ed.), Scenario B2: Irrigation with Contaminated Groundwater, BIOMOVS Tech. Rep. No. 6, Swedish National Institute for Radiation Protection, Stockholm (1989).

[64] INTERNATIONAL UNION OF RADIOECOLOGISTS, 7th Rep. of the IUR Working Group on Soil–Plant Relations, IUR, Balen (1990).

[65] HANSEN, H.S., HOVE, K., Radiocaesium bioavailability: Transfer of Chernobyl and tracer radiocaesium to goat milk, Health Phys. **60** (1991) 665–673.

[66] BELLI, M., et al., Ingested soil as a source of ^{137}Cs to ruminants, Sci. Total Environ. **136** (1993) 243–249.

[67] CROUT, N.M.J., BERESDORF, N.A., HOWARD, B.J., Does soil adhesion matter when predicting radiocaesium transfer to animals, J. Environ. Radioact. **20** (1993) 201–212.

[68] VANDERPLOEG, H.A., PARZYCK, D.C., WILCOX, W.H., KERCHER, J.R., KAYE S.V., Bioaccumulation Factors for Radionuclides in Freshwater Biota, Rep. ORNL-5002, ESD Pub. No. 783, Oak Ridge Natl Lab., TN (1975).

[69] BLAYLOCK, B.G., Radionuclide data bases available for bioacumulation factors for freshwater biota, Nucl. Safety **23** (1982) 427–438.

[70] KILLOUGH, G.G., MCKAY, L., BIORAD Bioaccumulation Factors for Freshwater Biota, Rep. ORNL-4992, Oak Ridge Natl Lab., TN (1976).

[71] THOMPSON, S.E., BURTON, C.A., QUINN, D.J., NG, Y.C., Concentration Factors of Chemical Elements in Edible Aquatic Organisms, Rep. UCRL-50567, Lawrence Livermore Natl Lab., Livermore, CA (1972).

[72] NEWMAN, G., Concentration Factors for Stable Metals and Radionuclides in Fish, Mussels, and Crustaceans: A Literature Survey, Rep. SNUPM 1976E, National Swedish Environmental Protection Board, Solna (1985).

[73] COPELAND, R.A., BEETHE, R.H., PRATER, W.W., Trace Element Distributions in Lake Michigan Fish: A Baseline Study with Calculations of Concentration Factors and Equilibrium Radioiostope Distributions, Special Report No. 2, Environmental Research Group, Ann Arbor, MI (1973).

[74] POSTON, T.M., KLOPFER, D.C., A Literature Review of the Concentration Ratios of Selected Radionuclides in Freshwater and Marine Fish, Rep. PNL-5484, Pacific Northwest Natl Lab., Richland, WA (1988).

[75] ECKERMAN, K.F., RYMAN, J.C., External Exposure to Radionuclides in Air, Water and Soil, Federal Guidance Report No. 12, US Environmental Protection Agency, Washington, DC (1993).

[76] KOCHER, D.C., Dose Rate Conversion Factors for External Exposure to Photons and Electrons, Rep. ORNL/NUREG-79 (NUREG/CR-1918), Oak Ridge Natl Lab., TN (1981).

[77] JACOB, P., ROSENBAUM, H., PETOUSSI, N., ZANKL, M., Calculation of Organ Doses from Environmental Gamma Rays Using Human Phantoms and Monte Carlo Methods, Part II: Radionuclides Distributed in the Air or Deposited on the Ground, Rep. 12/90, Gesellschaft für Strahlen-und Umweltforschung, Neuherberg (1990).

[78] UNITED NATIONS, Sources and Effects of Ionising Radiation (Report to the General Assembly), Scientific Committee of the Effects of Atomic Radiation (UNSCEAR), UN, New York (1982).

[79] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Age-dependent Doses to Members of the Public from Intake of Radionuclides, Part 1, ICRP Publication 56, Pergamon Press, Oxford and New York (1989).

[80] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Age-dependent Doses to Members of the Public from Intake of Radionuclides, Part 2, Ingestion Dose Coefficients, ICRP Publication 67, Pergamon Press, Oxford and New York (1993).

[81] NORTHROP, R., et al., "Health effects of aerosols emitted from an activated sludge plant", Waste Water Aerosols and Disease (Proc. Symp. Cincinnati, 1980), Environmental Protection Agency Publication No. EPA-600/9-80-028, Health Effects Research Lab., Cincinnati, OH (1980).

[82] INTERNATIONAL ATOMIC ENERGY AGENCY, The Radiological Impact of Radionuclides Dispersed on a Regional and Global Scale: Methods for Assessment and their Application, Technical Reports Series No. 250, IAEA, Vienna (1985).

[83] UNITED NATIONS, Dose Assessment Methodologies, Draft R602, Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), UN, New York (1999).

[84] BEXON, A.P., Radiological Impact of Routine Discharges from UK Civil Nuclear Sites in the Mid 1990s, Rep. NRPB-R312, National Radiological Protection Board, Chilton (1999).

Annex I

SCREENING DOSE CALCULATION FACTORS

This annex provides two types of dose[1] calculation factors for estimating hypothetical critical group doses. The first type, no dilution screening factors, are intended to be used in conjunction with maximum predicted annual average radionuclide concentrations (at the point of discharge) in either air or water. This is the simplest and most pessimistic method included in the iterative dose assessment procedure illustrated in Fig. 1. Consistent with this procedure, further effort to refine the dose assessment may be warranted if the calculated screening dose exceeds the relevant dose criterion (e.g. dose constraint). In this case the dilution and dispersion of radionuclides in the environment should be taken into account, and appropriate generic methods are presented in this report for this purpose.

The no dilution and generic environmental dose calculation screening factors presented in this annex were calculated using the methods and data presented in this Safety Report. The generic factors were derived using additional standardized assumptions about the discharge characteristics and location of a hypothetical critical group. These generic dose calculation factors are intended to be a general guide only — it is generally preferable to take account of the actual discharge conditions and the location of the actual critical group when setting or assessing discharge limits for a particular site. In particular, it is recommended that the site specific discharge conditions and the actual critical group location be taken into account if the predicted doses exceed a reference level of around 10% of the dose constraint.

I–1. SCREENING FACTORS (MAXIMUM ANNUAL DOSE PER UNIT DISCHARGE CONCENTRATION)

Screening no dilution dose calculation factors for atmospheric and surface water discharges have been derived using the methodologies, default parameter values and dose coefficients presented in this report. These factors represent the maximum effective doses in the 30th year of discharge that could be received by a hypothetical critical group member from external and internal radiation from a unit concentration of a radionuclide discharged into the air or water.

---

[1] In this annex 'dose' refers to the effective dose received during the 30th year of discharge. Internal doses are calculated using committed effective dose coefficients from the BSS and are presented in Section 6 of this Safety Report [1].

These screening factors are calculated on the basis of the extremely pessimistic assumption that the hypothetical critical group is exposed at the point of discharge. Doses may be estimated by multiplying a screening factor by the initial (undiluted) annual average concentration of the radionuclide at the point of discharge (C_0, Bq/m^3) using either Eq. (1) for discharges to the atmosphere or Eq. (11) for discharges into water (e.g. a river). This undiluted concentration in air or water (C_0) may then be multiplied by the appropriate screening factor from tables given in this annex to give a screening estimate of the maximum annual dose to a hypothetical critical group member. Screening factors for discharges to the atmosphere are presented in Table I–I and for discharges into surface waters (e.g. a river) in Table I–II.

TABLE I–I. SCREENING DOSE CALCULATION FACTORS FOR DISCHARGES TO THE ATMOSPHERE BASED ON THE NO DILUTION APPROACH (Sv/a per Bq/m^3)[a]

Nuclide	Screening dose (Sv/a per Bq/m^3)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
Ac-228	5.9×10^{-5}	0	0	0	5	95
Ag-110m	1.1×10^{-2}	25	1	4	70	0
Am-241*	2.1×10^{-1}	57	0	0	1	42
As-76	1.1×10^{-5}	0	20	0	65	15
At-211*	2.4×10^{-4}	0	4	0	0	96
Au-198	2.3×10^{-5}	32	2	1	59	7
Bi-206	5.2×10^{-4}	28	27	1	44	1
Bi-210	3.1×10^{-4}	27	37	1	1	34
Bi-212*	6.6×10^{-5}	0	0	0	2	98
Br-82	2.0×10^{-4}	0	78	0	22	1
Cd-109	1.8×10^{-2}	13	86	0	1	0
Ce-141	6.7×10^{-4}	88	7	0	4	1
Ce-144	9.1×10^{-3}	86	7	0	6	1
Cm-242	2.1×10^{-2}	69	0	0	0	31
Cm-244*	1.3×10^{-1}	55	0	0	0	45
Co-58	4.4×10^{-3}	16	42	24	17	0
Co-60	1.1×10^{-1}	6	27	19	48	0
Cr-51	7.5×10^{-5}	32	2	53	13	0
Cs-134*	4.7×10^{-2}	24	21	31	24	0
Cs-135*	4.4×10^{-3}	31	8	61	0	0
Cs-136	1.6×10^{-3}	30	26	24	19	0
Cs-137*	4.6×10^{-2}	18	5	34	43	0
Cu-64	2.3×10^{-6}	0	37	0	54	9
Eu-154	4.3×10^{-2}	6	0	0	94	0

TABLE I–I. (cont.)

Nuclide	Screening dose (Sv/a per Bq/m³)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
Eu-155	1.7×10^{-3}	27	0	1	70	0
Fe-55	1.2×10^{-3}	42	3	55	0	0
Fe-59	4.2×10^{-3}	42	3	41	14	0
Ga-67	8.7×10^{-6}	30	1	0	65	4
Hg-197	9.1×10^{-6}	17	56	1	22	5
Hg-197m	4.8×10^{-6}	0	60	0	22	18
Hg-203	2.2×10^{-3}	70	10	14	6	0
I-123	1.2×10^{-5}	0	88	0	9	2
I-125	4.0×10^{-2}	22	56	22	0	0
I-129	2.3×10^{-1}	20	53	27	0	0
I-131	3.7×10^{-2}	12	80	8	0	0
I-132	3.7×10^{-6}	0	0	0	91	9
I-133	6.0×10^{-4}	0	98	0	1	1
I-134	2.3×10^{-6}	0	0	0	94	6
I-135	1.4×10^{-5}	0	51	0	39	9
In-111	1.8×10^{-5}	12	13	0	73	2
In-113m*	3.4×10^{-7}	0	0	0	88	12
Mn-54	3.7×10^{-3}	18	2	1	79	0
Mo-99	1.3×10^{-4}	3	88	0	7	1
Na-22	3.3×10^{-1}	1	70	22	7	0
Na-24	4.4×10^{-4}	0	93	0	6	0
Nb-95	6.9×10^{-4}	56	0	0	44	0
Ni-59	1.1×10^{-2}	2	96	2	0	0
Ni-63	2.6×10^{-2}	2	96	2	0	0
Np-237*	1.8×10^{-1}	46	0	16	11	27
Np-239	1.1×10^{-5}	29	15	1	42	14
P-32	1.1×10^{-2}	10	82	8	0	0
Pa-231*	7.4×10^{-1}	60	0	0	0	40
Pa-233	7.0×10^{-4}	91	0	0	9	1
Pb-210	9.3×10^{-1}	91	7	2	0	0
Pd-103	1.0×10^{-4}	93	3	0	2	1
Pd-107	1.1×10^{-4}	96	3	1	0	1
Pd-109	1.2×10^{-6}	0	20	0	22	58
Pm-147	4.3×10^{-4}	92	1	5	0	1
Po-210	3.0	53	41	6	0	0
Pu-238*	2.3×10^{-1}	58	0	0	0	42
Pu-239*	2.5×10^{-1}	58	0	0	0	41
Pu-240*	2.5×10^{-1}	58	0	0	0	41
Pu-241*	4.8×10^{-3}	59	0	0	1	40

TABLE I–I. (cont.)

Nuclide	Screening dose (Sv/a per Bq/m^3)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
Pu-242*	2.4×10^{-1}	58	0	0	0	41
Ra-224	1.1×10^{-2}	19	51	1	1	28
Ra-225	1.1×10^{-1}	63	25	5	0	7
Ra-226	5.7×10^{-1}	47	16	8	27	2
Rb-86	5.6×10^{-2}	3	96	1	0	0
Rh-105	6.1×10^{-6}	1	67	0	22	10
Rh-107*	2.2×10^{-7}	0	0	0	84	16
Ru-103	1.4×10^{-3}	43	0	41	15	0
Ru-106	2.4×10^{-2}	42	0	52	6	0
S-35	1.1×10^{-2}	9	49	42	0	0
Sb-124	4.1×10^{-3}	60	4	6	30	0
Sb-125	6.6×10^{-3}	19	1	3	77	0
Se-75	9.0×10^{-3}	26	7	61	6	0
Sn-113	1.7×10^{-3}	53	13	12	21	0
Sr-85	1.4×10^{-3}	35	31	8	26	0
Sr-87m*	5.9×10^{-7}	0	0	0	92	7
Sr-89	5.5×10^{-3}	47	41	11	1	0
Sr-90	1.7×10^{-1}	18	60	20	2	0
Tc-99	6.6×10^{-3}	53	43	4	0	0
Tc-99m	4.4×10^{-7}	0	2	0	91	7
Te-125m	4.0×10^{-3}	25	35	39	1	0
Te-127m	1.5×10^{-2}	23	34	42	0	0
Te-129m	1.1×10^{-2}	26	38	36	0	0
Te-131m	2.0×10^{-4}	0	87	0	11	1
Te-132	1.4×10^{-3}	5	86	2	7	0
Th-228*	4.9×10^{-1}	84	0	0	2	14
Th-230*	3.7×10^{-1}	33	0	0	42	24
Th-232*	4.4×10^{-1}	30	0	0	48	21
Tl-201	1.6×10^{-5}	6	74	1	19	1
Tl-202	3.4×10^{-4}	31	40	9	19	0
U-232*	4.3×10^{-1}	48	3	4	26	18
U-234*	2.1×10^{-1}	15	1	1	74	9
U-235*	6.7×10^{-2}	44	3	4	22	27
U-238*	2.2×10^{-1}	13	1	1	77	8
Y-87	4.0×10^{-5}	19	4	1	74	2
Y-90	3.4×10^{-5}	56	22	3	10	9
Y-91	3.4×10^{-3}	80	1	17	1	0
Zn-65	3.2×10^{-2}	15	28	53	5	0
Zr-95	2.0×10^{-3}	44	0	0	55	0

[a] Infants are the limiting exposure group unless otherwise noted by an asterisk, in which case adults are the limiting group.

I–2.   GENERIC FACTORS (DOSE PER UNIT DISCHARGE)

Generic dose calculation factors, based on the generic environmental model presented in this report, are also presented in this annex. These factors take account of the dispersion of radionuclides in the environment, and are therefore of a less pessimistic nature than the screening values given above. In deriving these generic factors care has been taken to ensure that doses to a hypothetical critical group are unlikely to exceed those received by an actual critical group by more than a factor of ten. The result is that, in reality, these generic factors are still likely to represent overestimates of the doses received. The hypothetical critical group doses from unit discharges of radionuclides to the atmospheric and surface water environments have been calculated using a few standardized assumptions about the discharge conditions and the location of the critical group. The models and data used in these calculations are described in detail in the main text. The assumptions applied here are similar to those used to calculate clearance levels in Ref. [I–1], and have been described in more detail there. The main underlying assumptions are summarized here for ease of reference.

I–2.1.   Atmospheric discharges

Annual average doses to a member of a hypothetical critical group, corresponding to a continuous discharge of 1 Bq/s over a 30 year period, are presented in Table I–III. An atmospheric discharge was assumed to occur from a pipe with diameter 0.5 m, located on the side of a building with a cross-sectional area of 500 m^2. A hypothetical critical group member is assumed to live at a distance of 20 m from the source. In terms of atmospheric dispersion, this receptor point is within the cavity zone, as explained in the main text. An annual geometric mean wind speed of 2 m/s was assumed, and the fraction of the year for which the wind blows in the sector of interest was assumed to be 0.25. The average air concentration at 20 m corresponding to a discharge rate of 1 Bq/s under these dispersion conditions is 2×10^{-3} Bq/m^3. A mean deposition velocity of 1000 m/d was assumed in order to estimate ground deposition levels. At 20 m the average deposition rate on the ground is 2 Bq·m^{-2}·d^{-1}.

It is assumed that terrestrial foods are produced at a greater distance from the source; crops from a distance of 100 m, and milk and meat from 800 m. These locations are within the wake zone for dispersion purposes, as explained in the main text. The ground deposition rates for these locations are 1.3×10^{-1} and 7.5×10^{-3} Bq·m^{-2}·d^{-1}, respectively.

The exposure pathways considered in these calculations are as follows: inhalation and external exposure from immersion in the cloud (at 20 m), external exposure from material deposited on the ground (at 20 m) and ingestion of crops (from 100 m) and milk and meat (from 800 m). Effective doses were calculated for

TABLE I–II. SCREENING DOSE CALCULATION FACTORS FOR DISCHARGES INTO SURFACE WATERS BASED ON THE NO DILUTION APPROACH (Sv/a per Bq/m^3)[a]

Nuclide	Screening dose (Sv/a per Bq/m^3)	Contribution (%)		
		Drinking water	Fish	External dose
Ac-228[b]	1.5×10^{-9}	48	42	10
Ag-110m*[b]	1.3×10^{-8}	12	3	84
Am-241*	3.0×10^{-7}	39	59	1
As-76[b]	8.5×10^{-8}	3	97	0
At-211[b]	3.8×10^{-8}	54	46	0
Au-198[b]	5.7×10^{-9}	33	67	0
Bi-206[b]	4.2×10^{-9}	62	36	2
Bi-210[b]	4.0×10^{-9}	63	37	0
Bi-212[b]	7.4×10^{-10}	63	37	0
Br-82[b]	1.6×10^{-8}	4	96	0
Cd-109[b]	3.4×10^{-8}	7	85	8
Ce-141	5.1×10^{-9}	26	45	29
Ce-144	4.6×10^{-8}	22	38	40
Cm-242	5.4×10^{-8}	37	63	0
Cm-244	2.1×10^{-7}	37	63	0
Co-58	4.5×10^{-8}	3	44	53
Co-60*	3.4×10^{-7}	1	9	90
Cr-51	1.3×10^{-9}	5	54	41
Cs-134*	5.8×10^{-6}	0	99	1
Cs-135*	6.0×10^{-7}	0	100	0
Cs-136	1.4×10^{-6}	0	100	0
Cs-137*	3.9×10^{-6}	0	99	0
Cu-64[b]	2.7×10^{-9}	8	92	0
Eu-154	2.5×10^{-8}	12	36	52
Eu-155	2.8×10^{-9}	20	58	22
Fe-55	7.8×10^{-9}	8	92	0
Fe-59	6.0×10^{-8}	6	65	30
Ga-67[b]	7.5×10^{-9}	4	96	0
Hg-197[b]	2.4×10^{-8}	2	98	0
Hg-197m[b]	5.2×10^{-8}	2	98	0
Hg-203[b]	1.7×10^{-7}	2	98	0
I-123	1.6×10^{-9}	30	70	0
I-125	4.9×10^{-8}	30	70	0
I-129*	2.0×10^{-7}	33	67	0
I-131	1.5×10^{-7}	30	70	0
I-132	2.1×10^{-9}	30	70	0
I-133	3.8×10^{-8}	30	70	0

TABLE I–II. (cont.)

Nuclide	Screening dose $(Sv/a \text{ per } Bq/m^3)$	Contribution (%)		
		Drinking water	Fish	External dose
I-134	6.5×10^{-10}	30	70	0
I-135	7.7×10^{-9}	30	70	0
In-111[b]	2.6×10^{-7}	0	100	0
In-113m[b]	2.7×10^{-8}	0	100	0
Mn-54	3.2×10^{-8}	3	59	39
Mo-99[b]	1.9×10^{-9}	48	27	25
Na-22*[b]	6.4×10^{-8}	3	3	94
Na-24[b]	1.5×10^{-9}	40	46	13
Nb-95*[b]	7.7×10^{-8}	0	7	93
Ni-59[b]	6.0×10^{-10}	15	85	0
Ni-63[b]	1.5×10^{-9}	15	85	0
Np-237*	1.7×10^{-7}	40	60	0
Np-239	4.0×10^{-9}	37	63	0
P-32	1.4×10^{-5}	0	100	0
Pa-231*[b]	6.4×10^{-7}	66	33	1
Pa-233*[b]	4.5×10^{-9}	36	21	43
Pb-210[b]	1.7×10^{-5}	5	95	0
Pd-103[b]	7.0×10^{-10}	52	30	19
Pd-107[b]	1.1×10^{-10}	63	37	0
Pd-109[b]	1.7×10^{-9}	63	36	1
Pm-147	1.4×10^{-9}	37	63	0
Po-210[b]	8.9×10^{-6}	26	74	0
Pu-238*	3.5×10^{-7}	40	60	0
Pu-239*	3.8×10^{-7}	40	60	0
Pu-240*	3.8×10^{-7}	40	60	0
Pu-241*	7.7×10^{-9}	37	56	7
Pu-242*	3.6×10^{-7}	40	60	0
Ra-224	6.7×10^{-7}	26	74	0
Ra-225	1.2×10^{-6}	26	74	0
Ra-226	9.9×10^{-7}	25	73	2
Rb-86[b]	6.1×10^{-7}	1	99	0
Rh-105[b]	1.1×10^{-9}	61	35	4
Rh-107[b]	6.8×10^{-11}	62	36	3
Ru-103	2.7×10^{-9}	44	26	30
Ru-106	2.3×10^{-8}	56	32	12
S-35[b]	6.6×10^{-8}	2	98	0
Sb-124	2.9×10^{-8}	15	84	2
Sb-125	1.1×10^{-8}	14	82	4
Se-75[b]	4.2×10^{-8}	8	92	0

TABLE I–II. (cont.)

Nuclide	Screening dose (Sv/a per Bq/m^3)	Contribution (%)		
		Drinking water	Fish	External dose
Sn-113[b]	8.9×10^{-9}	15	84	1
Sr-85	7.0×10^{-9}	11	50	39
Sr-87m	2.4×10^{-10}	19	80	1
Sr-89	2.5×10^{-8}	19	80	1
Sr-90	1.0×10^{-7}	18	79	2
Tc-99	2.7×10^{-9}	46	54	0
Tc-99m	7.3×10^{-11}	46	54	0
Te-125m[b]	3.9×10^{-8}	4	96	0
Te-127m[b]	1.1×10^{-7}	4	96	0
Te-129m[b]	1.5×10^{-7}	4	96	0
Te-131m[b]	8.8×10^{-8}	4	96	0
Te-132b	1.9×10^{-7}	4	96	0
Th-228*	2.9×10^{-6}	15	75	10
Th-230*	1.2×10^{-6}	11	54	35
Th-232*	1.4×10^{-6}	10	50	40
Tl-201b	8.6×10^{-9}	2	96	2
Tl-202b	3.5×10^{-8}	2	89	10
U-232	3.4×10^{-7}	63	36	0
U-234	5.5×10^{-8}	61	35	4
U-235	5.3×10^{-8}	63	36	0
U-238	5.1×10^{-8}	61	35	4
Y-87[b]	3.0×10^{-9}	27	47	25
Y-90[b]	1.4×10^{-8}	36	63	1
Y-91[b]	1.4×10^{-8}	33	58	9
Zn-65	2.5×10^{-7}	2	97	2
Zr-95	3.5×10^{-8}	4	73	23

[a] Infants are the limiting exposure group unless otherwise noted by an asterisk, in which case adults are the limiting group.

[b] Indicates those radionuclides for which K_d values are not available and for which K_d estimates have been made on the basis of chemical similarities.

both adults and infants. The dose to the age group receiving the higher dose is presented in Table I–III. The habit assumptions used in these calculations are given in Table XIV and the effective dose coefficients for external and internal doses are given in Tables XV–XVII.

TABLE I–III. DOSE CALCULATION FACTORS FOR DISCHARGES TO THE ATMOSPHERE BASED ON THE GENERIC ENVIRONMENTAL MODEL (Sv/a per Bq/s)[a]

Nuclide	Generic dose factor (Sv/a per Bq/s)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
Ac-228	4.7×10^{-7}	0	0	0	5	95
Ag-110m	6.3×10^{-5}	2	0	0	98	0
Am-241*	7.9×10^{-4}	8	0	0	2	90
As-76	6.9×10^{-8}	0	0	0	81	19
At-211*	1.8×10^{-6}	0	0	0	0	100
Au-198	1.2×10^{-7}	3	0	0	87	10
Bi-206	1.9×10^{-6}	4	0	0	95	1
Bi-210*	1.6×10^{-6}	1	0	0	1	98
Bi-212*	5.3×10^{-7}	0	0	0	2	98
Br-82	3.7×10^{-7}	0	1	0	96	2
Cd-109	2.8×10^{-6}	44	17	0	35	4
Ce-141	5.6×10^{-7}	54	0	0	39	6
Ce-144	9.3×10^{-6}	44	0	0	51	5
Cm-242*	9.1×10^{-5}	4	0	0	0	96
Cm-244*	4.9×10^{-4}	7	0	0	0	92
Co-58	6.7×10^{-6}	6	1	0	93	0
Co-60	4.1×10^{-4}	1	0	0	99	0
Cr-51	9.3×10^{-8}	13	0	1	85	1
Cs-134*	9.8×10^{-5}	6	0	0	93	0
Cs-135*	8.2×10^{-7}	86	1	10	2	1
Cs-136	2.8×10^{-6}	9	0	0	89	1
Cs-137*	1.6×10^{-4}	3	0	0	97	0
Cu-64*	1.2×10^{-8}	0	0	0	83	17
Eu-154	3.2×10^{-4}	0	0	0	99	0
Eu-155	1.0×10^{-5}	2	0	0	97	1
Fe-55	2.9×10^{-7}	90	0	7	0	3
Fe-59	5.6×10^{-6}	16	0	1	82	1
Ga-67*	5.0×10^{-8}	1	0	0	91	8
Hg-197*	2.1×10^{-8}	1	0	0	75	24
Hg-197m*	1.7×10^{-8}	0	0	0	48	52
Hg-203	1.8×10^{-6}	44	0	1	54	1
I-123	1.1×10^{-8}	0	3	0	78	19
I-125	5.7×10^{-6}	79	12	5	4	1
I-129*	3.9×10^{-5}	86	4	6	3	2
I-131	3.8×10^{-6}	61	24	2	7	5
I-132	3.0×10^{-8}	0	0	0	91	9

TABLE I–III. (cont.)

Nuclide	Generic dose factor (Sv/a per Bq/m^3)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
I-133	1.2×10^{-7}	0	15	0	44	42
I-134	1.8×10^{-8}	0	0	0	94	6
I-135	5.4×10^{-8}	0	0	0	80	19
In-111	1.1×10^{-7}	1	0	0	96	3
In-113m*	2.7×10^{-9}	0	0	0	88	12
Mn-54	2.4×10^{-5}	1	0	0	98	0
Mo-99	9.3×10^{-8}	2	4	0	80	14
Na-22	1.9×10^{-4}	1	4	1	94	0
Na-24	2.4×10^{-7}	0	5	0	93	2
Nb-95	2.6×10^{-6}	8	0	0	92	1
Ni-59	4.5×10^{-7}	27	71	2	0	1
Ni-63	1.0×10^{-6}	27	71	2	0	1
Np-237*	5.9×10^{-4}	7	0	0	27	66
Np-239	4.9×10^{-8}	3	0	0	73	24
P-32	1.0×10^{-6}	56	27	3	11	4
Pa-231*	2.6×10^{-3}	9	0	0	1	90
Pa-233	8.5×10^{-7}	39	0	0	57	4
Pb-210	4.7×10^{-4}	94	0	0	4	2
Pd-103	7.6×10^{-8}	67	0	0	26	7
Pd-107	6.0×10^{-8}	90	0	0	0	9
Pd-109*	8.3×10^{-9}	0	0	0	25	75
Pm-147	2.6×10^{-7}	79	0	0	1	19
Po-210	9.0×10^{-4}	94	4	1	0	1
Pu-238*	8.4×10^{-4}	8	0	0	0	92
Pu-239*	9.2×10^{-4}	8	0	0	0	92
Pu-240*	9.2×10^{-4}	8	0	0	0	92
Pu-241*	1.7×10^{-5}	9	0	0	2	89
Pu-242*	8.8×10^{-4}	8	0	0	0	92
Ra-224*	5.8×10^{-5}	1	0	0	1	99
Ra-225*	1.4×10^{-4}	6	0	0	0	94
Ra-226*	1.5×10^{-3}	7	0	0	82	11
Rb-86	2.7×10^{-6}	29	58	1	11	1
Rh-105*	1.7×10^{-8}	0	0	0	64	35
Rh-107*	1.7×10^{-9}	0	0	0	84	16
Ru-103	2.0×10^{-6}	15	0	1	83	1
Ru-106	1.8×10^{-5}	29	0	2	65	4
S-35	8.1×10^{-7}	62	20	17	0	2
Sb-124	1.1×10^{-5}	11	0	0	88	1
Sb-125	4.1×10^{-5}	2	0	0	98	0

TABLE I–III. (cont.)

Nuclide	Generic dose factor (Sv/a per Bq/m^3)	Contribution (%)				
		Crops	Milk	Meat	External	Inhalation
Se-75	5.6×10^{-6}	22	0	3	74	0
Sn-113	3.4×10^{-6}	14	0	0	85	1
Sr-85	3.2×10^{-6}	8	0	0	91	0
Sr-87m*	4.7×10^{-9}	0	0	0	93	7
Sr-89	1.8×10^{-6}	74	4	1	17	5
Sr-90*	5.1×10^{-5}	31	2	2	60	5
Tc-99	1.9×10^{-6}	95	4	0	0	0
Tc-99m	3.4×10^{-9}	0	0	0	93	7
Te-125m	8.2×10^{-7}	64	5	6	24	1
Te-127m	2.5×10^{-6}	75	6	8	9	2
Te-129m	2.0×10^{-6}	76	6	6	9	2
Te-131m	2.1×10^{-7}	0	2	0	87	10
Te-132	8.7×10^{-7}	4	4	0	86	6
Th-228*	8.4×10^{-4}	25	0	0	11	64
Th-230*	2.0×10^{-3}	3	0	0	61	36
Th-232*	2.5×10^{-3}	3	0	0	67	30
Tl-201	2.6×10^{-8}	2	1	0	93	3
Tl-202	5.8×10^{-7}	9	1	0	89	1
U-232*	1.6×10^{-3}	7	0	0	55	38
U-234*	1.4×10^{-3}	1	0	0	88	11
U-235*	2.8×10^{-4}	6	0	0	43	52
U-238*	1.5×10^{-3}	1	0	0	90	9
Y-87	2.5×10^{-7}	2	0	0	96	3
Y-90	6.2×10^{-8}	16	0	0	44	40
Y-91	1.9×10^{-6}	73	0	1	21	5
Zn-65	1.6×10^{-5}	16	2	3	79	0
Zr-95	9.1×10^{-6}	5	0	0	94	1

[a] Infants are the limiting exposure group unless otherwise noted by an asterisk, in which case adults are the limiting group.

I–2.2. Liquid discharges

Two discharge conditions were considered: the discharge of material into a sewerage system and directly into a river. Hypothetical critical group[2] annual average doses from a unit discharge rate from each mode of discharge are presented in Tables I–IV and I–V. The assumptions made in each case are described in turn.

I–2.2.1. Discharge into a sewerage system

In this scenario material is assumed to be discharged into the sewerage system at a rate of 1 Bq/a and retained in the sewage sludge. The maximum annual average activity concentration in sludge was calculated by assuming an annual sewage production of 400 t/a (dry weight) as 2.5×10^{-6} Bq/kg per Bq/a discharged.

Two exposure pathways were considered, both of which arise within the sewage plant itself — external irradiation from radionuclides in the sludge and inhalation of radionuclides resuspended into the air from the sludge.

The hypothetical maximum annual external dose from radionuclides at the surface of a container full of sewage sludge was calculated. The maximum annual surface activity concentration was estimated from the total sludge concentration by assuming a density of 1×10^3 kg/m^3 and a sludge container depth of 1 m. Workers within a sewage plant were assumed to be exposed for a working year of 2000 h/a (or 0.288 of the year). The annual dose factors for external dose from surface deposits, given in Table XV, were used to calculate external dose.

Maximum committed effective doses from inhalation of resuspended particles during the working year were calculated using the dose coefficients for inhalation (for adults) given in Table XVI. An occupancy factor of 0.288 was applied, as explained above. An ambient dust loading of 1×10^{-7} kg/m^3 was assumed as described in Ref. [I–1]. An average adult inhalation rate of 8400 m^3/a, given in Table XIV, was used to estimate the inhalation dose.

The maximum annual committed effective inhalation and external doses resulting from an annual discharge into the sewerage system were summed for each radionuclide and are presented in Table I–IV.

---

[2] The doses calculated for discharges into the sewerage system are those to a hypothetical person working in the sewerage plant. As such, they are intended to represent the maximum dose likely to be experienced as a result of the discharge.

TABLE I–IV. DOSE CALCULATION FACTORS FOR DISCHARGES INTO A
SEWER BASED ON THE GENERIC ENVIRONMENTAL MODEL (Sv/a per Bq/a)[a]

Nuclide	Total dose (Sv/a per Bq/a)	Contribution (%)	
		External	Inhalation
Ac-228	8.8×10^{-13}	100	0
Ag-110m	2.4×10^{-12}	100	0
Am-241	2.6×10^{-14}	96	4
As-76	4.9×10^{-13}	100	0
At-211	3.7×10^{-14}	100	0
Au-198	3.8×10^{-13}	100	0
Bi-206	2.9×10^{-12}	100	0
Bi-210	3.2×10^{-14}	100	0
Bi-212	1.2×10^{-12}	100	0
Br-82	2.3×10^{-12}	100	0
Cd-109	2.1×10^{-14}	100	0
Ce-141	6.8×10^{-14}	100	0
Ce-144	1.6×10^{-13}	100	0
Cm-242	1.1×10^{-15}	88	12
Cm-244	1.5×10^{-15}	57	43
Co-58	8.6×10^{-13}	100	0
Co-60	2.1×10^{-12}	100	0
Cr-51	2.8×10^{-14}	100	0
Cs-134	1.4×10^{-12}	100	0
Cs-135	3.1×10^{-17}	100	0
Cs-136	1.9×10^{-12}	100	0
Cs-137	5.1×10^{-13}	100	0
Cu-64	1.7×10^{-13}	100	0
Eu-154	1.1×10^{-12}	100	0
Eu-155	5.4×10^{-14}	100	0
Fe-55	1.8×10^{-20}	0	100
Fe-59	1.0×10^{-12}	100	0
Ga-67	1.4×10^{-13}	100	0
Hg-197	5.9×10^{-14}	100	0
Hg-197m	7.9×10^{-14}	100	0
Hg-203	2.1×10^{-13}	100	0
I-123	1.5×10^{-13}	100	0
I-125	3.9×10^{-14}	100	0
I-129	2.4×10^{-14}	100	0
I-131	3.4×10^{-13}	100	0
I-132	2.1×10^{-12}	100	0
I-133	5.8×10^{-13}	100	0
I-134	2.4×10^{-12}	100	0
I-135	1.4×10^{-12}	100	0

TABLE I–IV. (cont.)

Nuclide	Total dose (Sv/a per Bq/a)	Contribution (%)	
		External	Inhalation
In-111	3.6×10^{-13}	100	0
In-113m	2.3×10^{-13}	100	0
Mn-54	7.4×10^{-13}	100	0
Mo-99	2.6×10^{-13}	100	0
Na-22	1.9×10^{-12}	100	0
Na-24	3.3×10^{-12}	100	0
Nb-95	6.8×10^{-13}	100	0
Ni-59	4.3×10^{-21}	0	100
Ni-63	1.1×10^{-20}	0	100
Np-237	2.0×10^{-13}	100	0
Np-239	1.5×10^{-13}	100	0
P-32	7.7×10^{-14}	100	0
Pa-231	4.1×10^{-14}	92	8
Pa-233	1.8×10^{-13}	100	0
Pb-210	3.4×10^{-14}	100	0
Pd-103	1.2×10^{-14}	100	0
Pd-107	1.4×10^{-20}	0	100
Pd-109	3.6×10^{-14}	100	0
Pm-147	3.2×10^{-17}	100	0
Po-210	2.2×10^{-17}	34	66
Pu-238	1.9×10^{-15}	43	57
Pu-239	1.6×10^{-15}	23	77
Pu-240	2.0×10^{-15}	40	60
Pu-241	8.7×10^{-16}	98	2
Pu-242	1.8×10^{-15}	37	63
Ra-224	1.3×10^{-12}	100	0
Ra-225	2.5×10^{-13}	100	0
Ra-226	1.6×10^{-12}	100	0
Rb-86	1.5×10^{-13}	100	0
Rh-105	7.0×10^{-14}	100	0
Rh-107	3.2×10^{-13}	100	0
Ru-103	4.2×10^{-13}	100	0
Ru-106	3.2×10^{-13}	100	0
S-35	1.6×10^{-17}	100	0
Sb-124	1.6×10^{-12}	100	0
Sb-125	3.9×10^{-13}	100	0
Se-75	3.4×10^{-13}	100	0
Sn-113	2.5×10^{-13}	100	0
Sr-85	4.6×10^{-13}	100	0
Sr-87m	2.9×10^{-13}	100	0

TABLE I–IV. (cont.)

Nuclide	Total dose (Sv/a per Bq/a)	Contribution (%)	
		External	Inhalation
Sr-89	6.2×10^{-14}	100	0
Sr-90	1.0×10^{-13}	100	0
Tc-99	7.2×10^{-17}	100	0
Tc-99m	1.1×10^{-13}	100	0
Te-125m	3.3×10^{-14}	100	0
Te-127m	2.0×10^{-14}	100	0
Te-129m	5.4×10^{-14}	100	0
Te-131m	1.4×10^{-12}	100	0
Te-132	2.3×10^{-12}	100	0
Th-228	1.3×10^{-12}	100	0
Th-230	1.6×10^{-12}	100	0
Th-232	2.2×10^{-12}	100	0
Tl-201	7.9×10^{-14}	100	0
Tl-202	4.2×10^{-13}	100	0
U-232	1.3×10^{-12}	100	0
U-234	1.6×10^{-12}	100	0
U-235	1.5×10^{-13}	100	0
U-238	1.7×10^{-12}	100	0
Y-87	6.9×10^{-13}	100	0
Y-90	9.9×10^{-14}	100	0
Y-91	6.7×10^{-14}	100	0
Zn-65	5.0×10^{-13}	100	0
Zr-95	1.3×10^{-12}	100	0

[a] Adults are the only exposure group considered for this pathway.

I–2.2.2. Discharge into a river

In order to calculate the dose factors presented in Table I–V radionuclides are assumed to be discharged into a small river with a flow of 0.1 m^3/s. The hypothetical critical group is assumed to live at a distance of 500 m downstream from the discharge point and on the same side of the river. Other river dimensions assumed are that the river is 3.47 m wide with a depth of 0.058 m. These dimensions are compatible with a river flow velocity of approximately 0.5 m/s and a partial mixing coefficient of 1.6. The corresponding annual activity concentration in water at the distance of 500 m for a continuous discharge of 1 Bq/s during 1 year is 16 Bq/m^3. The river dimensions and the calculation of this activity concentration are explained in more detail in Ref. [I–1]. The concentrations in sediment and filtered water were

TABLE I–V. DOSE CALCULATION FACTORS FOR DISCHARGES INTO SURFACE WATER BASED ON THE GENERIC ENVIRONMENTAL MODEL (Sv/a per Bq/s)[a]

Nuclide	Generic dose factor Sv/a per Bq/s	Contribution (%)		
		Drinking water	Fish	External
Ac-228[b]	2.3×10^{-8}	48	42	10
Ag-110m*[b]	2.2×10^{-7}	38	29	33
Am-241*	4.9×10^{-6}	12	6	82
As-76[b]	1.4×10^{-6}	3	97	0
At-211[b]	5.9×10^{-7}	54	46	0
Au-198[b]	9.0×10^{-8}	33	67	0
Bi-206[b]	6.7×10^{-8}	62	36	2
Bi-210[b]	6.4×10^{-8}	63	37	0
Bi-212[b]	9.8×10^{-9}	63	37	0
Br-82[b]	2.6×10^{-7}	4	96	0
Cd-109[b]	5.4×10^{-7}	7	85	8
Ce-141	8.1×10^{-8}	26	45	29
Ce-144	7.3×10^{-7}	22	38	40
Cm-242	8.6×10^{-7}	37	63	0
Cm-244	3.3×10^{-6}	37	63	0
Co-58	7.2×10^{-7}	3	44	53
Co-60*	5.4×10^{-6}	1	9	90
Cr-51	2.0×10^{-8}	5	54	41
Cs-134*	9.2×10^{-5}	0	99	1
Cs-135*	9.6×10^{-6}	0	100	0
Cs-136	2.3×10^{-5}	0	100	0
Cs-137*	6.3×10^{-5}	0	99	0
Cu-64[b]	4.3×10^{-8}	8	92	0
Eu-154	4.0×10^{-7}	12	36	52
Eu-155	4.5×10^{-8}	20	58	22
Fe-55	1.3×10^{-7}	8	92	0
Fe-59	9.7×10^{-7}	6	65	30
Ga-67[b]	1.2×10^{-7}	4	96	0
Hg-197[b]	3.9×10^{-7}	2	98	0
Hg-197m[b]	8.2×10^{-7}	2	98	0
Hg-203[b]	2.7×10^{-6}	2	98	0
I-123	2.6×10^{-8}	30	70	0
I-125	7.8×10^{-7}	30	70	0
I-129*	3.2×10^{-6}	33	67	0
I-131	2.5×10^{-6}	30	70	0
I-132	3.0×10^{-8}	30	70	0
I-133	6.0×10^{-7}	30	70	0
I-134	8.3×10^{-9}	30	70	0

TABLE I–V. (cont.)

Nuclide	Generic dose factor Sv/a per Bq/s	Contribution (%)		
		Drinking water	Fish	External
I-135	1.2×10^{-7}	30	70	0
In-111[b]	4.1×10^{-6}	0	100	0
In-113m[b]	3.9×10^{-7}	0	100	0
Mn-54	5.1×10^{-7}	3	59	39
Mo-99[b]	3.1×10^{-8}	48	27	25
Na-22*[b]	1.0×10^{-6}	3	3	94
Na-24[b]	2.3×10^{-8}	40	46	13
Nb-95*[b]	1.2×10^{-6}	0	7	93
Ni-59[b]	9.6×10^{-9}	15	85	0
Ni-63[b]	2.4×10^{-8}	15	85	0
Np-237*	2.6×10^{-6}	40	60	0
Np-239	6.5×10^{-8}	37	63	0
P-32	2.3×10^{-4}	0	100	0
Pa-231[b]	8.6×10^{-6}	63	36	1
Pa-233[b]	7.1×10^{-8}	36	21	43
Pb-210[b]	2.7×10^{-4}	5	95	0
Pd-103[b]	1.1×10^{-8}	52	30	19
Pd-107[b]	1.8×10^{-9}	63	37	0
Pd-109[b]	2.7×10^{-8}	63	36	1
Pm-147	2.2×10^{-8}	37	63	0
Po-210[b]	1.4×10^{-4}	26	74	0
Pu-238*	5.5×10^{-6}	40	60	0
Pu-239*	6.0×10^{-6}	40	60	0
Pu-240*	6.0×10^{-6}	40	60	0
Pu-241*	1.2×10^{-7}	37	56	7
Pu-242*	5.8×10^{-6}	40	60	0
Ra-224	1.1×10^{-5}	26	74	0
Ra-225	1.9×10^{-5}	26	74	0
Ra-226	1.6×10^{-5}	25	73	2
Rb-86[b]	9.7×10^{-6}	1	99	0
Rh-105[b]	1.8×10^{-8}	61	35	4
Rh-107[b]	6.3×10^{-10}	62	36	3
Ru-103	4.3×10^{-8}	44	26	30
Ru-106	3.7×10^{-7}	56	32	12
S-35[b]	1.1×10^{-6}	2	98	0
Sb-124	4.6×10^{-7}	15	84	2
Sb-125	1.8×10^{-7}	14	82	4
Se-75[b]	6.8×10^{-7}	8	92	0
Sn-113[b]	1.4×10^{-7}	15	84	1
Sr-85	1.1×10^{-7}	11	50	39

TABLE I–V. (cont.)

Nuclide	Generic dose factor Sv/a per Bq/s	Contribution (%)		
		Drinking water	Fish	External
Sr-87m	3.6×10^{-9}	19	80	1
Sr-89	4.0×10^{-7}	19	80	1
Sr-90	1.7×10^{-6}	18	79	2
Tc-99	4.3×10^{-8}	46	54	0
Tc-99m	1.1×10^{-9}	46	54	0
Te-125m[b]	6.3×10^{-7}	4	96	0
Te-127m[b]	1.8×10^{-6}	4	96	0
Te-129m[b]	2.4×10^{-6}	4	96	0
Te-131m[b]	1.4×10^{-6}	4	96	0
Te-132[b]	3.0×10^{-6}	4	96	0
Th-228*	4.6×10^{-5}	15	75	10
Th-230*	1.9×10^{-5}	11	54	35
Th-232*	2.2×10^{-5}	10	50	40
Tl-201[b]	1.4×10^{-7}	2	96	2
Tl-202[b]	5.7×10^{-7}	2	89	10
U-232	5.4×10^{-6}	63	36	0
U-234	8.9×10^{-7}	61	35	4
U-235	8.6×10^{-7}	63	36	0
U-238	8.2×10^{-7}	61	35	4
Y-87[b]	4.9×10^{-8}	27	47	25
Y-90[b]	2.3×10^{-7}	36	63	1
Y-91[b]	2.3×10^{-7}	33	58	9
Zn-65	4.0×10^{-6}	2	97	2
Zr-95	5.5×10^{-7}	4	73	23

[a] Infants are the limiting exposure group unless otherwise noted by an asterisk, in which case adults are the limiting group.

[b] Indicates those radionuclides for which K_d values are not available and for which K_d estimates have been made on the basis of chemical similarities.

calculated using the K_d values for each radionuclide given in Table VI, supplemented by assumptions based on chemical analogy, and a default suspended sediment fraction of 5×10^{-2} kg/m³ (see Ref. [I–1] for a more detailed explanation).

The following exposure pathways were considered: drinking water, ingestion of freshwater fish and external irradiation from radionuclides in shore/beach sediment. The dose from drinking water was calculated assuming that the water was filtered before consumption. The average annual activity concentration in fish was calculated

from the water concentration using the bioaccumulation factors given in Table XIII. The ingestion rates for drinking water and fish were taken from Table XIV and the dose coefficients for ingestion were taken from Table XVII.

The maximum annual surface activity concentration in beach sediment was calculated from the annual average water concentration and the relevant K_d value and using a factor of 60 kg/m^2 for the top 5 cm and the bulk density of sediments. A conversion factor of 0.1 was also applied to take account of the fact that the K_d value for beach sediment is one tenth of that for suspended sediment. Buildup during one year was taken into account, and the dose coefficients for external dose from deposits given in Table XV were applied to calculate maximum annual external doses, together with the occupancy factors for each age group presented in Table XIV.

The total dose from the three pathways was estimated for adults and children for each radionuclide. The dose to the age group with the higher dose is presented in Table I–V.

REFERENCE

[I–1] INTERNATIONAL ATOMIC ENERGY AGENCY, Evaluating the Reliability of Predictions Made Using Environmental Transfer Models, Safety Series No. 100, IAEA, Vienna (1989).

RADIONUCLIDE HALF-LIVES AND DECAY CONSTANTS

Half-lives $(T_{1/2})$ and decay constants (λ_i) for the radionuclides of concern are provided in Table II–I. Values for the half-lives were obtained from Ref. [II–1] and are provided in units of years (a), days (d) and hours (h). Decay constants were calculated from the half-lives and are provided in units of per day (d^{-1}) and per second (s^{-1}).

TABLE II–I. RADIONUCLIDE HALF-LIVES AND DECAY CONSTANTS

Nuclide	Half-life $(T_{1/2})$	λ_i (s^{-1})	λ_i (d^{-1})
Ac-228	6.13 h	3.14×10^{-5}	2.71
Ag-110m	250 d	3.21×10^{-8}	2.77×10^{-3}
Am-241	4.32×10^2 a	5.09×10^{-11}	4.40×10^{-6}
As-76	1.10 d	7.29×10^{-6}	6.30×10^{-1}
At-211	7.21 h	2.67×10^{-5}	2.31
Au-198	2.69 d	2.98×10^{-6}	2.58×10^{-1}
Bi-206	6.24 d	1.29×10^{-6}	1.11×10^{-1}
Bi-210	5.01 d	1.60×10^{-6}	1.38×10^{-1}
Bi-212	1.01 h	1.91×10^{-4}	1.65×10^1
Br-82	1.47 d	5.46×10^{-6}	4.72×10^{-1}
Cd-109	1.27 a	1.73×10^{-8}	1.50×10^{-3}
Ce-141	32.5 d	2.47×10^{-7}	2.13×10^{-2}
Ce-144	284 d	2.82×10^{-8}	2.44×10^{-3}
Cm-242	163 d	4.92×10^{-8}	4.25×10^{-3}
Cm-244	18.1 a	1.21×10^{-9}	1.05×10^{-4}
Co-58	70.8 d	1.13×10^{-7}	9.79×10^{-3}
Co-60	5.27 a	4.17×10^{-9}	3.60×10^{-4}
Cr-51	27.7 d	2.90×10^{-7}	2.50×10^{-2}
Cs-134	2.06 a	1.07×10^{-8}	9.22×10^{-4}
Cs-135	2.30×10^6 a	9.56×10^{-15}	8.26×10^{-10}
Cs-136	13.1 d	6.12×10^{-7}	5.29×10^{-2}
Cs-137	30.0 a	7.33×10^{-10}	6.33×10^{-5}
Cu-64	12.7 h	1.52×10^{-5}	1.31
Eu-154	8.80 a	2.50×10^{-9}	2.16×10^{-4}
Eu-155	4.96 a	4.43×10^{-9}	3.83×10^{-4}
Fe-55	2.70 a	8.14×10^{-9}	7.03×10^{-4}
Fe-59	44.5 d	1.80×10^{-7}	1.56×10^{-2}
Ga-67	3.26 d	2.46×10^{-6}	2.13×10^{-1}
Hg-197	2.67 d	3.00×10^{-6}	2.60×10^{-1}
Hg-197m	23.8 h	8.09×10^{-6}	6.99×10^{-1}

TABLE II–I. (cont.)

Nuclide	Half-life ($T_{1/2}$)	λ_i (s^{-1})	λ_i (d^{-1})
Hg-203	46.6 d	1.72×10^{-7}	1.49×10^{-2}
I-123	13.2 h	1.46×10^{-5}	1.26
I-125	60.1 d	1.33×10^{-7}	1.15×10^{-2}
I-129	1.57×10^7 a	1.40×10^{-15}	1.21×10^{-10}
I-131	8.04 d	9.98×10^{-7}	8.62×10^{-2}
I-132	2.30 h	8.37×10^{-5}	7.23
I-133	20.8 h	9.26×10^{-6}	8.00×10^{-1}
I-134	0.876 h	2.20×10^{-4}	1.90×10^1
I-135	6.61 h	2.91×10^{-5}	2.52
In-111	2.83 d	2.83×10^{-6}	2.45×10^{-1}
In-113m	1.66 h	1.16×10^{-4}	1.00×10^1
Mn-54	312 d	2.57×10^{-8}	2.22×10^{-3}
Mo-99	2.75 d	2.92×10^{-6}	2.52×10^{-1}
Na-22	2.60 a	8.45×10^{-9}	7.30×10^{-4}
Na-24	15.0 h	1.28×10^{-5}	1.11
Nb-95	35.1 d	2.29×10^{-7}	1.97×10^{-2}
Ni-59	7.50×10^4 a	2.93×10^{-13}	2.53×10^{-8}
Ni-63	96.0 a	2.29×10^{-10}	1.98×10^{-5}
Np-237	2.14×10^6 a	1.03×10^{-14}	8.87×10^{-10}
Np-239	2.36 d	3.40×10^{-6}	2.94×10^{-1}
P-32	14.3 d	5.61×10^{-7}	4.85×10^{-2}
Pa-231	3.27×10^4 a	6.72×10^{-13}	5.81×10^{-8}
Pa-233	27.0 d	2.97×10^{-7}	2.57×10^{-2}
Pb-210	22.3 a	9.86×10^{-10}	8.52×10^{-5}
Pd-103	17.0 d	4.72×10^{-7}	4.08×10^{-2}
Pd-107	6.50×10^6 a	3.38×10^{-15}	2.92×10^{-10}
Pd-109	13.4 h	1.44×10^{-5}	1.24
Pm-147	2.62 a	8.39×10^{-9}	7.25×10^{-4}
Po-210	138 d	5.81×10^{-8}	5.02×10^{-3}
Pu-238	87.7 a	2.51×10^{-10}	2.17×10^{-5}
Pu-239	2.41×10^4 a	9.12×10^{-13}	7.88×10^{-8}
Pu-240	6.54×10^3 a	3.36×10^{-12}	2.90×10^{-7}
Pu-241	14.4 a	1.53×10^{-9}	1.32×10^{-4}
Pu-242	3.76×10^5 a	5.85×10^{-14}	5.05×10^{-9}
Ra-224	3.66 d	2.19×10^{-6}	1.89×10^{-1}
Ra-225	14.8 d	5.42×10^{-7}	4.68×10^{-2}
Ra-226	1.60×10^3 a	1.37×10^{-11}	1.19×10^{-6}
Rb-86	18.7 d	4.29×10^{-7}	3.71×10^{-2}
Rh-105	1.47 d	5.46×10^{-6}	4.72×10^{-1}
Rh-107	0.362 h	5.32×10^{-4}	4.60×10^1
Ru-103	39.3 d	2.04×10^{-7}	1.76×10^{-2}

TABLE II–I. (cont.)

Nuclide	Half-life ($T_{1/2}$)	λ_i (s^{-1})	λ_i (d^{-1})
Ru-106	1.01 a	2.18×10^{-8}	1.88×10^{-3}
S-35	87.4 d	9.18×10^{-8}	7.93×10^{-3}
Sb-124	60.2 d	1.33×10^{-7}	1.15×10^{-2}
Sb-125	2.77 a	7.93×10^{-9}	6.86×10^{-4}
Se-75	120 d	6.69×10^{-8}	5.78×10^{-3}
Sn-113	115 d	6.98×10^{-8}	6.03×10^{-3}
Sr-85	64.8 d	1.24×10^{-7}	1.07×10^{-2}
Sr-87m	2.80 h	6.88×10^{-5}	5.94
Sr-89	50.5 d	1.59×10^{-7}	1.37×10^{-2}
Sr-90	29.1 a	7.55×10^{-10}	6.53×10^{-5}
Tc-99	2.13×10^5 a	1.03×10^{-13}	8.92×10^{-9}
Tc-99m	6.02 h	3.20×10^{-5}	2.76
Te-125m	58.0 d	1.38×10^{-7}	1.20×10^{-2}
Te-127m	109 d	7.36×10^{-8}	6.36×10^{-3}
Te-129m	33.6 d	2.39×10^{-7}	2.06×10^{-2}
Te-131m	1.25 d	6.42×10^{-6}	5.55×10^{-1}
Te-132	3.26 d	2.46×10^{-6}	2.13×10^{-1}
Th-228	1.91 a	1.15×10^{-8}	9.94×10^{-4}
Th-230	7.70×10^4 a	2.85×10^{-13}	2.47×10^{-8}
Th-232	1.40×10^{10} a	1.57×10^{-18}	1.36×10^{-13}
Tl-201	3.04 d	2.64×10^{-6}	2.28×10^{-1}
Tl-202	12.2 d	6.58×10^{-7}	5.68×10^{-2}
U-232	72.0 a	3.05×10^{-10}	2.64×10^{-5}
U-234	2.44×10^5 a	9.01×10^{-14}	7.78×10^{-9}
U-235	7.04×10^8 a	3.12×10^{-17}	2.70×10^{-12}
U-238	4.47×10^9 a	4.92×10^{-18}	4.25×10^{-13}
Y-87	3.35 d	2.39×10^{-6}	2.07×10^{-1}
Y-90	2.67 d	3.00×10^{-6}	2.60×10^{-1}
Y-91	58.5 d	1.37×10^{-7}	1.18×10^{-2}
Zn-65	244 d	3.29×10^{-8}	2.84×10^{-3}
Zr-95	64.0 d	1.25×10^{-7}	1.08×10^{-2}

REFERENCE

[II–1] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Radionuclide Transformations — Energy and Intensity of Emissions. Report of a Task Group of Committee 2 of the International Commission on Radiological Protection on Data Used in ICRP Publication 30, ICRP Publication 38, Vols 11–13, Pergamon Press, Oxford and New York (1983).

Annex III

SPECIAL CONSIDERATIONS FOR ASSESSMENT OF DISCHARGES OF TRITIUM AND CARBON-14

Since ^{3}H and ^{14}C can be incorporated into a great variety of different chemical compounds within the human body, the radiological dose from releases of these radionuclides is best assessed using models that employ a specific activity approach [III–1, III–2]. These models are based on the assumption that a steady state equilibrium has been attained between the environment and the exposed individuals, so that the ratio between the radionuclide and its stable counterpart is fixed. Such models are considered to give conservative dose estimates when an individual is assumed to be in complete equilibrium with maximum levels of environmental specific activity of ^{3}H and ^{14}C. However, more advanced models have been developed to assess exposure under non-equilibrium conditions and to include the special behaviour of ^{3}H and ^{14}C in organic compounds [III–3]. They indicate that taking account of incorporation into organic compounds can lead to estimated doses that are higher than those obtained using the specific activity model. For simplicity, and taking account of other conservatisms inherent in the dose calculation, the specific activity approach is adopted here. More detailed models may be needed if the total dose to the critical group exceeds the reference level discussed in Sections 2 and 8 and exposure to ^{3}H or ^{14}C is a significant contributor to that dose.

III–1. TRITIUM

The specific activity model for ^{3}H assumes that the nuclide is transferred through the environment and incorporated into the organism through its association with water molecules. It is also assumed that the concentration of ^{3}H in humans is derived from equilibrium concentrations of ^{3}H in water vapour present in the atmosphere receiving the airborne discharges, and from water of the aquatic environments receiving liquid discharges. Thus the dose rate is calculated as follows.

$$D_T^{\max} = [(C_A)_{x_1}^{\max} \times (f_A)_{x_1} + (C_W)_{x_2}^{\max} \times (f_W)_{x_2}]g \qquad \text{(III–1)}$$

where

$D_T^{\max}$ is the dose rate (Sv/a) for ^{3}H to the body of a representative member of the critical group;

$(C_A)^{\mathrm{max}}_{x_1}$ is the steady state concentration of ^{3}H in atmospheric water vapour (Bq/L) at location x_1 resulting from airborne releases (this is the atmospheric concentration relevant for calculating critical group doses);

$(f_A)_{x_1}$ is the fraction of the total water intake that is derived from atmospheric water vapour at location x_1 (this fraction includes water absorbed through the skin and through inhalation, as well as water vapour included in the formation of rain that is incorporated into foods and drinking water and other liquids produced at location x_1 and subsequently consumed by the individual);

$(C_W)^{\mathrm{max}}_{x_2}$ is the steady state concentration of ^{3}H in water at location x_2 resulting from releases to the aquatic environment (Bq/L) (this is the concentration in the aquatic environment relevant for calculating critical group doses);

$(f_W)_{x_2}$ is the fraction of the total water intake of the potentially exposed person that is derived from water at location x_2 that has been contaminated with aquatic discharges of tritiated water (this fraction includes the consumption of water in foods that have been irrigated with water from location x_2, as well as drinking water and other water based beverages derived from this location);

g is the dose rate conversion factor (Sv/a per Bq/L of human body water content).

The concentration of ^{3}H atmospheric water vapour at location x_1 is calculated as

$$(C_A)^{\mathrm{max}}_{x_1} = \frac{(X)^{\mathrm{max}}_{x_1}}{(H)_{x_1}} \qquad\qquad\qquad (\mathrm{III}\text{--}2)$$

where

$(X)^{\mathrm{max}}_{x_1}$ is the concentration of ^{3}H in air at location x_1 (Bq/m^3) resulting from a release to the atmosphere (computations of $(X)^{\mathrm{max}}_{x_1}$ are performed using the equations described in Section 3 of this report),

$(H)_{x_1}$ is the absolute humidity of the atmosphere, assumed as a default value to be 6×10^{-3} L/m^3 of air or 6×10^{-3} kg/m^3.

The concentration of ^{3}H in water at location x_2, resulting from discharges of ^{3}H to the aquatic environment $(C_W)_{x_1}$, may be estimated using the models described in Section 4.

For the parameters $(f_A)_{x_1}$ and $(f_A)_{x_2}$ default values of unity are proposed. This value of unity results in a sufficiently cautious estimate of dose rates, given accurate estimates of the maximum specific activity of ^{3}H at locations x_1 and x_2.

The error introduced by assuming that the concentration of ^{3}H in the body, including its organic molecules, equals that of water can be neglected in these calculations. Doses to the body can be obtained by multiplying the activity levels by the appropriate dose rate factor g. This relates the dose rate for the total body of an exposed individual to the concentration of ^{3}H per litre of water in the body. For ^{3}H the dose rate factor is 2.6×10^{-8} Sv/a per Bq/L [III–1].

III–2. CARBON-14

The specific activity model for ^{14}C is based on the following assumptions [III–2].

(a)   Ingestion of carbon originating from the atmosphere is the primary mode of exposure, and all other pathways of exposure will contribute less than 1% of the total dose;
(b)   The ^{14}C released is associated with CO_2 molecules and is subsequently fixed within plant tissues during photosynthesis;
(c)   The organic molecules thus formed are transported along with stable ^{12}C through food chains and into the human body.

Consequently, the dose rate at equilibrium will be directly proportional to the concentration of ^{14}C in air relative to the concentration of stable carbon at a given location x, multiplied by the fraction of total dietary carbon derived from this location. The most sensitive aspects of this model are expected to be the assumption of equilibrium, the determination of the specific activity to which food products are ultimately exposed $(A)_x^{\mathrm{max}}$, and the determination of the total dietary carbon contaminated with ^{14}C.

$$D^{\mathrm{max}} = (A)_x^{\mathrm{max}} \times (f_{c,a})_x \times g \qquad\qquad \text{(III–3)}$$

where

D^{max} is the effective dose rate (Sv/a).
$(A)_x^{\mathrm{max}}$ is the specific activity (Bq ^{14}C per gram of C) to which food products at location x will be chronically exposed — this is the specific activity assumed to contribute most significantly to the individuals of interest.
$(f_{c,a})_x$ is the fraction of total dietary carbon derived from location x by the representative member of the critical group — for the purposes of the generic calculational approach recommended in this report, $(f_{c,a})_x$ is assumed cautiously to be unity.

g is the effective dose rate factor that relates the annual dose rate (Sv/a) to the concentration of ^{14}C per gram of carbon in people (Bq/g). The dose rate factor recommended for screening is 5.6×10^{-5} Sv/a per Bq/g [III–2].

Values of $(A)_x^{max}$ can be calculated as follows.

$$(A)_x^{max} = \frac{(X)_x^{max}}{(C)_x} \qquad\qquad\qquad (III–4)$$

where

$(X)_x^{max}$ is the concentration of ^{14}C in air at location x (Bq/m^3) calculated using the equations described in Section 3 of this report;

$(C)_x$ is the concentration of airborne carbon, assumed to be 1.8×10^{-1} g/m^3, corresponding to an average atmospheric CO_2 concentration of 330 ppm (vol.).

REFERENCES

[III–1] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Age-dependent Doses to Members of the Public from Intake of Radionuclides, Part 5, Compilation of Ingestion and Inhalation Dose Coefficients, ICRP Publication No. 72, Pergamon Press, Oxford and New York (1996).

[III–2] KILLOUGH, G.G., ROHWER, P.S., A new look at the dosimetry of ^{14}C released to the atmosphere as carbon dioxide, Health Phys. **34** (1978) 141–159.

[III–3] GALERIU, D., Transfer Parameters for Routine Releases of HTO — Consideration of OBT, Rep. AECL-11052, COG-94-76, Atomic Energy of Canada, Chalk River, ON (1994).

Annex IV

EXAMPLE CALCULATIONS

IV–1. EXAMPLE CALCULATION FOR DISCHARGES TO THE ATMOSPHERE WHEN $H > 2.5H_B$

IV–1.1. Scenario description

Iodine-131 is released at an average rate $Q_i = 1$ Bq/s from a stack with height $H = 60$ m. The nearest building with a potential for influencing the flow of this plume has a height $H_B = 20$ m. A farm is located 1 km downwind of this release point. Assuming that an individual lives an entire year at this farm, what is the annual average air concentration and the ground deposition rate at this location?

IV–1.2. Calculational procedure

The first step is to determine which dispersion model is most appropriate. In this case $2.5H_B = (2.5) (20)$ m $= 50$ m. Thus $H > 2.5H_B$, and since $x < 20$ m, Eq. (2) should be used in this calculation

$$C_A = (P_p)(F)(Q_i)/u_a$$

In the absence of site specific information, use the default values $P_p = 0.25$ and $u_a = 2$ m/s. From Table I for $H = 46–80$ m and $x = 1000$ m, $F = 1 \times 10^{-5}$ m^{-2}. Substituting into Eq. (2) gives

$$
\begin{aligned}
C_A &= (0.25)(1 \times 10^{-5} \text{ m}^{-2})(1 \text{ Bq/s})/(2 \text{ m/s})^{-1} \\
&= 1.3 \times 10^{-6} \text{ Bq/m}^3
\end{aligned}
$$

The deposition rate for this location is given by the equation

$$\dot{d}_i = (V_T)(C_A)$$

(Section 3.9). Use of the default value of $V_T = 1000$ m/d for ^{131}I gives

$$
\begin{aligned}
\dot{d}_i &= (1000 \text{ m/d})(1.3 \times 10^{-6} \text{ Bq/m}^3) \\
&= 1.3 \times 10^{-3} \text{ Bq·m}^{-2}\text{·d}^{-1}
\end{aligned}
$$

IV–2. EXAMPLE CALCULATION FOR DISCHARGES TO THE ATMOSPHERE FOR RECEPTORS IN THE WAKE AND CAVITY ZONES

IV–2.1. Scenario description

Assume the same information as in Example IV–1, except that the radioactive material is being released to the atmosphere from a small vent pipe in the side of a building. The diameter of the vent pipe is 0.5 m and the cross-sectional area of the building A_B is 500 m². On the same side of the building, 5 m from the release point, is located the air intake for a private residence. An individual resides in that location at all times. Further, all of this person's food is provided by a farm located 1 km downwind of this release point. What are the annual average air concentration and ground deposition rates at these two locations?

IV–2.2. Calculational procedure

IV–2.2.1. Residence

Since the source and receptor are located on the same building surface, the procedures outlined in Section 3.6.1 should be used. The concentration in air may be calculated using Eq. (7) since x (5 m) is greater than three times the diameter of the vent stack (3 × 0.5 m = 1.5 m).

$$C_A = (B_0)(Q_i)/(u_a)(x^2)$$

Using the parameter values appropriate for this problem

$$C_A = (30)(1 \text{ Bq/s})/(2 \text{ m/s})(5 \text{ m})^2$$
$$= 0.6 \text{ Bq/m}^3$$

Again, using Section 3.9 to calculate the ground deposition rate

$$\dot{d}_i = (1000 \text{ m/d})(0.6 \text{ Bq/m}^3)$$
$$= 600 \text{ Bq·m}^{-2} \cdot \text{d}^{-1}$$

IV–2.2.2. Farm

Since 2.5 times the square root of the cross-sectional area of this building is 56 m, which is less than the downwind distance to the farm of 1 km, Eq. (4) in Section 3.5 is appropriate.

$$C_A = (P_p)(B)(Q_i)/u_a$$

From Table II for A_B = 401–800 m^2 and x = 1000 m, B = 4 × 10^{-5} m^{-2}; thus

$$C_A = (0.25)(4 \times 10^{-5} \text{ m}^{-2})(1 \text{ Bq/s})/(2 \text{ m/s})$$
$$= 5 \times 10^{-6} \text{ Bq/m}$$

It follows that the ground deposition rate for this location is

$$\dot{d}_i = (1000 \text{ m/d})(5 \times 10^{-6} \text{ Bq/m}^3)$$
$$= 5 \times 10^{-3} \text{ Bq·m}^{-2}\text{·d}^{-1}$$

IV–3. EXAMPLE CALCULATION FOR DISCHARGES TO THE ATMOSPHERE WHEN $H < 2.5H_B$ AND $X \leq 2.5 \sqrt{A_B}$ AND THE SOURCE AND RECEPTOR ARE NOT ON THE SAME BUILDING SURFACE

IV–3.1. Scenario description

Assume the same information as in the previous examples, except that the radioactive material is released from a short stack located on the top of a building. The height of the stack H is 33 m. The building where the stack is located has a height H_B of 30 m and a frontal cross-sectional area of 5000 m^2. A continuously occupied residence is located 150 m downwind of this source. The food for this residence is supplied by a farm located 1 km downwind of the source. What are the annual average air concentration and ground deposition rates at these two locations?

IV–3.2. Calculational procedure

IV–3.2.1. Residence

The stack height H (33 m) is less than 2.5 times the height of the building H_B (2.5 × 30 = 75 m), the distance to the residence (150 m) is less than 2.5 times the square root of the cross-sectional area of the building (177 m), and the source and the receptor are not on the same building surface. As a result, Eq. (8) in Section 3.6.2 is appropriate.

$$C_A = (P_p)(Q_i)/(3.14)(u_a)(H_B)(K)$$
$$= (0.25)(1 \text{ Bq/s})/(3.14)(2 \text{ m/s})(30 \text{ m})(1 \text{ m})$$
$$= 1.3 \times 10^{-3} \text{ Bq/m}^3$$

147

The ground deposition rate at this location is again given by

$$\dot{d}_i = (1000 \text{ m/d})(1.3 \times 10^{-3} \text{ Bq/m}^3)$$
$$= 1.3 \text{ Bq·m}^{-2}\text{·d}^{-1}$$

IV–3.2.2. Farm

Since the downwind distance to the farm (1000 m) is greater than 2.5 times the square root of the frontal cross-sectional area, this problem is identical to the farm portion of Example IV–2; that is

$$C_A = 5 \times 10^{-6} \text{ Bq/m}^3$$
$$\dot{d}_i = 5 \times 10^{-3} \text{ Bq·m}^{-2}\text{·d}^{-1}$$

IV–4. EXAMPLE CALCULATION FOR DISCHARGES INTO A RIVER

IV–4.1. Scenario description

Caesium-137 is to be released to a relatively small river at a release rate of 3.7×10^{10} Bq/a with a radionuclide effluent discharge rate of 1 m^3/s. A 30 year low annual riverflow rate is not known, but a representative river width under normal (mean annual) river discharge conditions is estimated to be 50 m. The nearest potentially exposed individual lives 1 km downstream from the outfall on the same side of the river. Determine the ^{137}Cs concentration 1 km downstream.

IV–4.2. Calculational procedure

$Q_i = 3.7 \times 10^{10}$ Bq/a = 1170 Bq/s

$F = 1$ m^3/s

$\lambda_i = 7.33 \times 10^{-10}$ s^{-1} (from Annex II)

$\bar{B} = 50$ m, $x = 1000$ m

The river conditions under a 30 year low annual mean flow must first be estimated. From Table III $\bar{B} = 50$ m corresponds to a mean annual river flow rate of around $\bar{q}_r = 30$ m^3/s. Thus the 30 year low annual riverflow rate $q_r = \bar{q}_r/3 = 10$ m^3/s.
From Table III $q_r = 10$ m^3/s gives $B = 28.8$ m and $D = 0.48$ m, and since $U = q_r/BD$, $U = 0.72$ m/s. $L_z = 7 \times D = 3.4$ m, $x = 1000 > L_z$, so correction for partial mixing is required.

$$A = 1.5Dx/B^2 = 0.87$$

From Table IV the river partial mixing correction factor (P_r) corresponding to $A = 0.87$ is approximately 2.7. From Eq. (12) the fully mixed radionuclide concentration (e.g. on the opposite bank), C_t, is

$$C_t = \left(\frac{1170}{10}\right) \times \exp\left[\frac{-(7.33 \times 10^{-10})1000}{0.72}\right] = 117 \ \text{Bq/m}^3$$

For the same bank Eq. (14) applies, and the radionuclide concentration at 1 km downstream is

$$C_{w, \ tot} = C_t \times P_r = 117 \times 2.7 = 316 \ \text{Bq/m}^3$$

IV–5.   EXAMPLE CALCULATION FOR DISCHARGES INTO AN ESTUARY

IV–5.1. Scenario description

Strontium-90 is to be discharged into a small estuary at a rate of 3.7×10^{10} Bq/a with a radionuclide effluent discharge of 1 m^3/s. The tide for this estuary is semi-diurnal (i.e. occurring twice per day). The 30 year low annual freshwater river discharge rate is 20 m^3/s. The average estuary width and depth are 100 m and 2 m, respectively. Both the maximum ebb and flow tide are estimated to be 1.0 m/s. The nearest potentially exposed individual lives 1 km upstream from the radionuclide discharge point on the same side of the estuary as the discharge. Determine the ^{90}Sr concentration in the estuary 1 km upstream.

IV–5.2. Calculational procedure

$Q_i = 3.7 \times 10^{10}$ Bq/a = 1170 Bq/s

$F\ = 1$ m^3/s, $x = 1000$ m

$\lambda_i = 7.55 \times 10^{-10}$ s^{-1} (from Annex II)

$q_r = 20$ m^3/s, $B = 100$ m, $D = 2$ m, $U_e = U_f = 1$ m/s

$T_p = 45\ 000$ s (or 12.5 h)

The net freshwater velocity $U = q_r/(BD) = (20)/(100)(2) = 0.1$ m/s. The mean tidal flow speed $U_t = 0.32 (|U_e| + |U_f|) = 0.32 (1.0 + 1.0) = 0.64$ m/s. Thus $q_w = DBU_t = (2)(100)(0.64) = 128$ m^3/s.

Since the receptor is located upstream, we must determine how far the radionuclide can travel upstream.

$$L_u = 0.32 |U_f| T_p = (0.32)(1.0)(4.5 \times 10^4) = 14\ 400 \text{ m.}$$

Note that $x > L_z = 7D = (7)(2) = 14$ m. Moreover, $x = 1000$ m $< L_u = 14\ 400$ m. Thus the radionuclide can reach this receptor location. The fully mixed radionuclide concentration can be obtained from Eq. (16) as

$$C_t = \left(\frac{1170}{128}\right) \times \exp\left[\frac{-(7.55 \times 10^{-10})1000}{0.1}\right] = 9.14 \text{ Bq/m}^3$$

In order to calculate the estuary partial mixing correction factor P_e, first calculate M from Eq. (17).

$$M = \frac{0.3DU_tT_p}{B^2} = \frac{(0.3)(2)(0.64)(4.5 \times 10^4)}{(100)^2} = 1.73$$

From Table V $N = 0.855$ with $M = 1.73$, therefore, from Eq. (18)

$$A = \frac{1.5Dx}{NB^2} \times \frac{U}{U_t} = \frac{(1.5)(2)(1000)(0.1)}{[(0.855)(100)^2(0.64)]} = 0.0548$$

From Fig. 14 with $A = 0.0548$ and $N = 0.855$, $P_e = 3.5$. Thus the ^{90}Sr concentration 1 km upstream, from Eq. (19), is

$$C_{w,\ tot} = (9.14)(3.5) = 32.0 \text{ Bq/m}^3$$

IV–6. EXAMPLE CALCULATION FOR DISCHARGES INTO COASTAL WATERS

IV–6.1. Scenario description

Ruthenium-106 is to be released to coastal water at the rate of 3.7×10^{10} Bq/a with a radionuclide effluent discharge rate of 1 m^3/s. The effluent outfall is located 50 m offshore, and the water depth at the release point is 30 m. Determine the radionuclide concentration at a location 2000 m downcurrent.

IV–6.2. Calculational procedure

$Q_i = 3.7 \times 10^{10}$ Bq/a = 1170 Bq/s, $F = 1$ m³/s

$D = 30$ m , $y_0 = 50$ m, $U = 0.1$ m/s (default value), $x = 2000$ m

$\lambda_i = 2.18 \times 10^{-8}$ s^{-1} (see Annex II)

The radionuclide concentration C is 'For fishing' (i.e. for use as the basis for radionuclide activity concentrations in fish), using Eq. (20)

$$C_{w,\,tot} \Big|_{y=y_0} = \frac{962 \times 0.1^{0.17} \times 1170}{30 \times 2000^{1.17}}$$

$$\times \exp\left[-\frac{(2.18 \times 10^{-8} \times 2000)}{0.1} \right] = 3.48 \, \text{Bq/m}^3$$

For this situation $C_{w,\,tot} = 3.48$ Bq/m³ 'For the shoreline' (i.e. for use as the basis for activity concentrations in shoreline sediments for calculation of external doses), using Eq. (21)

$$C_{w,\,tot} = \frac{(962)(0.1)^{(0.17)}(1170)}{(30)(2000)^{(1.17)}} \times \exp\left[\frac{-(7.28 \times 10^5)(0.1)^{2.34}(50)^2}{(2000)^{2.34}} \right]$$

$$\times \exp\left[\frac{-(2.18 \times 10^{-8})(2000)}{0.1} \right]$$

For this situation $C_{w,\,tot} = 2.98$ Bq/m³.

IV–7. EXAMPLE CALCULATION FOR DISCHARGES INTO A SMALL LAKE

IV–7.1. Scenario description

Cobalt-60 is to be discharged directly into a lake having a river inflow and outflow. The radionuclide release rate is 3.7×10^7 Bq/a with a radionuclide effluent discharge of 0.1 m³/s. The lake is 400 m long, 100 m wide and 10 m deep. The 30 year low annual riverflow rate is not known, but the representative river width under a normal (mean annual) river discharge is estimated to be 10 m. The nearest potentially exposed individual is living along the lake. What is the radionuclide concentration in the lake water?

IV–7.2. Calculational procedure

$Q_i = 3.7 \times 10^7$ Bq/a = 1.17 Bq/s, $\dot{d} = 0$

$F = 0.1$ m^3/s, $\bar{B} = 10$ m, $B = 100$ m, $D = 10$ m, $A_l = (400)(100) = 40\ 000$ m^2

$\lambda_i = 4.17 \times 10^{-9}$ s^{-1} (Annex II)

Firstly, the 30 year low annual flow rate of the river must be estimated. From Table III the mean annual river flow rate $\bar{q}_r = 1$ m^3/s, for $\bar{B} = 10$ m. Thus $q_r = 1/3 = 0.33$ m^3/s. The lake surface area $A_l = 40\ 000$ m$^2 = 0.04$ km^2, that is < 400 km^2; thus the lake is classified as a small lake. The lake volume is $V = (400)(100)(10) = 400\ 000$ m^3. Note that $(q_r/V + \lambda_i) = (0.33/400\ 000) + 4.17 \times 10^{-9} = 8.29 \times 10^{-7} > 10^{-8}$.

Since there is no radionuclide contribution to the lake from its watershed, $\dot{d} = 0$. Thus $Q'_i = Q_i = 1.17$ Bq/s. With Eq. (24) the totally mixed radionuclide concentration in the lake is

$$C_{w,\ tot} = \frac{(1.17)}{[0.33 + (4.17 \times 10^{-9})(400\ 000)]} = 3.53 \text{ Bq/cm}^3$$

IV–8. EXAMPLE CALCULATION OF RADIONUCLIDE CONCENTRATIONS IN SEDIMENT

IV–8.1. Scenario description

A nuclear facility has been discharging ^{137}Cs and ^{131}I into a river for 30 years. The concentrations of ^{137}Cs and ^{131}I in unfiltered river water are each determined to be 100 Bq/m^3 at a location along the river bank. The suspended sediment concentration there is 0.1 kg/m^3 (or 100 mg/L). What are the ^{137}Cs and ^{131}I concentrations in the river bottom sediment and in the river shore sediment?

IV–8.2. Calculational procedure

$S_s = 0.1$ kg/m^3, $T_e = 3.15 \times 10^7$ s

For ^{137}Cs, $K_d = 1000$ L/kg and $\lambda_i = 7.33 \times 10^{-10}$ s^{-1} (see Table VI and Annex II). For ^{131}I, $K_d = 10$ L/kg and $\lambda_i = 9.98 \times 10^{-7}$ s^{-1} (see Table VI and Annex II).

IV–8.2.1. ^{137}Cs

From Eq. (27) the ^{137}Cs concentration in the river bottom sediment is

$$C_{s,b} = \frac{(0.1)(0.001)(1000)(100)}{1+(0.001)(0.1)(1000)} \times \frac{1-e^{-(7.33 \times 10^{-10})(3.15 \times 10^{7})}}{(7.33 \times 10^{-10})(3.15 \times 10^{7})}$$

$$= (9.09)(0.989) = 8.99 \text{ Bq/kg}$$

From Eq. (28) the ^{137}Cs concentration in the river shore sediment is

$$C_{s,s} = \frac{(0.1)(0.001)(1000)(60)(100)}{1+(0.001)(0.1)(1000)} \times \frac{1-e^{-(7.33 \times 10^{-10})(3.15 \times 10^{7})}}{(7.33 \times 10^{-10})(3.15 \times 10^{7})}$$

$$= (545)(0.989) = 539 \text{ Bq/kg}$$

IV–8.2.2. ^{131}I

From Eq. (27) the ^{131}I concentration in the river bottom sediment is

$$C_{s,b} = \frac{(0.1)(0.001)(10)(100)}{1+(0.001)(0.1)(10)} \times \frac{1-e^{-(9.98 \times 10^{-7})(3.15 \times 10^{7})}}{(9.98 \times 10^{-7})(3.15 \times 10^{7})}$$

$$= 0.1 \times (0.0318) = 3.18 \times 10^{-3} \text{ Bq/kg}$$

From Eq. (28) the ^{131}I concentration in the river shore sediment is

$$C_{s,s} = \frac{(0.1)(0.001)(10)(60)(100)}{1+(0.001)(0.1)(10)} \times \frac{1-e^{-(9.98 \times 10^{-7})(3.15 \times 10^{7})}}{(9.98 \times 10^{-7})(3.15 \times 10^{7})}$$

$$= (5.99)(0.0318) = 0.19 \text{ Bq/kg}$$

IV–9. EXAMPLE CALCULATION OF FOOD CONCENTRATIONS FROM ATMOSPHERIC DEPOSITION

IV–9.1. Scenario description

Iodine-131 is released to the atmosphere, resulting in a continuous deposition rate of 1 Bq·m^{-2}·d^{-1} over agricultural land. What are the annual average concentrations of ^{131}I in crops, forage, milk and meat?

IV–9.2. Calculational procedure

IV–9.2.1. Concentrations in food crops from direct deposition

The concentration of ^{131}I on vegetation surfaces owing to direct deposition from the atmosphere is given by Eq. (30).

$$C_{v,1} = \frac{\dot{d}\alpha[1-\exp(-\lambda_{E^v}t_e)]}{\lambda_{E^v}}$$

where $\dot{d} = 1.0$ Bq·m^{-2}·d^{-1}, $\alpha = 0.3$ m^2/kg (for fresh weight of food crops, from Table VII), $\lambda_{E^v} = \lambda_i + \lambda_w = 8.62 \times 10^{-2}$ [d^{-1}] $+ 5 \times 10^{-2}$ [d^{-1}] $\approx 1.4 \times 10^{-1}$ [d^{-1}], $\lambda_i = 8.62 \times 10^{-2}$ d^{-1} (Annex II), $\lambda_w = 5 \times 10^{-2}$ d^{-1} (Table VII) and $t_e = 60$ d (from Table VIII).

$$C_{v,1\,(\text{food crops})} =$$

$$\frac{1\,[\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}]\times 0.3\,[\text{m}^2/\text{kg}][1-\exp(-1.4\times 10^{-1}[\text{d}^{-1}]\times 60\,[\text{d}])]}{1.4\times 10^{-1}[\text{d}^{-1}]}$$

$$= \frac{0.3\,[\text{Bq}\cdot\text{kg}^{-1}\cdot\text{d}^{-1}]}{1.4\times 10^{-1}[\text{d}^{-1}]}$$

$$= 2.1 \text{ Bq/kg fresh weight in food crops from direct deposition}$$

IV–9.2.2. Concentrations in food crops from uptake from soil

The concentration of ^{131}I in soil based on a unit deposition rate from the atmosphere is given by Eq. (32).

$$C_s = \frac{\dot{d}\,[1-\exp(-\lambda_{E^s}t_b)]}{\rho\lambda_{E^s}}$$

where $\dot{d} = 1.0$ Bq·m^{-2}·d^{-1}, $\lambda_{E^s} = \lambda_i + \lambda_s = 8.62 \times 10^{-2}$ [d^{-1}] $+ 1.4 \times 10^{-3}$ [d^{-1}] $= 8.8 \times 10^{-2}$ [d^{-1}], $\lambda_i = 8.62 \times 10^{-2}$ d^{-1} (Annex II), $\lambda_s = 1.4 \times 10^{-3}$ (Table X), $t_b = 1.1 \times 10^4$ d (Table VIII), and $\rho = 260$ kg/m^2 for cultivated soil other than peat (Table IX). Thus

$$C_s = \frac{1\,[\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}][1-\exp(-8.8\times 10^{-2}[\text{d}^{-1}]\times 1.1\times 10^4[\text{d}])]}{260\,[\text{kg/m}^2]\times 8.8\times 10^{-2}[\text{d}^{-1}]}$$

$$= \frac{1\,[\text{Bq}\cdot\text{m}^{-2}\cdot\text{d}^{-1}]}{260\times 8.8\times 10^{-2}[\text{kg}\cdot\text{m}^{-2}\cdot\text{d}^{-1}]} = 4.4\times 10^{-2}\,\text{Bq/kg dry weight of soil}$$

The concentration of ^{131}I in vegetation resulting from uptake from soil is given by Eq. (31).

$$C_{v,\,2} = F_v \times C_s$$

where $F_v = 0.02$ for food crops fresh weight (Table XI). Therefore the concentration of ^{131}I in food crops owing to uptake of ^{131}I from soil is $C_{v,\,2\text{ (food crops)}} = 0.02$ [unitless] $\times 4.4 \times 10^{-2}$ [Bq/kg] $= 8.8 \times 10^{-4}$ Bq/kg fresh weight.

IV–9.2.3. Total concentration in food crops

The concentration of ^{131}I in food crops from both direct deposition and uptake from soil is given by Eq. (33).

$$C_{v\text{ (food crops)}} = (C_{v,\,1\text{ (food crops)}} + C_{v,\,2\text{ (food crops)}})\,\exp(-\lambda_i\,t_h)$$

where $t_h = 14$ d (from Table VIII). $C_{v\text{ (food crops)}} = (2.1$ [Bq/kg] $+ 8.8 \times 10^{-4}$ [Bq/kg]) $\times \exp(-8.62 \times 10^{-2}$ [d^{-1}] $\times 14$ [d]) $= 6.3 \times 10^{-1}$ Bq/kg fresh weight.

IV–9.2.4. Pasture concentrations

For pasture forage the concentration is calculated on a dry weight basis, and the soil is assumed to be uncultivated (not ploughed). The necessary parameters are (a) α and t_e from Eq. (30), (b) F_v from Eq. (31), (c) ρ from Eq. (32) and (d) t_h from Eq. (33). The parameter values specific to forage are $\alpha = 3$ m^2/kg (Table VII), $t_e = 30$ d (Table VIII), $F_v = 0.1$ (Table XI), $\rho = 130$ kg/m^2 (Table IX) and $t_h = 0$ (Table VIII). Therefore the concentration on forage owing to direct deposition of ^{131}I is (Eq. (30))

$$C_{v,\,1(\text{pasture})} = \frac{1\,[\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}] \times 3\,[\text{m}^2/\text{kg}] \times [1 - \exp(-1.4 \times 10^{-1}[\text{d}^{-1}] \times 30\,[\text{d}])]}{1.4 \times 10^{-1}\,[\text{d}]}$$

$$= 21 \text{ Bq/kg dry weight forage}$$

The concentration in pasture soil is (Eq. (32))

$$C_s = \frac{1\,[\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}][1 - \exp(-8.8 \times 10^{-2}\,[\text{d}^{-1}] \times 1.1 \times 10^4\,[\text{d}])]}{130\,[\text{kg/m}^2] \times 8.8 \times 10^{-2}\,[\text{d}^{-1}]}$$

$$= 8.7 \times 10^{-2} \text{ Bq/kg dry weight of soil}$$

The uptake from soil by pasture vegetation is (Eq. (31))

$C_{v, 2 \text{ (pasture)}} = 0.1 \text{ [unitless]} \times 8.7 \times 10^{-2} \text{ [Bq/kg]} = 8.7 \times 10^{-3} \text{ Bq/kg}$

The total contamination in pasture from direct deposition and uptake from soil is (Eq. (33)) $C_{v \text{ (pasture)}} = (2.1 \times 10^1 \text{ [Bq/kg]} + 8.7 \times 10^{-3} \text{ [Bq/kg]}) \times \exp(-8.62 \times 10^{-2} \text{ [d}^{-1}] \times 0 \text{ [d]}) = 21 \text{ Bq/kg dry weight forage.}$

IV–9.2.5. Concentrations in stored feed and average concentrations for feeds

If pasture is harvested and stored for $t_h = 90$ d (Table VIII), the concentration on stored feed is (Eq. (33))

$$C_{p \text{ (pasture)}} = (2.1 \times 10^1 \text{ [Bq/kg]} + 8.7 \times 10^{-3} \text{ [Bq/kg]}) \times \exp(-8.62 \times 10^{-2} \text{ [d}^{-1}]$$
$$\times 90 \text{ [d])}$$
$$= 9.0 \times 10^{-3} \text{ Bq/kg dry weight forage}$$

Assuming that both dairy and beef cattle are fed a diet of 70% fresh forage (Table XII), the average annual concentration of ^{131}I in animal feed is (Eq. (35))

$$C_a = f_p\, C_{v \text{ (pasture)}} + (1 - f_p)\, C_{p \text{ (pasture)}} = 0.7 \times 21 \text{ [Bq/kg]} + (1 - 0.7)\, 9.0$$

$$\times 10^{-3} \text{ [Bq/kg]} = 15 \text{ Bq/kg dry weight}$$

IV–9.2.6. Concentration in milk

Assuming that the concentration of ^{131}I in water is negligible, the concentration in milk is (Eq. (36))

$$C_M = F_m\, C_a\, Q_M \exp(-\lambda_i\, t_m)$$

where $F_m = 0.01$ d/L (Table XI), $C_a = 15$ Bq/kg, $Q_M = 16$ kg/d (Table XII) and $t_m = 1$ d (Table VIII).

$$C_M = 0.01 \text{ [d/L]} \times 15 \text{ [Bq/kg]} \times 16 \text{ [kg/d]}$$
$$\times \exp(-8.62 \times 10^{-2} \text{ [d}^{-1}] \times 1 \text{ [d]}) = 2.2 \text{ Bq/L}$$

IV–9.2.7. Concentration in meat

Likewise, the concentration of ^{131}I in meat is (Eq. (37))

$$C_F = F_f\, C_a\, Q_f \exp(-\lambda_i\, t_f)$$

where F_f = 0.05 d/kg (Table XI), C_a = 15 Bq/kg, Q_f = 12 kg/d (Table XII) and t_f = 20 d (Table VIII).

$$C_f = 0.05 \text{ [d/kg]} \times 15 \text{ [Bq/kg]} \times 12 \text{ [kg/d]} \times \exp(-8.62 \times 10^{-2} \text{ [d}^{-1}\text{]} \times 20 \text{ [d]})$$
$$= 1.6 \text{ Bq/kg in meat}$$

IV–9.2.8. Summary

$$C_{\text{food crops}} = 0.63 \text{ Bq/kg}, \, C_{\text{pasture}} = 21 \text{ Bq/kg}$$

$$C_{\text{milk}} = 2.2 \text{ Bq/L}, \, C_{\text{meat}} = 1.6 \text{ Bq/kg}$$

IV–10. EXAMPLE CALCULATION OF FOOD CONCENTRATIONS FROM CONCENTRATIONS IN WATER

IV–10.1. Scenario description

The water of a small lake located near a village contains water with a concentration (C_w) of ^{99}Tc of 1 Bq/m^3. The lake is used by the inhabitants of the village for fishing and for irrigating crops and pasture fields. These fields are irrigated on one day per week for a four month period each year. The quantity of water used in one day is about 9 L/m^2. The soil in the agricultural area is a peat soil. Animals also drink the lake water. Calculate the concentration of ^{99}Tc in the major food items produced in the area (fish, food crops, milk and meat). Which mechanism contributes most to plant contamination?

IV–10.2. Calculational procedure

The concentration of ^{99}Tc in fish is given by Eq. (38).

$$C_F = C_w \, B_p / 1000$$

where C_w = 1 Bq/m^3, B_p = 50 L/kg (Table XIII). Thus C_F = 1 [Bq/m^3] $\times$ 50 [L/kg]/1000 m^3/L = 0.050 Bq/kg.

The radionuclide concentration in vegetation is assumed to be due to irrigation only. The deposition rate due to irrigation is given by Eq. (34).

$$\dot{d} = C_w \times I_w$$

where $C_w = 1$ Bq/m^3 and I_w is the average irrigation rate (m$^3 \cdot$m$^{-2} \cdot$d^{-1}).

For deposition on plant surfaces, the average irrigation rate over the irrigation period is required; for deposition on soil, the annual average irrigation rate is required. The amount used in one day is 9 L/m^2 for 16 days over a 120 day period (once a week for four months). Thus the average irrigation rate over the irrigation period is

$$I_w = 9 \, [\text{L} \cdot \text{m}^{-2} \cdot \text{d}^{-1}] \times \frac{16\,\text{d}}{120\,\text{d}} = 1.3 \, [\text{L} \cdot \text{m}^{-2} \cdot \text{d}^{-1}] = 1.3 \times 10^{-3} \, [\text{m}^3 \cdot \text{m}^{-2} \cdot \text{d}^{-1}]$$

The annual average irrigation rate

$$I_{w\,(\text{ann})} = 1.3 \times 10^{-3} \, [\text{m}^3 \cdot \text{m}^{-2} \cdot \text{d}^{-1}] \times \frac{120\,\text{d}}{365\,\text{d}} = 4.3 \times 10^{-4} \, [\text{m}^3 \cdot \text{m}^{-2} \cdot \text{d}^{-1}]$$

Therefore the deposition rate on vegetation during the irrigation period is

$$\dot{d} = 1 \, [\text{Bq/m}^3] \times (1.3 \times 10^{-3}) [\text{m}^3 \cdot \text{m}^{-2} \cdot \text{d}^{-1}] = 1.3 \times 10^{-3} \, \text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$$

and the annual average deposition rate on soil is

$$\dot{d} = 1 \, [\text{Bq/m}^3] \times (4.3 \times 10^{-4}) [\text{m}^3 \cdot \text{m}^{-2} \cdot \text{d}^{-1}] = 4.3 \times 10^{-4} \, \text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}$$

The concentration of ^{99}Tc on vegetation surfaces owing to direct deposition from irrigation is given by Eq. (30).

$$C_{v,1} = \frac{\dot{d}\alpha[1 - \exp(-\lambda_{E^v} t_e)]}{\lambda_{E^v}}$$

where $\dot{d} = 1.3 \times 10^{-3}$ Bq$\cdot$m$^{-2}\cdot$d^{-1}, $\alpha = 0.3$ m^2/kg fresh weight for food crops (Table VII) and 3 m^2/kg dry weight for forage (Table VII), $\lambda_i = 8.92 \times 10^{-9}$ d^{-1} (Annex II) and $\lambda_w = 0.05$ d^{-1} (Table VII), so $\lambda_{E^v} = \lambda_i + \lambda_w = 8.92 \times 10^{-9}$ [d^{-1}] + 5 $\times$ 10^{-2} [d^{-1}] = 5 $\times$ 10^{-2} [d^{-1}], $t_e = 60$ d for food crops (Table VIII) and 30 d for forage (Table VIII).

$$C_{v,1} = \frac{1.3 \times 10^{-3} \, [\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}] \times 0.3 \, [\text{m}^2/\text{kg}] \times [1 - \exp(-5 \times 10^{-2} [\text{d}^{-1}] \times 60 [\text{d}^{-1}])]}{5 \times 10^{-2} [\text{d}^{-1}]}$$

$$= 7.4 \times 10^{-3} \, \text{Bq/kg fresh weight food crops from direct deposition,}$$

$$C_{v,1} = \frac{1.3 \times 10^{-3} \, [\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}] \times 3 \, [\text{m}^2/\text{kg}] \times [1 - \exp(-5 \times 10^{-2} [\text{d}^{-1}] \times 30 [\text{d}^{-1}])]}{5 \times 10^{-2} [\text{d}^{-1}]}$$

$$= 0.061 \, \text{Bq/kg dry weight pasture from direct deposition}$$

The concentration of ^{99}Tc in vegetation resulting from uptake from soil is given by Eqs (31) and (32).

$$C_{v,2} = F_v \times C_s = F_v \frac{\dot{d}[1 - \exp(-\lambda_{E^s} t_b)]}{\rho \lambda_{E^s}}$$

where F_v = 5 for food crops (fresh weight) (Table XI) and 80 for pasture (dry weight) (Table XI), $\dot{d}$ = 4.3 × 10^{-4} Bq·m^{-2}·d^{-1}, ρ = 100 kg/m^2 of dry weight peat soil (Table IX) for food crops and 50 kg/m^2 dry weight peat soil (Table IX) for pasture, λ_i = 8.92 × 10^{-9} d^{-1} (Annex II), λ_s = 0 (Table X) and λ_{E^s} = $\lambda_i + \lambda_s$ = 8.92 × 10^{-9} d^{-1}.

It is assumed that after irrigation ^{99}Tc is found in soil in non-anionic forms. This assumption is conservative and therefore suitable for screening calculations.

t_b = 30 a ≈ 1.1 × 10^4 d (Table VIII)

$$C_{v,2} = 5 \times \frac{4.3 \times 10^{-4} \,[\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}]}{100\,[\text{kg/m}^2]} \times \frac{1 - \exp(-8.92 \times 10^{-9}\,[\text{d}^{-1}] \times 1.1 \times 10^4\,[\text{d}])}{8.92 \times 10^{-9}\,[\text{d}^{-1}]}$$

$$= 0.24 \text{ Bq/kg fresh weight of food crops from uptake}$$

$$C_{v,2} = 80 \times \frac{4.3 \times 10^{-4} \,[\text{Bq} \cdot \text{m}^{-2} \cdot \text{d}^{-1}]}{50\,[\text{kg/m}^2]} \times \frac{1 - \exp(-8.92 \times 10^{-9}\,[\text{d}^{-1}] \times 1.1 \times 10^4\,[\text{d}])}{8.92 \times 10^{-9}\,[\text{d}^{-1}]}$$

$$= 7.6 \text{ Bq/kg dry weight of pasture from uptake}$$

The concentration of ^{99}Tc in vegetation is produced by both direct deposition from irrigation and uptake from irrigated soil (Eq. (33)). Owing to the long radioactive half-life of ^{99}Tc, the radioactive decay during the delay between harvest and consumption is negligible.

$$C_v = (C_{v,1} + C_{v,2}) \exp(-\lambda_i t_h) \approx C_{v,1} + C_{v,2}$$

where $C_{v,1}$ = 7.4 × 10^{-3} Bq/kg for food crops and 0.061 Bq/kg for pasture and $C_{v,2}$ = 0.24 Bq/kg for food crops and 7.6 Bq/kg for pasture. Therefore C_v = 7.4 × 10^{-3} [Bq/kg] + 0.24 [Bq/kg] ≈ 0.25 Bq/kg fresh weight for food crops and C_v= 0.061 [Bq/kg] + 7.6 [Bq/kg] ≈ 7.7 Bq/kg dry weight for pasture.

The concentration of ^{99}Tc in animal feed is given by Eq. (35). For the case of ^{99}Tc the decrease of the concentration in pasture during the storage time is negligible, owing to the very long half-life of ^{99}Tc. Thus the concentration in stored feed (C_P) is practically identical to the concentration in pasture (C_v) expressed in dry weight.

$$C_a = f_p\, C_v + (1 - f_p)\, C_p \approx C_v = 7.7 \text{ Bq/kg of dry animal feed}$$

The concentration of ^{99}Tc in milk is given by Eq. (36). Again, the radioactive decay can be neglected.

$$C_M = F_m \, (C_a \, Q_M + C_w \, Q_w) \exp(-\lambda_i t_m) \approx F_m \, (C_a \, Q_M + C_w \, Q_w)$$

where $F_m = 0.001$ d/L (Table XI), $C_a = 7.7$ Bq/kg, $Q_M = 16$ kg/d (Table XII), $C_w = 1$ Bq/m^3 and $Q_w = 0.06$ m^3/d (Table XII). $C_M = 0.001$ [d/L] (7.7 [Bq/kg] $\times$ 16 [kg/d] + 1 [Bq/m^3] $\times$ 0.06 [m^3/d]) = 0.001 [d/L] (123 [Bq/d] + 0.06 [Bq/d]) = 0.12 Bq/L for milk.

The concentration of ^{99}Tc in meat is given by Eq. (37). The radioactive decay is negligible.

$$C_F = F_f \, (C_a Q_f + C_w Q_w) \exp(-\lambda_i t_f) \approx F_f \, (C_a Q_f + C_w Q_w)$$

where $F_f = 0.001$ d/kg (Table XI), $C_a = 7.7$ Bq/kg, $Q_f = 12$ kg/d (Table XII), $C_w = 1$ Bq/m^3 and $Q_w = 0.04$ m^3/d (Table XII). $C_F = 0.001$ [d/kg] $\times$ (7.7 [Bq/kg] $\times$ 12 [kg/d] + 1 [Bq/m^3] $\times$ 0.04 [m^3/d]) = 0.001 [d/kg] $\times$ (92 [Bq/d] + 0.04 [Bq/d]) = 0.092 Bq/kg for meat.

IV–10.3. Summary

$C_{\text{fish}} = 0.050$ Bq/kg, $C_{\text{food crops}} = 0.24$ Bq/kg (fresh weight)

$C_{\text{milk}} = 0.12$ Bq/L, $C_{\text{meat}} = 0.092$ Bq/kg

For ^{99}Tc the most important mechanism determining the concentration of radionuclides in plants is uptake from soil, rather than direct deposition on to foliar surfaces.

IV–11. EXAMPLE INDIVIDUAL DOSE CALCULATION

IV–11.1. Scenario description

Iodine-131 is released at a rate of 1 Bq/s over an entire year from a 60 m stack, and a farm is located 1 km downwind; assuming that people live the entire year at that point, what annual effective dose will be received by an infant and by an adult, respectively?

IV–11.2. Calculational procedure

IV–11.2.1. Concentrations of radionuclides in air and on the ground

Assuming no building wake effects

$$C_A = \frac{P_P F Q_i}{u_a}, \; \dot{d} = C_A V_T$$

assuming P_p = 0.25, u_a = 2 m/s and F = 1 × 10^{-5} m^{-2} (from Table I), C_A = 1.3 × 10^{-6} Bq/m^3 and $\dot{d}$ = 1.3 × 10^{-6} × 1000 = 1.3 × 10^{-3} Bq·m^{-2}·d^{-1}

$$C_{gr} = \frac{\dot{d}[1 - \exp(-\lambda_{E^s} t_B)]}{\lambda_{E^s}}$$

λ_{E^s} = $\lambda_i + \lambda_s$ = 0.0862 d^{-1} (Annex II) + 0.0014 d^{-1} (Table X) = 0.088 d^{-1}, t_B = 1.1 × 10^4 d (Table VIII). Therefore

$$C_{gr} = \frac{(1.3 \times 10^{-3})\{1 - \exp[(-0.088)(1.1 \times 10^4)]\}}{0.088}$$

$$= \frac{1.3 \times 10^{-3}}{0.088} = 0.015 \, \text{Bq/m}^2$$

IV–11.2.2. External dose from immersion in the plume

$$E_{im} = C_A \, DF_{im} O_f$$

assuming DF_{im} = 5.8 × 10^{-7} Sv/Bq (from Table XV) and O_f = 1. E_{im} = 1.3 × 10^{-6} × 5.8 × 10^{-7} × 1 = 7.5 × 10^{-13} Sv/a.

IV–11.2.3. Dose from inhalation

The dose from inhalation is

$$E_{inh} = C_A \, R_{inh} \, DF_{inh}$$

assuming R_{inh} = 1400 m^3/a for infants and 8400 m^3/a for adults (Table XIV) and D_{inh} = 7.2 × 10^{-8} Sv/Bq for infants and 7.4 × 10^{-9} Sv/Bq for adults (from Table XVI). E_{inh} = 1.3 × 10^{-10} for infants and 8.1 × 10^{-11} Sv/a for adults.

IV–11.2.4. External dose from ground deposition

The external dose from ground deposition is

$$E_{gr} = C_{gr} \, DF_{gr} O_f$$

assuming $DF_{gr} = 1.2 \times 10^{-8}$ Sv/a per Bq/m^2 (Table XV) and $O_f = 1$. $E_{gr} = 0.015 \times 1.2 \times 10^{-8} \times 1 = 1.8 \times 10^{-10}$ Sv/a.

IV–11.2.5. Dose from food ingestion

The dose from food ingestion is

$$E_{ing, \, p} = C_{p,i} \, H_p D_{ing}$$

Three food types are considered: crops, milk and meat. This calculation may be achieved by multiplying the example concentrations per unit deposition rate in Section 5 by the deposition rate of 1.3×10^{-3} Bq·m^{-2}·d^{-1}.

$D_{ing} = 1.8 \times 10^{-7}$ for infants and 2.2×10^{-8} for adults (Table XVII)

(a) Crops: the concentration for unit deposition is 0.63 Bq/kg. Thus $C_{p,i} = 0.63 \times 1.3 \times 10^{-3} = 8.2 \times 10^{-4}$ Bq/kg, $H_p = 150$ for infants and 410 kg/a for adults (Table XIV). Thus $E_{ing, \, crops} = 8.2 \times 10^{-4} \times 150 \times 1.8 \times 10^{-7} = 2.2 \times 10^{-8}$ Sv/a for infants and $8.2 \times 10^{-4} \times 410 \times 2.2 \times 10^{-8} = 7.4 \times 10^{-9}$ Sv/a for adults.

(b) Milk: the concentration for unit deposition is 2.2 Bq/L. Thus $C_{p,i} = 2.2 \times 1.3 \times 10^{-3} = 2.9 \times 10^{-3}$ Bq/L, $H_p = 300$ for infants and 250 L/a for adults (Table XIV). Thus $E_{ing, \, milk} = 2.9 \times 10^{-3} \times 300 \times 1.8 \times 10^{-7} = 1.6 \times 10^{-7}$ Sv/a for infants and $2.9 \times 10^{-3} \times 250 \times 2.2 \times 10^{-8} = 1.6 \times 10^{-8}$ Sv/a for adults.

(c) Meat: the concentration for unit deposition is 1.6 Bq/kg. Thus $C_{p,i} = 1.6 \times 1.3 \times 10^{-3} = 2.1 \times 10^{-3}$ Bq/kg, $H_p = 40$ for infants and 100 kg/a for adults (Table XIV). Thus $E_{ing, \, meat} = 2.1 \times 10^{-3} \times 40 \times 1.8 \times 10^{-7} = 1.5 \times 10^{-8}$ Sv/a for infants and $2.1 \times 10^{-3} \times 100 \times 2.2 \times 10^{-8} = 4.6 \times 10^{-9}$ Sv/a for adults.

$$E_{ing} = \sum_p E_{ing,p}$$

$E_{ing} = 2.0 \times 10^{-7}$ for infants and 2.8×10^{-8} for adults

IV–11.2.6. Total dose

The total individual dose is

$$E_{tot} = E_{im} + E_{inh} + E_{gr} + E_{ing}$$

$$E_{tot} = 7.5 \times 10^{-13} + 1.3 \times 10^{-10} + 1.8 \times 10^{-10} + 2.0 \times 10^{-7} = 2.0 \times 10^{-7} \text{ Sv/a for infants}$$

and

$$E_{tot} = 7.5 \times 10^{-13} + 8.1 \times 10^{-11} + 1.8 \times 10^{-10} + 2.8 \times 10^{-8} = 2.8 \times 10^{-8} \text{ Sv/a for adults}$$

IV–12.   EXAMPLE COLLECTIVE DOSE CALCULATION

IV–12.1. Scenario description

A facility intends to discharge 4×10^3 Bq/a of ^{131}I to the atmosphere and 6×10^4 Bq/a into the sea. What are the approximate collective doses due to these discharges?

IV–12.2. Calculational procedure

For discharges to the atmosphere, the screening collective effective dose commitment per unit activity of ^{131}I discharged is 10^{-12} man·Sv/Bq (Table XIX).
The collective effective dose commitment for each year of discharge is therefore

$$10^{-12} \times (4 \times 10^3) = 4 \times 10^{-9} \text{ man·Sv}$$

If the discharge continued for 30 years, the total collective dose commitment would be

$$30 \times (4 \times 10^{-9}) = 1.2 \times 10^{-7} \text{ man·Sv}$$

Similarly, for discharges into the sea, the screening collective dose commitment per unit activity of liquid ^{131}I discharged is 10^{-16} man·Sv/Bq (Table XX). The collective effective dose commitment for each year of discharge is therefore

$$10^{-16} \times (6 \times 10^4) = 6 \times 10^{-12} \text{ man·Sv}$$

If the discharge continued for 30 years, the total collective dose commitment would be

$$30 \times (6 \times 10^{-11}) = 1.8 \times 10^{-10} \text{ man·Sv}$$

Annex V

DESCRIPTION OF THE GAUSSIAN PLUME MODEL

For a continuous release from an elevated point source under constant wind velocity and atmospheric conditions the Gaussian plume model [V–1, V–2] may be represented by

$$C_A(x,y,z) = \frac{Q_i}{2\pi\sigma_y\sigma_z u} \exp\left(-\frac{y^2}{2\sigma_y^2}\right)\left\{\exp\left[-\frac{(z-H)^2}{2\sigma_z^2}\right] + \exp\left[-\frac{(z+H)^2}{2\sigma_z^2}\right]\right\} \quad \text{(V–1)}$$

where

$C_A(x,y,z)$ is the air concentration (Bq/m^3) at a point (x,y,z) downwind of the release;

x is the downwind distance (m);

y is the crosswind distance (m);

z is the height above ground (m);

Q_i is the release rate for radionuclide i (Bq/s);

σ_y, σ_z are the diffusion parameters (m), which are a function of downwind distance x and atmospheric stability;

u is the mean wind speed (m/s);

H is the height of release (m).

The value of H used in the Gaussian plume model is the physical stack height of the release point plus an allowance for any additional plume rise owing to momentum, for example fan driven exhausts, or buoyancy for significantly heated exhausts. In most cases involving the release of radionuclides the effective stack height corresponds closely to the physical stack height.

The diffusion parameters used in the Gaussian plume model incorporate the basic assumption of the model that a plume spreads both laterally and vertically with a Gaussian distribution. The form of the model represented by Eq. (V–1) includes reflection of the plume at ground level.

In theory, the Gaussian plume model is limited to rather simple dispersion situations [V–3].

(a)   Dispersion over flat, non-complex terrain;
(b)   Short range transport (about 100 m to 20 km downwind);
(c)   Steady state meteorological conditions;
(d)   No elevated temperature inversions;
(e)   Quasi-continuous releases;

(f)   Transport and mixing in the lee of isolated point sources;

(g)   Non-depositing materials, such as noble gases.

In practice, however, the Gaussian plume model has been successfully applied to a large variety of dispersion problems. This is possible because the model is firmly supported by extensive experimental data, as well as being one of the most widely validated of the general dispersion models [V–4].

The diffusion parameters used in Eq. (V–1) are of critical importance in the applicability of a model to a given assessment situation. A number of different sets of such parameters are available in the literature for a variety of release locations, such as flat, grassy fields, rural villages and urban locations. Comparisons of air concentration predictions using a variety of diffusion parameter sets have shown large differences among sets [V–5].

All diffusion parameters are a function of atmospheric stability. Ideally, the stability of the atmosphere should be considered a continuous variable, like temperature or wind speed. Traditionally, however, the stability of the atmosphere is most often parameterized in terms of discrete stability classes. These classes range from A, most unstable, to F or G, most stable. Class D represents neutral stability. Stability classes may be estimated using a variety of common meteorological measurements, such as solar insolation or wind speed, but all are potentially subject to errors in classifying the true conditions of the atmosphere [V–6].

To calculate annual averaged air concentrations as a function of downwind distance from a source, it is common to use statistical summaries of annual averaged meteorological data. Since wind directions are traditionally reported in terms of one of 12 or 16 cardinal directions (N, NNE, NE, etc.), the concentration is averaged across the resulting wind direction sector. Further, for each reported stability category, the wind speeds measured are divided into specific wind speed categories. The joint frequency distribution of the wind directions, atmospheric stability categories and wind speed categories may be used to calculate the annual averaged ground level air concentration as follows.

$$C_A = \sqrt{\frac{2}{\pi}} \frac{N_p Q_i}{2\pi x} \sum_{j=1}^{N_j} \frac{\exp(-H^2/2\sigma_{zj}^2)}{\sigma_{zj}} \sum_{k=1}^{N_k} \frac{P_{pjk}}{u_k} \qquad (V–2)$$

where

Q_i is the annual average release rate for radionuclide i (Bq/s);

C_A is the annual average ground level air concentration (Bq/m³) at downwind distance x in sector p;

N_p is the number of wind direction sectors, usually 12 or 16;

P_{pjk} is the joint frequency of wind direction sector p, stability category j and wind speed class k;

N_k is the number of wind speed classes;

N_j is the number of stability categories;

u_k is the wind speed associated with wind speed class k (m/s);

σ_{zj} is the vertical diffusion parameter (m) associated with stability category j.

Equation (V–2) is often used in environmental radiological dose assessment calculations. However, proper use of this form of the Gaussian plume model requires appropriate long term meteorological information for the site of interest. In addition, a computer may be necessary for managing all of the individual terms that must be summed in order to obtain the final average air concentration estimate. Equation (V–2) forms the basis for most of the screening procedures presented in Section 3.

REFERENCES

[V–1] GIFFORD, F.A., "An outline of theories of diffusion in the lower layers of the atmosphere", Meteorology and Atomic Energy (SLADE, D.H., Ed.), Rep. TID-24 190, United States Army Environmental Center, Washington, DC (1968).

[V–2] PASQUILL, F., SMITH, F.B., Atmospheric Diffusion, 3rd edn, Ellis Horwood, Chichester (1983).

[V–3] TILL, J.E., MEYER, H.R. (Eds), Radiological Assessment — A Textbook on Environmental Dose Analysis, Rep. NUREG/CR-3332, ORNL-5968, United States Nuclear Regulatory Commission, Washington, DC (1983).

[V–4] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment, Rep. No. 76, NCRP, Bethesda, MD (1984).

[V–5] VOGT, K.J., Empirical investigations of the diffusion of waste air plumes in the atmosphere, Nucl. Technol. **34** (1977) 43–57.

[V–6] INTERNATIONAL ATOMIC ENERGY AGENCY, Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases: Exposures of Critical Groups, Safety Series No. 57, IAEA, Vienna (1982).

Annex VI

RADIONUCLIDE TRANSPORT IN SURFACE WATERS

The mathematical expressions discussed in Section 4 for rivers, estuaries, coastal waters, small lakes and large lakes are derived in this annex, and the associated assumptions are outlined. In general, radionuclide transport and fate in surface waters may be expressed in the following three dimensional advection–diffusion equation [VI–1, VI–2].

$$
\frac{\partial C_{w,\,tot}}{\partial t} + U \frac{\partial C_{w,\,tot}}{\partial x} + V \frac{\partial C_{w,\,tot}}{\partial y} + W \frac{\partial C_{w,\,tot}}{\partial z}
$$

$$
= \varepsilon_x \frac{\partial^2 C_{w,\,tot}}{\partial x^2} + \varepsilon_y \frac{\partial^2 C_{w,\,tot}}{\partial y^2} + \varepsilon_z \frac{\partial^2 C_{w,\,tot}}{\partial z^2} - \lambda_i C_{w,\,tot} + S \qquad\text{(VI–1)}
$$

where

$C_{w,\,tot}$ is the radionuclide concentration (Bq/m^3);

U, V, W are the flow velocities in the x, y and z directions, respectively (m/s);

S is the radionuclide addition or subtraction, for example production of a daughter product (Bq·m^{-3}·s^{-1});

t is the time (s);

x, y, z are the longitudinal, lateral and vertical directions, respectively, in Cartesian co-ordinates (m);

$\varepsilon_x, \varepsilon_y, \varepsilon_z$ are the dispersion coefficients in the x, y and z directions, respectively (m^2/s);

λ_i is the radionuclide decay constant (s^{-1}).

Various simplifications were applied to this general governing equation to obtain the appropriate solutions used in this Safety Report. The radionuclide transport models described here are for use in estimating radionuclide concentrations for steady, continuous releases of radionuclides into each of the five types of surface water with steady state flow conditions. For time varying radionuclide release cases radionuclide concentrations in a receiving surface water body may be obtained by using the convolution technique [VI–3].

VI–1. RIVERS

VI–1.1. Basic river characteristics

To obtain radionuclide concentrations, the river channel and flow characteristics (such as river discharge, velocity, width and dispersion coefficients) must be determined. Although it is desirable to use locally measured river characteristics, the following methods are provided to obtain default values of river discharge, depth and velocity with known or assumed river width.

Most rivers experience a wide range of flows, causing a large variation of width, depth and velocity of flows. However, it is generally accepted that the width, depth and velocity increase with a river discharge as power functions for a wide range of mean annual river flow rates [VI–4]. The following relationships were derived, based on the data of Leopold et al. [VI–4].

$$D = 0.163 q_r^{0.447} \tag{VI–2}$$

$$B = 10 q_r^{0.460} \tag{VI–3}$$

$$U = \frac{q_r}{DB} \tag{VI–4}$$

where

B is the river width (m),
D is the water depth (m),
q_r is the river flow rate (m³/s),
U is the flow velocity (m³/s).

To estimate the river flow rate, width and depth under the 30 year low annual river flow rate, the following assumptions are made [VI–5].

— The 30 year low annual river flow rate q_r is one third of the mean annual river flow rate;
— The relationships between the river flow rate, width and depth for the mean annual flow rate developed by Leopold et al. [VI–4] are also valid when applied to the same parameters in relation to the 30 year low annual river flow rate.

The corresponding flow velocity (U) was obtained using the far right side of Eq. (VI–4) with the known discharge, depth and width. These data and assumptions were used to obtain default values of river width, depth, discharge and velocity, as described in Section 4.3.1.

VI–1.2. Dispersion coefficients and complete mixing distances

Dispersion coefficients vary significantly from river to river, or with river channel and flow conditions even in the same river [VI–6]. Examples of longitudinal and lateral dispersion coefficients for different rivers are shown in Tables VI–I and VI–II [VI–3, VI–6, VI–7].

The most common expression of vertical dispersion coefficient in a river is [VI–3]

$$\varepsilon_z = 0.067u_*D \tag{VI–5}$$

where

ε_z is the vertical dispersion coefficient (m^2/s),
u_* is the shear velocity (m/s).

Longitudinal and lateral dispersion coefficients in rivers vary by several orders of magnitude from small creeks to large rivers such as the Mississippi and Columbia Rivers, and there are many formulations available to estimate these coefficients [VI–3, VI–5, VI–6]. The following expressions provide reasonably good estimates of the longitudinal and lateral dispersion coefficients over a wide range of river conditions [VI–3, VI–6].

$$\varepsilon_x = \frac{U^2B^2}{30Du_*} \tag{VI–6}$$

$$\varepsilon_y = \alpha Du_* \tag{VI–7}$$

where

ε_x, ε_y are the longitudinal and lateral dispersion coefficients, respectively (m^2/s);
α is the proportionality coefficient.

The proportionality coefficient α between ε_y and Du_* in Eq. (VI–7) varies with a width to depth ratio ranging from 0.1 to 0.2 for small laboratory flumes and medium sized irrigation canals to 0.6 to 2.0 for the Missouri and MacKenzie Rivers [VI–3, VI–7]. In this report we have selected $\alpha = 0.6$. Assuming that [VI–7]

$$u_* = 0.1U \tag{VI–8}$$

and substituting Eq. (VI–8) into Eq. (VI–5) yields

TABLE VI–I. EXAMPLES OF LONGITUDINAL DISPERSION COEFFICIENTS IN RIVERS

Channel	Width, B (m)	Mean depth, D (m)	Mean velocity, U (m/s)	Shear velocity, u_* (m/s)	Dispersion coefficient, ε_x (m²/s)	ε_x/Du_*
Chicago Ship Canal	48.8	8.07	0.27	0.0191	3.0	20
Sacramento River		4.00	0.53	0.051	15	74
River Derwent		0.25	0.38	0.14	4.6	131
South Platte River		0.46	0.66	0.069	16.2	510
Yuma Mesa A Canal		3.45	0.68	0.345	0.76	8.6
Green–Duwamish River	20	1.10		0.049	6.5–8.5	120–160
Missouri River	200	2.70	1.55	0.074	1500	7500
Copper Creek	16	0.49	0.27	0.080	20	500
	18	0.85	0.60	0.100	21	250
	16	0.49	0.26	0.080	9.5	245
	19	0.40	0.16	0.116	9.9	220
Clinch River, Tennessee	47	0.85	0.32	0.067	14	235
	60	2.10	0.94	0.104	54	245
	53	2.10	0.83	0.107	47	210
Clinch River, Virginia	36	0.58	0.21	0.049	8.1	280
Powell River	34	0.85	0.15	0.055	9.5	200
Coachella Canal	24	1.56	0.71	0.043	9.6	140
Bayou Anacoco	26	0.94	0.34	0.067	33	520
	37	0.91	0.40	0.067	39	640
Nooksack River	64	0.76	0.67	0.27	35	170
Wind/Bighorn Rivers	59	1.10	0.88	0.12	42	320
	69	2.16	1.55	0.17	160	440
John Day River	25	0.58	1.01	0.14	14	170
	34	2.47	0.82	0.18	65	150
Comite River	16	0.43	0.37	0.05	14	650
Sabine River	104	2.04	0.58	0.05	315	3100
	127	4.75	0.64	0.08	670	1800
Yadkin River	70	2.35	0.43	0.10	110	470
	72	3.84	0.76	0.13	260	520

TABLE VI–II. EXAMPLES OF LATERAL DISPERSION COEFFICIENTS IN RIVERS

River	Channel description	Channel, width, B (m)	Mean depth, D (m)	Mean velocity, U (m/s)	Shear velocity, u_* (m/s)	Lateral dispersion coefficient, ε_y (m²/s)	ε_y/Du_*
Missouri River near Blair, Nebraska	Meandering river	200	2.7	1.75	0.074	0.12	0.6
Ijssel River	Groins on sides and gentle curvature	69.5	4.0	0.96	0.075	0.15	0.51
Mackenzie River from Fort Simpson to Norman Wells	Generally straight alignment or slight curvature; numerous islands and sand bars	1240	6.7	1.77	0.152	0.67	0.66
Missouri River downstream of Cooper Nuclear Station, Nebraska	Reach includes one 90° and one 180° bend	210–270	4.0	5.4	0.08	1.1	3.4
Potomac River; 29 km reach below the Dickerson Power Plant	Gently meandering river with up to 60° bends	350	0.73–1.74	0.29–0.58	0.033–0.051	0.013–0.058	0.52–0.65
Aristo Feeder Canal		18.3	0.67	0.67	0.062	0.0093	0.22
Bernado Conveyance Channel		20.1	0.70	1.25	0.061	0.013	0.30
Athabasca River	Below Fort McMurray	373	2.19	0.95	0.056	0.092	0.75
Athabasca River	Below Athabasca	320	2.05	0.86	0.079	0.066	0.41
North Saskatchewan River	Below Edmonton	213	1.55	0.58	0.080	0.031	0.25
Bow River	At Calgary	104	1.00	1.05	0.139	0.085	0.61
Beaver River	Near Cold Lake	42.7	0.96	0.50	0.044	0.042	0.99
Grand River	Below Kitchener	59.2	0.51	0.35	0.069	0.0090	0.26
Columbia River		305	3.05	1.35	0.088	0.20	0.75
Missouri River	Two mild alternating bends	183	2.66	1.74	0.073	0.12	0.62
Missouri River	Sinuous, severe bends	234	3.96	1.98	0.042	0.55	3.3
South River	Few mild bends	18.2	0.40	0.21	0.040	0.0048	0.30
Lesser Slave River	Contorted meander	43.0	2.53	0.65	0.049	0.041	0.33
Mobile River	Mostly straight, one mild curve	430	4.93	0.30	0.018	0.64	7.2

$$\varepsilon_z = 0.0067UD \qquad\qquad (VI–9)$$

Similarly, substituting Eq. (VI–8) into Eqs (VI–6) and (VI–7) yields the following expressions of longitudinal and lateral dispersion coefficients for rivers.

$$\varepsilon_x = \frac{U^2B^2}{30Du_*} \qquad\qquad (VI–10)$$

$$\varepsilon_y = 0.06DU \qquad\qquad (VI–11)$$

To obtain longitudinal distances for achieving relatively complete mixing, it is assumed that complete lateral and vertical mixing is achieved when the minimum concentration is at least one half of the maximum concentration along the same vertical and lateral lines, respectively. Assuming that the radionuclide is released from one of the river banks at a depth of $D/2$, the longitudinal distances required to achieve this complete lateral and vertical mixing are given by L_y and L_z, respectively.

$$L_y = 0.18\frac{UB^2}{\varepsilon_y} \qquad\qquad (VI–12)$$

$$L_z = 0.045\frac{UD^2}{\varepsilon_z} \qquad\qquad (VI–13)$$

As discussed later, these distances were obtained by solving the two dimensional advection–diffusion equations with mirror image source technology [VI–3]. Substituting Eqs (VI–11) and (VI–9) into Eqs (VI–12) and (VI–13) yields the following results.

$$L_z = 7D \qquad\qquad (VI–14)$$

$$L_y = 3\frac{B^2}{D} \qquad\qquad (VI–15)$$

Since almost any river has a width greater than its depth, the distance L_y is greater than L_z. Thus the concentration in a region downstream of the distance L_y is completely mixed in both vertical and lateral directions, thus completely mixed over the river cross-section.

In the region where $x < L_z$ a radionuclide distribution is still three dimensional. However, since L_z is only seven times the water depth near the river bank, a radionuclide in this region is assumed to have no dilution.

VI–1.3. Governing equation and its solution after complete vertical mixing $(x > L_z)$

After the radionuclide is completely mixed vertically, the radionuclide transport equation is reduced from the three dimensional Eq. (VI–1) to the following two dimensional advection–diffusion equation with radionuclide decay but without other radionuclide subtraction and addition.

$$U \frac{\partial C_{w,\, tot}}{\partial x} = \varepsilon_x \frac{\partial^2 C_{w,\, tot}}{\partial x^2} + \varepsilon_y \frac{\partial^2 C_{w,\, tot}}{\partial y^2} - \lambda_i C_{w,\, tot} \qquad (VI–16)$$

where

$C_{w,\, tot}$ is the radionuclide concentration (Bq/m^3);
x, y are the longitudinal and lateral directions (m), respectively;
λ_i is the radionuclide decay constant (s^{-1}).

Assuming that the flow field is infinitely wide and that the receiving surface water has no initial radionuclide content, Eq. (VI–16) yields the radionuclide concentration at a given location [VI–5].

$$C_{w,\, tot}(x, y) = \frac{Q}{2\pi D \sqrt{\varepsilon_x \varepsilon_y}} \exp\left(\frac{U_x}{2\varepsilon_x} - \frac{\lambda_i x}{U} \right)$$

$$\times K_0 \left[\frac{U}{2\varepsilon_x} \sqrt{x^2 + \frac{\varepsilon_x}{\varepsilon_y}(y - y_0)^2} \right] \qquad (VI–17)$$

where

Q is the radionuclide release rate (Bq/s).
y_0 is the lateral distance of the radionuclide release point measured from the river bank.
$K_0[\ ]$ is the modified Bessel function of the second kind of the zeroth order. Values of $K_0[\ ]$ are presented in Table VI–III.

Once a radionuclide plume reaches the river banks, the radionuclide spread is restricted. In this case, instead of Eq. (VI–17), a solution to Eq. (VI–16) can be calculated by using the reflection or mirror image source technique [VI–3] as below (Eq. (VI–18)).

TABLE VI–III. MODIFIED BESSEL FUNCTIONS OF THE SECOND KIND OF THE ZEROTH ORDER

Modified Bessel functions		Modified Bessel functions		Modified Bessel functions	
x	$e^x K_0(x)$	x	$e^x K_0(x)$	x	$e^x K_0(x)$
0.0	∞	3.6	0.64 045 596 47	7.2	0.45 953 077 56
0.1	2.68 232 610 23	3.7	0.63 221 805 91	7.3	0.45 646 856 18
0.2	2.14 075 732 33	3.8	0.62 429 158 12	7.4	0.45 346 685 94
0.3	1.85 262 730 07	3.9	0.61 665 731 47	7.5	0.45 052 369 91
0.4	1.66 268 208 91	4.0	0.60 929 766 93	7.6	0.44 763 719 96
0.5	1.52 410 938 57	4.1	0.60 219 650 64	7.7	0.44 480 556 36
0.6	1.41 673 762 14	4.2	0.59 533 898 89	7.8	0.44 202 707 24
0.7	1.33 012 365 62	4.3	0.58 871 144 86	7.9	0.43 930 008 19
0.8	1.25 820 312 16	4.4	0.58 230 127 04	8.0	0.43 662 301 85
0.9	1.19 716 338 03	4.5	0.57 609 678 97	8.1	0.43 399 437 54
1.0	1.14 446 307 97	4.6	0.57 008 720 22	8.2	0.43 141 270 84
1.1	1.09 833 028 28	4.7	0.56 426 248 40	8.3	0.42 887 663 29
1.2	1.05 748 453 22	4.8	0.55 861 331 94	8.4	0.42 638 482 14
1.3	1.02 097 316 13	4.9	0.55 313 103 97	8.5	0.42 393 599 93
1.4	0.98 806 999 61	5.0	0.54 780 756 43	8.6	0.42 152 894 33
1.5	0.95 821 005 33	5.1	0.54 263 535 19	8.7	0.41 916 247 81
1.6	0.93 094 598 08	5.2	0.53 760 735 40	8.8	0.41 683 547 43
1.7	0.90 591 813 86	5.3	0.53 271 697 44	8.9	0.41 454 684 62
1.8	0.88 283 352 70	5.4	0.52 795 803 29	9.0	0.41 229 554 93
1.9	0.86 145 061 68	5.5	0.52 332 473 16	9.1	0.41 008 057 83
2.0	0.84 156 821 51	5.6	0.51 881 162 52	9.2	0.40 790 096 62
2.1	0.82 301 715 25	5.7	0.51 441 359 38	9.3	0.40 364 412 45
2.2	0.80 565 398 12	5.8	0.51 012 581 83	9.5	0.40 156 513 22
2.3	0.78 935 613 12	5.9	0.50 594 375 83	9.6	0.39 951 796 93
2.4	0.77 401 814 07	6.0	0.50 186 313 09	9.7	0.39 750 183 13
2.5	0.75 954 869 03	6.1	0.49 787 989 29	9.8	0.39 551 594 16
2.6	0.74 586 824 30	6.2	0.49 399 022 37	9.9	0.39 355 955 06
2.7	0.73 290 715 15	6.3	0.49 019 050 93	10.0	0.39 163 193 44
2.8	0.72 060 412 51	6.4	0.48 647 732 91	10.2	0.38 786 025 39
2.9	0.70 890 497 74	6.5	0.48 284 744 13	10.4	0.38 419 558 46
3.0	0.69 776 159 80	6.6	0.47 929 777 29	10.6	0.38 063 295 49
3.1	0.68 713 110 10	6.7	0.47 582 540 66	10.8	0.37 716 771 25
3.2	0.67 697 511 39	6.8	0.47 242 757 23	11.0	0.37 379 549 71
3.3	0.66 725 918 31	6.9	0.46 910 163 70	11.2	0.37 051 221 56
3.4	0.65 795 227 25	7.0	0.46 584 509 59	11.4	0.36 731 402 43
3.5	0.64 902 633 77	7.1	0.46 265 556 57	11.6	0.36 419 730 76

TABLE VI–III. (cont.)

Modified Bessel functions		Modified Bessel functions		Modified Bessel functions	
x	$e^x K_0(x)$	x	$e^x K_0(x)$	x	$e^x K_0(x)$
11.8	0.36 115 866 16	14.6	0.32 530 020 91	17.4	0.29 836 652 76
12.0	0.35 819 487 84	14.8	0.32 312 983 64	17.6	0.29 668 936 57
12.2	0.35 530 293 18	15.0	0.32 100 235 34	17.8	0.29 504 018 17
12.4	0.35 247 996 43	15.2	0.31 891 636 55	18.0	0.29 341 820 62
12.6	0.34 972 327 46	15.4	0.31 687 054 05	18.2	0.29 182 269 87
12.8	0.34 703 030 81	15.6	0.31 486 360 51	18.4	0.29 025 294 72
13.0	0.34 439 864 55	15.8	0.31 289 434 24	18.6	0.28 870 826 54
13.2	0.34 182 599 43	16.0	0.31 096 158 80	18.8	0.28 718 799 33
13.4	0.33 931 018 06	16.2	0.30 906 422 69	19.0	0.28 569 149 44
13.6	0.33 684 914 05	16.4	0.30 720 119 19	19.2	0.28 421 815 54
13.8	0.33 444 091 42	16.6	0.30 537 145 92	19.4	0.28 276 738 48
14.0	0.33 208 363 83	16.8	0.30 357 404 87	19.6	0.28 133 861 17
14.2	0.32 977 554 02	17.0	0.30 180 801 93	19.8	0.27 993 128 62
14.4	0.32 751 493 32	17.2	0.30 007 246 78	20.0	0.27 854 487 66

$$C_{w,\text{tot}} = C_{w,\text{tot}}(x,y) + \sum_{n=1}^{\infty} \left\{ C\left[x, nB - \left(y_0 - \frac{B}{2} \right) + (-1)^n \left(y - \frac{B}{2} \right) \right] \right.$$

$$\left. + C\left[x, -nB - \left(y_0 - \frac{B}{2} \right) + (-1)^n \left(y - \frac{B}{2} \right) \right] \right\} \qquad \text{(VI–18)}$$

where

B   is the river width,
n   is the number of reflection cycles.

Note that there is generally no significant contribution to $C_{w,\text{tot}}$ from terms with $n > 4$ or 5. By using Eqs (VI–17) and (VI–18) radionuclide concentrations released at $y = y_0$ can be determined at any location within a river.

A radionuclide concentration will be the highest along the centre of the radionuclide plume at any river cross-section. To obtain a radionuclide concentration along the centre of the plume, $y = y_0$ is entered into Eqs (VI–17) and (VI–18). Assuming that the radionuclides are released from a river bank (i.e. $y_0 = 0$), as a conservative measure, Eqs (VI–17) and (VI–18) yield the following approximate radionuclide concentrations along the same side of the river bank ($y = 0$) from which the radionuclide is released.

$$C_{w,\text{tot}} \bigg|_{y=0} = \frac{Q}{\pi D \sqrt{\varepsilon_x \varepsilon_y}} \times \exp\left(\frac{U_x}{2\varepsilon_x} - \frac{\lambda_i x}{U} \right) \times K_0\left(\frac{U_x}{2\varepsilon_x} \right) \qquad \text{(VI–19)}$$

Equation (VI–19) has been obtained from Eqs (VI–17) and (VI–18) by truncating the series at $n = 1$ and neglecting the reflection from the opposite bank.

Substitution of Eqs (VI–10) and (VI–11) into Eq. (VI–19) yields the following.

$$C_{w,\text{tot}} \bigg|_{y=0} = \frac{Q}{0.142\pi \, DUB} \times \exp\left(\frac{1.5\,Dx}{B^2} - \frac{\lambda_i x}{U} \right) \times K_0\left(\frac{1.5\,Dx}{B^2} \right) \qquad \text{(VI–20)}$$

which can be written as

$$C_{w,\text{tot}} \bigg|_{y=0} = C_t P_r \qquad \text{(VI–21)}$$

where

$$C_t = \frac{Q}{q_r} \exp\left(-\frac{\lambda_i x}{U} \right) \qquad \text{(VI–22)}$$

176

$$P_r = \frac{1}{0.142\pi} \times \exp\left(\frac{1.5\,Dx}{B^2}\right) \times K_0\left(\frac{1.5\,Dx}{B^2}\right) \qquad \text{(VI–23)}$$

$$q_r = DUB = \text{river flow rate} \qquad \text{(VI–24)}$$

Owing to the truncation of series (VI–18), Eq. (VI–23) is valid when the flow field may be assumed to be semi-infinite. For large values of x the opposite bank effect becomes important and other terms of series (VI–18) must be considered to evaluate the partial mixing coefficient P_r.

Equation (VI–21) was selected for the river model. Values of P_r are given in Table IV. Note that C_t in Eq. (VI–22) is the completely mixed radionuclide concentration over a river cross-section. The variable P_r can be regarded as a correction factor for partial mixing and approaches unity as the downstream distance x increases. Note that P_r should be greater than or equal to unity.

When the downstream distance x becomes greater than L_y, complete lateral mixing is achieved, and thus complete mixing over the entire river cross-section. For this case $(x > L_y)$ P_r becomes unity, and Eq. (VI–21) is simplified to

$$C_{w,\,tot} = C_t \qquad \text{(VI–25)}$$

The radionuclide concentration on the opposite side of the river from the original release point is less than or equal to the completely mixed values over the river cross-section obtained from Eq. (VI–25). Thus Eq. (VI–25) is also used for that region.

VI–2. ESTUARIES

VI–2.1. Estuarine conditions

The mathematical model selected for a steady, continuous release of a radionuclide into an estuary is based on the same equation (Eq. (VI–16)) and solutions for the river case discussed in the previous section. The differences from river cases are as follows.

— A tidally averaged flow velocity (net freshwater flow velocity) U is used;
— The vertical dispersion coefficient is evaluated by Eq. (VI–9), but it is based on a mean flowspeed over a tidal cycle;
— Longitudinal and lateral dispersion coefficients are estimated, based on a mean flow speed over a tidal cycle, corrected to reflect tidally varying hydrodynamics;

— Water use locations both upstream and downstream from the radionuclide release point are analysed, and the radionuclide concentration in the upstream location can be corrected for the tidal effect [VI–7, VI–8].

VI–2.2. Dispersion coefficients and complete mixing distances

Dispersion processes occurring in estuaries are very complex, generally more complex than those occurring in rivers, owing to the additional effects of tides, density stratification, wind, and complex channel geometry and bathymetry. There are no general theoretical expressions covering a wide range of estuarine conditions. Thus, although this report provides some default dispersion expressions, the locally estimated values should be used whenever they are available. Table VI–IV presents examples of longitudinal dispersion coefficients in estuaries [VI–7, VI–9].

For the vertical dispersion coefficient Eq. (VI–5) is considered to be valid. However, Eqs (VI–8) and (VI–9) are used for this case by using the mean flow speed over a tidal cycle U_t instead of the 30 year low annual river flow rate used for a river case. Thus

$$u_* = 0.1U_t \tag{VI–26}$$

$$\varepsilon_z = 0.0067U_t D \tag{VI–27}$$

where

$$U_t = 0.32(|U_e| + |U_f|)$$

and U_e and U_f are the maximum ebb and flood velocities (m/s), respectively. Note that the tidal speed is assumed to vary sinusoidally with time.

The lateral mixing in an estuary tends to be several times larger than the corresponding lateral mixing in a non-tidal river, mainly owing to large lateral flows caused by irregular channel geometry and cross-section, tides, density stratification and wind [VI–7]. Thus the proportionality factor α in Eq. (VI–7) is assigned to be 3 for an estuary. Thus

$$\varepsilon_y = 3DU_* \tag{VI–28}$$

Substituting Eq. (VI–26) into Eq. (VI–28) yields the following lateral dispersion coefficient for an estuary.

$$\varepsilon_y = 0.3DU_t \tag{VI–29}$$

TABLE VI–IV. LONGITUDINAL DISPERSION COEFFICIENTS FOR ESTUARIES

Estuary	Freshwater inflow, q_r (m³/s)	Tidally averaged velocity, U (m/s)	Longitudinal dispersion coefficient, ε_x (m²/s)
Delaware River, Delaware, USA	70	0.037–0.31	100–1500
Hudson River, New York, USA	140	0.011	450–1500
East River, New York, USA	0	0	300
Cooper River, South Carolina, USA	280	0.076	900
Savannah River, Georgia, South Carolina, USA	200	0.052–0.21	300–600
Lower Raritan River, New Jersey, USA	4	0.0088–0.014	150
South River, New Jersey, USA	0.7	0.0031	150
Houston Ship Channel, Texas, USA	30	0.015	800
Cape Fear River, North Carolina, USA	30	0.0092–0.15	60–300
Compton Creek, New Jersey, USA	0.3	0.0040–0.031	30
Wappinger and Fishkill Creek, New York, USA	0.06	0.00031–0.0012	15–30
Potomac River, Virginia, USA	15	0.00092–0.0018	6–300
San Francisco Bay, California, USA			
Southern arm			20–200
Northern arm			50–2000
Rotterdam Waterway, Netherlands			280
Rio Quayas, Ecuador			760
Severn Estuary, United Kingdom			
Summer			50–120
Winter			120–500
Thames River, United Kingdom			
Low river flow			50–90
High river flow			300
Tay Estuary, United Kingdom			50–150
Narrows of Mersey, United Kingdom			100–400

The longitudinal mixing in an estuary is affected by an oscillating tidal flow. The ratio N of the longitudinal dispersion coefficient in an estuary to that in a river is expressed by the ratio M of the tidal period to the timescale for cross-sectional mixing [VI–7]. Expressing the cross-sectional mixing time by B^2/ε_y, and ε_y by Eq. (VI–29), the timescale ratio M becomes

$$M = \frac{0.3DU_t T_p}{B^2} \tag{VI–30}$$

where T_p is a tidal period. T_p will be 45 000 s for a tide occurring twice per day (those in the east coast of the USA), while it will be 90 000 s for a dominating tide occurring once per day (e.g. those generally occurring along the Gulf of Mexico and the west coast of the USA).

$$M = \frac{13500DU_t}{B^2} \text{ for a twice per day tide}$$
$$= \frac{27000DU_t}{B^2} \text{ for a once per day tide}$$

The ratio of the longitudinal dispersion coefficient in an estuary to that in a river N is obtained as a function of M, as shown in Table V. Since the longitudinal dispersion coefficient in a river is obtained by Eq. (VI–10), the longitudinal dispersion coefficient in an estuary is

$$\varepsilon_x = \frac{NU_t B^2}{3D} \tag{VI–31}$$

Substituting Eq. (VI–29) into Eq. (VI–12) yields the following longitudinal distance L_y required to achieve complete lateral mixing.

$$L_y = 0.6\frac{B^2}{D} \tag{VI–32}$$

Similarly, the longitudinal distance L_z required to achieve complete vertical mixing can be obtained by substituting Eq. (VI–27) into Eq. (VI–13) as

$$L_z = 7D \tag{VI–33}$$

As in the river case, 50% mixing is assumed to be complete mixing.

VI–2.3. Governing equation and its solutions beyond regions of complete vertical mixing ($x > L_z = 7D$)

The mathematical model selected for this case is the same as the river model, Eq. (VI–16), and its associated solutions expressed in Eq. (VI–17). Substituting the

estuarine longitudinal and lateral dispersion coefficients shown in Eqs (VI–31) and (VI–29) into Eq. (VI–17), one can obtain

$$C_{w,\,\text{tot}} = \frac{Q}{0.632\pi\,DU_t B\sqrt{N}} \times \exp\left(\frac{1.5DxU}{NB^2U_t}\right)$$

$$\times K_0\left(\frac{1.5D}{NB^2} \times \frac{U}{U_t}\sqrt{x^2 + \frac{NB^2}{0.9D^2}}(y-y_0)\right) \times \exp\left(-\frac{\lambda_i x}{U}\right) \qquad (\text{VI–34})$$

If the radionuclide dispersal is affected by the estuarine banks, then use Eq. (VI–18) to correct Eq. (VI–34) to obtain a generic solution. Thus, as with the river case (i.e. Eqs (VI–19) and (VI–20)), the radionuclide concentration along the estuarine shore downstream from where a radionuclide is released ($y = y_0$) can be obtained by

$$C_{w,\,\text{tot}} = C_{te}P_e \qquad (\text{VI–35})$$

where the concentration for complete mixing is given by

$$C_{te} = \frac{Q}{q_w}\exp\left(-\frac{\lambda_i x}{U}\right) \qquad (\text{VI–36})$$

and

$$q_w = DBU_t = \text{average tidal discharge} \qquad (\text{VI–37})$$

The factor P_e is given as

$$P_e = \frac{1}{0.32\pi\sqrt{N}} \times \exp\left(\frac{1.5Dx\,U}{NB^2U_t}\right) \times K_0\left(\frac{1.5Dx\,U}{NB^2U_t}\right) \qquad (\text{VI–38})$$

In any case, values of P_e should be chosen higher than one and lower than q_w/F, where F is the radionuclide effluent flow rate in m^3/s. Figure 14 (in Section 4) presents values of P_e as a function of the argument $A = 1.5DxU_a/NB^2U_t$ and N. Equation (VI–35) is used to calculate radionuclide concentrations both upstream and downstream from the radionuclide release point to account for the estuarine bank effects. Note that C_{te} is the concentration of a radionuclide after complete mixing over a cross-section is reached, and P_e can be considered as the correction factor for partial mixing.

Although the estuarine model selected for this Safety Report does not separate upstream radionuclide mixing from downstream mixing, the following correction may be made to improve the estimate of radionuclide concentrations in a region upstream of the radionuclide release point. For radionuclide concentrations upstream from the release point a correction to the partial mixing radionuclide concentration may be performed to Eqs (VI–34) and (VI–35). To estimate upstream radionuclide concentrations, a one dimensional equation is used to derive the necessary correction factor. A solution to the steady state, one dimensional advection–diffusion equation with the first order decay term, Eq. (VI–39), for a continuous release at $x = 0$ is shown in Eq. (VI–40) [VI–10].

$$U\frac{\partial C_{w,\text{tot}}}{\partial x} = \varepsilon_x \frac{\partial^2 C_{w,\text{tot}}}{\partial x^2} - \lambda_i C_{w,\text{tot}} \tag{VI–39}$$

$$C_{w,\text{tot}} = \frac{Q}{BDU\sqrt{1 + \dfrac{4\lambda_i\varepsilon_x}{U^2}}} \times \exp\left[\frac{Ux}{2\varepsilon_x}\left(1 \pm \sqrt{1 + \frac{4\lambda_i\varepsilon_x}{U^2}}\right)\right] \tag{VI–40}$$

Note that the plus and minus signs in the argument of the exponential function in Eq. (VI–40) are associated with upstream and downstream concentrations, respectively. Thus the ratio of upstream to downstream concentrations is

$$UCF = \exp\left(\frac{Ux}{\varepsilon_x}\sqrt{1 + \frac{4\lambda_i\varepsilon_x}{U^2}}\right) \tag{VI–41}$$

Values of x should be taken as negative values to calculate UCF defined in Eq. (VI–41). Substituting Eq. (VI–31) into Eq. (VI–41) gives

$$UCF = \exp\left(\frac{3xDU}{NB^2U_t}\right) \tag{VI–42}$$

Thus the partial mixing of the dissolved radionuclide concentration $C_{wu,\text{tot}}$ in an upstream region before complete mixing over the entire cross-section is

$$C_{wu}(x, y) = UCF \times C_{w,\text{tot}} \tag{VI–43}$$

where $C_{w,\text{tot}}$ is calculated by Eq. (VI–34) or (VI–35). To consider Eq. (VI–43), the upstream distance x must be shorter than the actual distance that the radionuclide can travel during a flood tide period.

VI–3. COASTAL WATERS

The mathematical model selected for coastal waters is based on the following vertically averaged, two dimensional advection–diffusion equation.

$$U \frac{\partial C_{w,\text{tot}}}{\partial x} = \varepsilon_y \frac{\partial^2 C_{w,\text{tot}}}{\partial y^2} - \lambda_i C_{w,\text{tot}} \qquad \text{(VI–44)}$$

where U is the coastal current. The above equation represents a time varying, longitudinally advecting, laterally dispersing concentration balance with radionuclide decay. Note that the longitudinal dispersion is not included in Eq. (VI–44), which results in a conservative estimate of the radionuclide concentration. Since the scale of the mixing length becomes larger as a radionuclide plume spreads further in a coastal water [VI–6, VI–11], the lateral dispersion coefficient is assumed to be a function of the longitudinal distance. The following lateral dispersion coefficient ε_y (m²/s), derived from Okubo's [VI–11] dispersion expression, may be used here as the default value.

$$\varepsilon_y = (3.44 \times 10^{-7}) \left(\frac{x}{U} \right)^{1.34} \qquad \text{(VI–45)}$$

However, if a site specific, scale dependent dispersion coefficient is available, it should be used instead of Eq. (VI–45).

Assume that the

(a) Shoreline is straight along the x axis ($y = 0$),
(b) Water depth D is constant,
(c) Coastal current U is constant and is parallel to the shoreline.

Under these conditions (see Fig. 16, Section 4), the solution to Eq. (VI–44) for a continuous release from a discharge point ($x = 0$, $y = y_0$) [VI–3, VI–12] is

$$C_{w,\text{tot}} = \frac{Q}{D\sqrt{\pi U \varepsilon_y x}} \times \left[\exp\left(-\frac{U(y-y_0)^2}{4\,\varepsilon_y x} - \frac{\lambda_i x}{U} \right) \right] \qquad \text{(VI–46)}$$

The coastal current velocity U should be measured at the study site. If it is not possible to obtain the site specific velocity, $U = 0.1$ m/s may be used as a default value.

The radionuclide concentration expressed in Eq. (VI–46) includes a shoreline effect to restrict lateral dispersion of the radionuclide plume (see Eq. (VI–18)).

Note that Eqs (VI–44) and (VI–46) can be obtained from the more general Eq. (VI–16) by assuming that the longitudinal dispersion is not important; that is imposing the following conditions on Eq. (VI–17).

$$\frac{\varepsilon_y}{\varepsilon_x}\left(\frac{y-y_0}{x}\right)^2 \ll 1 \tag{VI–47}$$

$$\frac{xU}{2\varepsilon_x} \gg 1 \tag{VI–48}$$

Assuming that [VI–5, VI–6]

$$\varepsilon_x = 4.66 \times 10^{-6}\left(\frac{x}{U}\right)^{1.34} \tag{VI–49}$$

and substituting Eqs (VI–45) and (VI–49) with $U = 0.1$ m/s into Eqs (VI–47) and (VI–48), the following conditions can be derived.

$$7D < x < 8 \times 10^7 \text{ m} \tag{VI–50}$$

$$\left|\frac{y-y_0}{x}\right| \ll 3.7 \tag{VI–51}$$

Substituting Eq. (VI–45) into Eq. (VI–46) yields

$$C_{w,\,\text{tot}} = \frac{962U^{0.17}Q}{Dx^{1.17}} \times \exp\left(-\frac{7.28\times10^5\,U^{2.34}(y-y_0)^2}{x^{2.34}} - \frac{\lambda_i x}{U}\right) \tag{VI–52}$$

Radionuclide concentrations along the plume centre and shoreline can be obtained by assigning $y = y_0$ and $y_0 = 0$, respectively, in Eqs (VI–46) or (VI–52).

VI–4. LAKES AND RESERVOIRS

For a small lake or reservoir, the radionuclide concentration is assumed to be uniform within the entire impoundment (see Fig. 17, Section 4). Under this condition, the governing equation of a radionuclide concentration in a lake or reservoir is

$$\frac{dC_{w,\,\text{tot}}}{dt} = -\frac{(q_r + \lambda_i V)\,C_{w,\,\text{tot}}}{V} + \frac{Q}{V} \tag{VI–53}$$

where V is the lake volume and q_r is the river inflow/outflow.

Assuming that at time $t = 0$, $C_{w,\,\text{tot}} = 0$, the solution to Eq. (VI–53) is

$$C_{w,\text{tot}} = \frac{Q}{q_r + \lambda_i V} \left\{ 1 - \exp\left[-\left(\frac{q_r}{V} + \lambda_i \right) t \right] \right\} \tag{VI–54}$$

Because the current method assesses potential long term radiological impacts of the operation of a nuclear facility over its plant lifetime (say 30 years), radionuclides with very short half-lives are not critical to the evaluation. Furthermore, if

$$\left(\frac{q_r}{V} + \lambda_i \right) > 10^{-8} \text{ s}^{-1} \tag{VI–55}$$

then

$$\exp\left[-\left(\frac{q_r}{V} + \lambda_i \right) t \right] << 1 \tag{VI–56}$$

Thus, under this condition, Eq. (VI–54) can be simplified to yield the following steady state solution to Eq. (VI–57).

$$C_{w,\text{tot}} = \frac{Q}{q_r + \lambda_i V} \tag{VI–57}$$

Mixing in a large lake is dominated by a wind induced flow and is similar to that occurring in a coastal water. Thus the long term radionuclide concentration is obtained by Eq. (VI–46) or (VI–52). As discussed in Section 4.6.3, even a large lake can achieve complete mixing within a relatively short time [VI–13]. Concentrations calculated by Eq. (VI–46) (or (VI–52)) and Eq. (VI–54) (or (VI–57)) may be added to include this elevated background radionuclide concentration due to complete mixing.

REFERENCES

[VI–1] INTERNATIONAL ATOMIC ENERGY AGENCY, Hydrological Dispersion of Radionuclide Material in Relation to Nuclear Power Plant Siting, Safety Series No. 50-SG-S6, IAEA, Vienna (1985).

[VI–2] ONISHI, Y., "Contaminant transport modeling in surface waters", Computer Modeling of Free-surface and Pressurized Flows (CHAUDHRY, M.H., MAYS, L.W., Eds), NATO ASI Series E, Applied Sciences, Vol. 274, Kluwer, Dordrecht (1994) 313–341.

[VI–3] SAYRE, N.W., "Natural mixing processes in rivers", Environmental Impact on Rivers (River Mechanics II), H.W. Shen, Fort Collins, CO (1973).

[VI–4] LEOPOLD, L.B., WOLMAN, M.G., MILLER, J.P., Fluvial Processes in Geomorphology, W.H. Froeman, San Francisco, CA (1964).

[VI–5]   NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Screening Models for Releases of Radionuclides to Atmosphere, Surface Water, and Ground, Rep. No. 123 I, NCRP, Bethesda, MD (1996).

[VI–6]   BOWIE, G.L., et al., Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling, 2nd edn, prepared for the US Environmental Protection Agency by Tetra Tech, Lafayette, CA, and Humbolt State University, Arcata, CA (1985).

[VI–7]   FISHER, H.G., LIST, E.J., KOH, R.C.Y., IMBERGER, J., BROOKS, N.H., Mixing in Inland and Coastal Waters, Academic Press, San Francisco (1979).

[VI–8]   HYDROSCIENCE, Simplified Mathematical Modeling of Water Quality, prepared for the United States Environmental Protection Agency, Office of Water Programs, Washington, DC (1997).

[VI–9]   WARD, P.R.B., Measurements of estuary dispersion coefficient, J. Environ. Eng. Div. **102** (1976) 855–859.

[VI–10]   O'CONNER, D.J., LAWLER, J.P., Mathematical Analysis of Estuarine Pollution, 55th Natl Mtg, Reprint No. 31a, American Institute of Chemical Engineers, Houston, TX (1965).

[VI–11]   OKUBO, A., Oceanic diffusion diagrams, Deep-Sea Res. **18** (1971) 789–802.

[VI–12]   EDINGER, J.E., BUCHAK, E.M., Offshore Nuclear Dispersion and Uptake Modeling, Appendix I: Coastal Advection and Dispersion for a Time Varying Continuous Source with Lateral Dispersion as a Function of Distance and with Radionuclide Decay, J.E. Edinger Associates, Wayne, IN (1977).

[VI–13]   UNITED STATES NUCLEAR REGULATORY COMMISSION, Liquid Pathway Generic Study, Impacts of Accidental Radionuclide Releases to the Hydrosphere from Floating and Land-Based Nuclear Power Plants, Rep. NUREG-0440, Washington, DC (1978).

Annex VII

METHODS USED IN THE ESTIMATION OF COLLECTIVE DOSES FOR SCREENING PURPOSES

VII–1. INTRODUCTION

As discussed in Section 7, collective doses are normally estimated for a particular location by summing the product of the number of individuals exposed and their average radiation dose. Models for estimating collective dose are normally relatively complex and are outside the scope of this report. However, estimates of collective dose per unit discharge have been made for use as part of the overall screening process, as described in Section 8. These are order of magnitude estimates only and should be used with caution. They are based on collective doses calculated in two different ways. The first uses complex models [VII–1], and the second is based on simple generic models developed by UNSCEAR [VII–2]. The source of the estimates and the choice of the order of magnitude screening values are discussed in this annex.

VII–2. THE MORE COMPLEX MODEL

A suite of mathematical models has been developed at the National Radiological Protection Board (NRPB) in the United Kingdom to estimate collective doses from releases of radionuclides to the environment [VII–1]. These models represent the transfer of radionuclides through the environment and the subsequent exposure of people. They cover dispersion over large distances and can be used to calculate doses to the population of the United Kingdom, Europe as a whole or the world. The models have been used to estimate collective doses from routine discharges of radionuclides from the United Kingdom's civil nuclear sites [VII–1]. As well as estimating the collective doses from actual discharges, the collective doses from unit discharges of particular radionuclides were also calculated. Both atmospheric and liquid releases were considered, and a range of values was obtained for the sites included in the study. It should be noted that not all radionuclides were considered for each site, only those radionuclides actually discharged at the site.

Tables VII–I, VII–II and VII–III show the range of collective doses per unit discharge given by the study for radionuclides released to the atmosphere and to marine and freshwater bodies, respectively. In some cases the radionuclide was discharged from only one site, so that only one value for the collective dose can be

given. These values are the collective effective dose commitment integrated to infinity. Collective doses truncated at 500 years are also available in Ref. [VII–1].

As seen in Table VII–I, the collective doses from atmospheric releases can have a range of more than an order of magnitude, depending on the radionuclide concerned and the location of the respective site. The highest collective doses are generally associated with sites located near areas of relatively high population density. For liquid releases the range in collective doses per unit discharge may be of several orders of magnitude depending on the radionuclides and sites concerned. The highest collective doses are generally associated with releases from inland sites discharging into rivers.

TABLE VII–I. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED TO THE ATMOSPHERE — DERIVATION OF VALUES FOR SCREENING PURPOSES

| Nuclide | Total (man·Sv/Bq) | Simple method | | | | More complex method range of results | Screening value (man·Sv/Bq) |
| | | Contribution (%) | | | | | |
		Inhalation	External (cloud)	External (deposit)	Ingestion		
Ac-228	1.1×10^{-12}	4	0	96	0		1×10^{-12}
Ag-110m	3.3×10^{-12}	1	0	90	9		1×10^{-12}
Am-241	1.0×10^{-10}	78	0	0	22	$5.7 \times 10^{-11} - 9.6 \times 10^{-10}$	1×10^{-9}
Ar-41						$9.6 \times 10^{-18} - 4.1 \times 10^{-16}$	1×10^{-16}
As-76	6.0×10^{-13}	0	0	100	0		1×10^{-12}
At-211	2.5×10^{-13}	81	0	18	1		1×10^{-13}
Au-198	4.7×10^{-13}	0	0	100	0		1×10^{-13}
Bi-206	3.5×10^{-12}	0	0	99	0		1×10^{-12}
Bi-210	2.2×10^{-13}	79	0	18	3		1×10^{-13}
Bi-212	1.5×10^{-12}	4	0	96	0		1×10^{-12}
Br-82	2.9×10^{-12}	0	0	100	0		1×10^{-12}
C-14						$8.6 \times 10^{-12} - 1.1 \times 10^{-11}$	1×10^{-11}
Cd-109	6.6×10^{-13}	2	0	4	94		1×10^{-12}
Ce-141	1.3×10^{-13}	5	0	62	32		1×10^{-13}
Ce-144	8.4×10^{-13}	12	0	24	64	$9.5 \times 10^{-14} - 2.3 \times 10^{-13}$	1×10^{-12}
Cm-242	1.1×10^{-11}	89	0	0	11		1×10^{-11}
Cm-244	6.4×10^{-11}	79	0	0	21		1×10^{-10}
Co-58	1.2×10^{-12}	0	0	90	9		1×10^{-12}
Co-60	3.7×10^{-12}	2	0	71	27	$2.3 \times 10^{-12} - 2.1 \times 10^{-11}$	1×10^{-11}
Cr-51	3.7×10^{-14}	0	0	93	7	7.5×10^{-15}	1×10^{-14}
Cs-134	5.6×10^{-12}	0	0	31	69	$2.7 \times 10^{-12} - 9.8 \times 10^{-12}$	1×10^{-11}
Cs-135	3.9×10^{-13}	0	0	0	100		1×10^{-13}
Cs-136	2.4×10^{-12}	0	0	96	4		1×10^{-12}

TABLE VII–I. (cont.)

Nuclide	Total (man·Sv/Bq)	Simple method				More complex method range of results	Screening value (man·Sv/Bq)
		Contribution (%)					
		Inhalation	External (cloud)	External (deposit)	Ingestion		
Cs-137	3.0×10^{-12}	0	0	21	79	$3.0 \times 10^{-12} – 2.4 \times 10^{-11}$	1×10^{-11}
Cu-64	2.1×10^{-13}	0	0	100	0		1×10^{-13}
Eu-154	1.7×10^{-12}	6	0	81	13	$1.6 \times 10^{-12} – 1.4 \times 10^{-11}$	1×10^{-11}
Eu-155	1.1×10^{-13}	11	0	58	31		1×10^{-13}
Fe-55	4.5×10^{-14}	3	0	0	97	$9.7 \times 10^{-15} – 1.8 \times 10^{-14}$	1×10^{-14}
Fe-59	1.4×10^{-12}	1	0	89	10	4.2×10^{-13}	1×10^{-12}
Ga-67	1.7×10^{-13}	0	0	100	0		1×10^{-13}
H-3						$1.3 \times 10^{-15} – 5.0 \times 10^{-15}$	1×10^{-15}
Hg-197	7.3×10^{-14}	1	0	99	0		1×10^{-13}
Hg-197m	9.8×10^{-14}	1	0	99	0		1×10^{-13}
Hg-203	4.1×10^{-13}	1	0	63	35		1×10^{-13}
I-123	1.9×10^{-13}	0	0	100	0		1×10^{-13}
I-125	2.1×10^{-12}	0	0	2	97		1×10^{-12}
I-129	2.2×10^{-11}	0	0	0	100	$8.0 \times 10^{-11} – 3.0 \times 10^{-10}$	1×10^{-10}
I-131	1.1×10^{-12}	1	0	37	62	$6.4 \times 10^{-14} – 2.1 \times 10^{-12}$	1×10^{-12}
I-132	2.5×10^{-12}	0	0	100	0		1×10^{-12}
I-133	7.2×10^{-13}	0	0	99	1		1×10^{-12}
I-134	2.9×10^{-12}	0	0	100	0		1×10^{-12}
I-135	1.7×10^{-12}	0	0	100	0		1×10^{-12}
In-111	4.4×10^{-13}	0	0	100	0		1×10^{-13}
In-113m	2.8×10^{-13}	0	0	100	0		1×10^{-13}
Kr-85						$2.4 \times 10^{-17} – 4.1 \times 10^{-17}$	1×10^{-17}
Kr-89						2.8×10^{-17}	1×10^{-17}
Mn-54	9.9×10^{-13}	0	0	91	8		1×10^{-12}
Mo-99	3.3×10^{-13}	1	0	99	1		1×10^{-13}
Na-22	9.4×10^{-12}	0	0	25	75		1×10^{-11}
Na-24	4.1×10^{-12}	0	0	100	0		1×10^{-12}
Nb-95	8.7×10^{-13}	0	0	95	4	$2.3 \times 10^{-14} – 6.6 \times 10^{-14}$	1×10^{-12}
Ni-59	2.4×10^{-13}	0	0	0	100		1×10^{-13}
Ni-63	5.4×10^{-13}	0	0	0	100		1×10^{-12}
Np-237	5.6×10^{-11}	77	0	0	23		1×10^{-10}
Np-239	1.8×10^{-13}	1	0	99	0		1×10^{-13}
P-32	3.1×10^{-13}	2	0	30	68	$6.4 \times 10^{-13} – 9.0 \times 10^{-13}$	1×10^{-12}
Pa-231	3.4×10^{-10}	76	0	0	24		1×10^{-10}
Pa-233	2.7×10^{-13}	3	0	80	17		1×10^{-13}
Pb-210	8.6×10^{-11}	2	0	0	98		1×10^{-10}
Pd-103	2.2×10^{-14}	4	0	65	31		1×10^{-14}
Pd-107	8.2×10^{-15}	13	0	0	87		1×10^{-14}
Pd-109	4.5×10^{-14}	2	0	98	0		1×10^{-14}
Pm-147	3.8×10^{-14}	25	0	0	75		1×10^{-14}
Po-210	1.3×10^{-10}	1	0	0	99		1×10^{-10}

TABLE VII–I. (cont.)

Nuclide	Total (man·Sv/Bq)	Simple method				More complex method range of results	Screening value (man·Sv/Bq)
		Contribution (%)					
		Inhalation	External (cloud)	External (deposit)	Ingestion		
Pu-238	1.1×10^{-10}	77	0	0	23	6.1×10^{-10}	1×10^{-9}
Pu-239	1.2×10^{-10}	77	0	0	23	$1.1 \times 10^{-10} - 6.6 \times 10^{-10}$	1×10^{-9}
Pu-240	1.2×10^{-10}	77	0	0	23		1×10^{-10}
Pu-241	2.2×10^{-12}	76	0	0	24	$1.9 \times 10^{-12} - 1.2 \times 10^{-11}$	1×10^{-11}
Pu-242	1.2×10^{-10}	77	0	0	23		1×10^{-10}
Ra-224	8.2×10^{-12}	78	0	20	2		1×10^{-11}
Ra-225	1.8×10^{-11}	80	0	2	19		1×10^{-11}
Ra-226	6.3×10^{-11}	28	0	3	69	$1.7 \times 10^{-10} - 2.8 \times 10^{-10}$	1×10^{-10}
Rb-86	1.1×10^{-12}	0	0	17	83		1×10^{-12}
Rh-105	8.7×10^{-14}	1	0	99	0		1×10^{-13}
Rh-107	3.9×10^{-13}	0	0	100	0		1×10^{-13}
Ru-103	5.8×10^{-13}	1	0	89	10		1×10^{-12}
Ru-106	1.4×10^{-12}	9	0	28	63	$1.6 \times 10^{-13} - 4.2 \times 10^{-13}$	1×10^{-12}
S-35	2.1×10^{-13}	1	0	0	99	$3.1 \times 10^{-13} - 2.0 \times 10^{-12}$	1×10^{-12}
Sb-124	2.2×10^{-12}	1	0	90	9		1×10^{-12}
Sb-125	6.2×10^{-13}	1	0	79	20	$2.6 \times 10^{-13} - 7.4 \times 10^{-13}$	1×10^{-12}
Se-75	7.7×10^{-13}	0	0	55	45	2.2×10^{-11}	1×10^{-11}
Sn-113	3.9×10^{-13}	1	0	79	19		1×10^{-13}
Sr-85	6.2×10^{-13}	0	0	91	9		1×10^{-12}
Sr-87m	3.5×10^{-13}	0	0	100	0		1×10^{-13}
Sr-89	3.3×10^{-13}	5	0	23	72		1×10^{-13}
Sr-90	1.1×10^{-11}	3	0	1	96	$3.2 \times 10^{-12} - 2.3 \times 10^{-11}$	1×10^{-11}
Tc-99	2.8×10^{-13}	0	0	0	100		1×10^{-13}
Tc-99m	1.4×10^{-13}	0	0	100	0		1×10^{-13}
Te-125m	1.5×10^{-13}	1	0	27	73		1×10^{-13}
Te-127m	4.0×10^{-13}	1	0	6	93		1×10^{-13}
Te-129m	3.6×10^{-13}	1	0	19	81		1×10^{-13}
Te-131m	1.7×10^{-12}	0	0	100	0		1×10^{-12}
Te-132	2.8×10^{-12}	0	0	99	1		1×10^{-12}
Th-228	1.4×10^{-10}	43	0	1	55		1×10^{-10}
Th-230	1.1×10^{-10}	76	0	2	22		1×10^{-10}
Th-232	1.1×10^{-10}	75	0	2	23	1.2×10^{-9}	1×10^{-9}
Tl-201	9.8×10^{-14}	0	0	100	0		1×10^{-13}
Tl-202	5.3×10^{-13}	0	0	97	3		1×10^{-12}
U-232	1.1×10^{-10}	62	0	1	36		1×10^{-10}
U-234	2.6×10^{-11}	68	0	8	24	$5.6 \times 10^{-11} - 5.6 \times 10^{-10}$	1×10^{-9}
U-235	2.2×10^{-11}	72	0	1	27	$7.5 \times 10^{-11} - 1.1 \times 10^{-10}$	1×10^{-10}
U-238	2.3×10^{-11}	66	0	9	25	$6.2 \times 10^{-11} - 8.8 \times 10^{-11}$	1×10^{-10}
Xe-133						$2.4 \times 10^{-17} - 1.1 \times 10^{-16}$	1×10^{-16}
Xe-135						$7.9 \times 10^{-18} - 1.6 \times 10^{-16}$	1×10^{-16}

TABLE VII–I. (cont.)

Nuclide	Total (man·Sv/Bq)	Simple method				More complex method range of results	Screening value (man·Sv/Bq)
		Contribution (%)					
		Inhalation	External (cloud)	External (deposit)	Ingestion		
Y-87	8.5×10^{-13}	0	0	100	0		1×10^{-12}
Y-90	1.3×10^{-13}	2	0	97	1		1×10^{-13}
Y-91	2.9×10^{-13}	6	0	28	66		1×10^{-13}
Zn-65	1.8×10^{-12}	0	0	34	65	$5.1 \times 10^{-12} - 2.0 \times 10^{-11}$	1×10^{-11}
Zr-95	1.7×10^{-12}	1	0	95	4	$8.2 \times 10^{-14} - 2.3 \times 10^{-13}$	1×10^{-12}

TABLE VII–II. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED INTO MARINE WATERS — DERIVATION OF VALUES FOR SCREENING PURPOSES

Nuclide	Total (man·Sv/Bq)	Simple method		More complex method range of results	Screening value (man·Sv/Bq)
		Contribution (%)			
		Fish ingestion	Shellfish ingestion		
Ac-228	2.4×10^{-18}	23	77		1×10^{-18}
Ag-110m	1.2×10^{-13}	23	77	$1.6 \times 10^{-13} - 2.6 \times 10^{-13}$	1×10^{-13}
Am-241	2.5×10^{-12}	1	99	$2.6 \times 10^{-14} - 6.2 \times 10^{-14}$	1×10^{-13}
As-76	2.4×10^{-16}	75	25		1×10^{-16}
At-211	6.2×10^{-18}	55	45		1×10^{-17}
Au-198	7.3×10^{-17}	38	63		1×10^{-16}
Bi-206	2.2×10^{-16}	11	89		1×10^{-16}
Bi-210	1.2×10^{-16}	11	89		1×10^{-16}
Bi-212	2.1×10^{-19}	11	89		1×10^{-19}
Br-82	3.8×10^{-19}	64	36		1×10^{-19}
C-14				$9.8 \times 10^{-12} - 1.1 \times 10^{-11}$	1×10^{-11}
Cd-109	2.5×10^{-13}	23	77		1×10^{-13}
Ce-141	9.5×10^{-17}	6	94	5.9×10^{-17}	1×10^{-16}
Ce-144	4.6×10^{-15}	6	94	$2.2 \times 10^{-16} - 5.4 \times 10^{-15}$	1×10^{-14}
Cm-242	3.9×10^{-14}	1	99	$1.9 \times 10^{-15} - 6.2 \times 10^{-14}$	1×10^{-13}
Cm-244	2.0×10^{-12}	1	99	$2.8 \times 10^{-14} - 1.4 \times 10^{-12}$	1×10^{-12}
Co-58	3.0×10^{-15}	55	45	$9.1 \times 10^{-16} - 4.9 \times 10^{-15}$	1×10^{-15}
Co-60	1.2×10^{-13}	55	45	$1.3 \times 10^{-14} - 7.3 \times 10^{-14}$	1×10^{-13}
Cr-51	3.7×10^{-17}	75	25	$7.3 \times 10^{-18} - 3.9 \times 10^{-17}$	1×10^{-17}

TABLE VII–II. (cont.)

| Nuclide | Simple method | | | More complex method range of results | Screening value (man·Sv/Bq) |
| | Total (man·Sv/Bq) | Contribution (%) | | | |
		Fish ingestion	Shellfish ingestion		
Cs-134	7.5×10^{-14}	95	5	$1.3 \times 10^{-14} - 5.2 \times 10^{-14}$	1×10^{-13}
Cs-135	1.6×10^{-14}	95	5		1×10^{-14}
Cs-136	4.0×10^{-16}	95	5		1×10^{-16}
Cs-137	9.6×10^{-14}	95	5	$1.3 \times 10^{-14} - 5.6 \times 10^{-14}$	1×10^{-13}
Cu-64	6.7×10^{-18}	68	32		1×10^{-17}
Eu-154	3.1×10^{-14}	20	80	$4.2 \times 10^{-15} - 3.1 \times 10^{-14}$	1×10^{-14}
Eu-155	7.8×10^{-16}	38	63	$3.5 \times 10^{-16} - 2.7 \times 10^{-15}$	1×10^{-15}
Fe-55	7.7×10^{-14}	38	63	$3.6 \times 10^{-15} - 5.0 \times 10^{-14}$	1×10^{-13}
Fe-59	4.1×10^{-14}	38	63	$9.9 \times 10^{-15} - 7.1 \times 10^{-14}$	1×10^{-13}
Ga-67	5.1×10^{-17}	86	14		1×10^{-16}
H-3				$3.2 \times 10^{-18} - 3.8 \times 10^{-18}$	1×10^{-18}
Hg-197	1.5×10^{-15}	86	14		1×10^{-15}
Hg-197m	1.1×10^{-15}	86	14		1×10^{-15}
Hg-203	2.0×10^{-13}	86	14		1×10^{-13}
I-123	1.4×10^{-19}	86	14		1×10^{-19}
I-125	9.9×10^{-16}	86	14	3.0×10^{-16}	1×10^{-16}
I-129	9.9×10^{-14}	86	14	$6.3 \times 10^{-15} - 4.7 \times 10^{-11}$	1×10^{-11}
I-131	2.1×10^{-16}	86	14		1×10^{-16}
I-132	3.3×10^{-20}	86	14		1×10^{-20}
I-133	4.4×10^{-18}	86	14		1×10^{-18}
I-134	4.8×10^{-21}	86	14		1×10^{-21}
I-135	3.0×10^{-19}	86	14		1×10^{-19}
In-111	2.2×10^{-16}	38	63		1×10^{-16}
In-113m	5.3×10^{-19}	38	63		1×10^{-18}
Mn-54	6.6×10^{-15}	32	68	$9.7 \times 10^{-16} - 6.7 \times 10^{-15}$	1×10^{-14}
Mo-99	4.5×10^{-18}	38	63		1×10^{-18}
Na-22	2.1×10^{-17}	67	33		1×10^{-17}
Na-24	4.1×10^{-21}	67	33		1×10^{-21}
Nb-95	3.9×10^{-16}	15	85	$1.2 \times 10^{-17} - 4.2 \times 10^{-16}$	1×10^{-16}
Ni-59	6.5×10^{-15}	75	25		1×10^{-14}
Ni-63	1.5×10^{-14}	75	25	$2.0 \times 10^{-16} - 3.8 \times 10^{-15}$	1×10^{-15}
Np-237	3.2×10^{-13}	13	87	$1.6 \times 10^{-12} - 1.8 \times 10^{-12}$	1×10^{-12}
Np-239	1.4×10^{-17}	13	87		1×10^{-17}
P-32	1.1×10^{-13}	90	10	$2.2 \times 10^{-14} - 1.5 \times 10^{-13}$	1×10^{-13}
Pa-231	7.3×10^{-12}	38	63		1×10^{-11}
Pa-233	3.1×10^{-16}	38	63		1×10^{-16}
Pb-210	1.8×10^{-11}	55	45		1×10^{-11}

TABLE VII–II. (cont.)

Nuclide	Simple method			More complex method range of results	Screening value (man·Sv/Bq)
	Total (man·Sv/Bq)	Contribution (%)			
		Fish ingestion	Shellfish ingestion		
Pd-103	1.1×10^{-16}	86	14		1×10^{-16}
Pd-107	1.0×10^{-15}	86	14		1×10^{-15}
Pd-109	1.1×10^{-17}	86	14		1×10^{-17}
Pm-147	7.1×10^{-16}	38	63	$1.0 \times 10^{-17} - 5.7 \times 10^{-16}$	1×10^{-15}
Po-210	1.5×10^{-10}	19	81		1×10^{-10}
Pu-238	4.7×10^{-12}	7	93	$1.2 \times 10^{-13} - 2.4 \times 10^{-12}$	1×10^{-12}
Pu-239	5.2×10^{-12}	7	93	$1.6 \times 10^{-13} - 3.8 \times 10^{-12}$	1×10^{-12}
Pu-240	5.2×10^{-12}	7	93		1×10^{-11}
Pu-241	8.8×10^{-14}	7	93	$2.6 \times 10^{-15} - 5.4 \times 10^{-14}$	1×10^{-13}
Pu-242	5.0×10^{-12}	7	93		1×10^{-12}
Ra-224	1.5×10^{-14}	75	25		1×10^{-14}
Ra-225	9.3×10^{-14}	75	25		1×10^{-13}
Ra-226	1.4×10^{-11}	75	25		1×10^{-11}
Rb-86	5.4×10^{-16}	97	3		1×10^{-15}
Rh-105	1.5×10^{-17}	38	63		1×10^{-17}
Rh-107	9.8×10^{-21}	38	63		1×10^{-20}
Ru-103	9.3×10^{-16}	1	99	$7.2 \times 10^{-17} - 6.0 \times 10^{-16}$	1×10^{-15}
Ru-106	5.9×10^{-14}	1	99	$2.9 \times 10^{-15} - 1.3 \times 10^{-13}$	1×10^{-13}
S-35	1.6×10^{-17}	75	25	$1.3 \times 10^{-18} - 3.4 \times 10^{-17}$	1×10^{-17}
Sb-124	6.6×10^{-15}	86	14	$2.0 \times 10^{-15} - 8.7 \times 10^{-15}$	1×10^{-14}
Sb-125	2.2×10^{-14}	86	14	$6.1 \times 10^{-15} - 3.1 \times 10^{-14}$	1×10^{-14}
Se-75	1.9×10^{-13}	86	14		1×10^{-13}
Sr-85	7.9×10^{-18}	86	14		1×10^{-17}
Sr-87m	8.2×10^{-22}	86	14		1×10^{-21}
Sr-89	2.9×10^{-17}	86	14	$8.8 \times 10^{-18} - 4.0 \times 10^{-17}$	1×10^{-17}
Sr-90	4.7×10^{-15}	86	14	$5.7 \times 10^{-16} - 2.7 \times 10^{-15}$	1×10^{-15}
Tc-99	9.7×10^{-15}	15	85	$1.7 \times 10^{-14} - 2.0 \times 10^{-14}$	1×10^{-14}
Tc-99m	1.1×10^{-19}	15	85		1×10^{-19}
Te-125m	5.6×10^{-15}	86	14	$1.0 \times 10^{-15} - 7.0 \times 10^{-15}$	1×10^{-14}
Te-127m	2.6×10^{-14}	86	14		1×10^{-14}
Te-129m	1.1×10^{-14}	86	14		1×10^{-14}
Te-131m	2.8×10^{-16}	86	14		1×10^{-16}
Te-132	1.5×10^{-15}	86	14		1×10^{-15}
Th-228	9.7×10^{-13}	78	22	6.3×10^{-15}	1×10^{-14}
Th-230	5.9×10^{-13}	78	22	3.3×10^{-14}	1×10^{-14}
Th-232	6.5×10^{-13}	78	22	9.3×10^{-15}	1×10^{-14}
Tl-201	1.7×10^{-16}	86	14		1×10^{-16}

TABLE VII–II. (cont.)

Nuclide	Simple method Total (man·Sv/Bq)	Contribution (%) Fish ingestion	Contribution (%) Shellfish ingestion	More complex method range of results	Screening value (man·Sv/Bq)
Tl-202	3.2×10^{-15}	86	14		1×10^{-15}
U-232	1.5×10^{-13}	17	83		1×10^{-13}
U-234	2.3×10^{-14}	17	83	$9.4 \times 10^{-14} - 1.2 \times 10^{-13}$	1×10^{-13}
U-235	2.2×10^{-14}	17	83	$9.2 \times 10^{-14} - 1.1 \times 10^{-13}$	1×10^{-13}
U-238	2.1×10^{-14}	17	83	$8.6 \times 10^{-14} - 1.1 \times 10^{-13}$	1×10^{-13}
Y-87	3.5×10^{-17}	11	89		1×10^{-17}
Y-90	1.4×10^{-16}	11	89	$4.6 \times 10^{-19} - 1.6 \times 10^{-17}$	1×10^{-17}
Y-91	2.5×10^{-15}	11	89	$1.1 \times 10^{-17} - 5.7 \times 10^{-17}$	1×10^{-16}
Zn-65	5.7×10^{-13}	11	89	$5.9 \times 10^{-14} - 6.3 \times 10^{-13}$	1×10^{-12}
Zr-95	4.4×10^{-16}	2	98	$6.4 \times 10^{-17} - 1.53 \times 10^{-15}$	1×10^{-15}

TABLE VII–III. COLLECTIVE EFFECTIVE DOSE COMMITMENTS PER UNIT ACTIVITY DISCHARGED TO FRESHWATER BODIES — DERIVATION OF VALUES FOR SCREENING PURPOSES

| Nuclide | Simple method | | | More complex method range of results | Screening value (man·Sv/Bq) |
| | Total (man·Sv/Bq) | Contribution (%) | | | |
		Fish ingestion	Drinking water		
Ac-228	8.0×10^{-18}	2	98		1×10^{-17}
Ag-110m	5.6×10^{-14}	1	99	7.25×10^{-13}	1×10^{-12}
Am-241	1.4×10^{-11}	5	95	$2.7 \times 10^{-12} – 2.3 \times 10^{-11}$	1×10^{-11}
As-76	3.5×10^{-16}	46	54		1×10^{-16}
At-211	3.6×10^{-16}	2	98		1×10^{-16}
Au-198	3.0×10^{-16}	6	94		1×10^{-16}
Bi-206	1.3×10^{-15}	2	98		1×10^{-15}
Bi-210	7.0×10^{-16}	2	98		1×10^{-15}
Bi-212	1.2×10^{-18}	2	98		1×10^{-18}
Br-82	1.4×10^{-16}	41	59		1×10^{-16}
C-14				7.5×10^{-13}	1×10^{-12}
Cd-109	5.5×10^{-14}	25	75		1×10^{-13}
Ce-141	1.7×10^{-15}	5	95		1×10^{-15}
Ce-144	8.0×10^{-14}	5	95		1×10^{-13}
Cm-242	1.4×10^{-13}	5	95	1.7×10^{-15}	1×10^{-13}
Cm-244	7.3×10^{-12}	5	95		1×10^{-11}
Co-58	6.2×10^{-15}	34	66	$2.5 \times 10^{-15} – 2.4 \times 10^{-13}$	1×10^{-13}
Co-60	2.4×10^{-13}	34	66	$3.8 \times 10^{-14} – 1.4 \times 10^{-12}$	1×10^{-12}
Cr-51	9.7×10^{-17}	25	75	7.6×10^{-17}	1×10^{-16}
Cs-134	1.3×10^{-11}	94	6	$1.7 \times 10^{-12} – 1.9 \times 10^{-11}$	1×10^{-11}
Cs-135	2.8×10^{-12}	94	6		1×10^{-12}
Cs-136	7.1×10^{-14}	94	6		1×10^{-13}
Cs-137	1.7×10^{-11}	94	6	$2.0 \times 10^{-12} – 1.5 \times 10^{-11}$	1×10^{-11}
Cu-64	9.0×10^{-18}	25	75		1×10^{-17}
Eu-154	1.4×10^{-13}	8	92	$1.0 \times 10^{-14} – 6.9 \times 10^{-13}$	1×10^{-12}
Eu-155	1.9×10^{-14}	8	92	$5.5 \times 10^{-16} – 1.1 \times 10^{-13}$	1×10^{-13}
Fe-55	1.6×10^{-14}	25	75	8.5×10^{-15}	1×10^{-14}
Fe-59	8.7×10^{-15}	25	75	2.3×10^{-14}	1×10^{-14}
Ga-67	1.1×10^{-16}	41	59		1×10^{-16}
H-3				$1.7 \times 10^{-15} – 2.4 \times 10^{-14}$	1×10^{-14}
Hg-197	1.7×10^{-16}	63	37		1×10^{-16}
Hg-197m	1.3×10^{-16}	63	37		1×10^{-16}
Hg-203	2.3×10^{-14}	63	37		1×10^{-14}
I-123	1.3×10^{-17}	6	94		1×10^{-17}
I-125	9.5×10^{-14}	6	94	1.52×10^{-11}	1×10^{-11}

TABLE VII–III. (cont.)

Nuclide	Simple method Total (man·Sv/Bq)	Contribution (%) Fish ingestion	Drinking water	More complex method range of results	Screening value (man·Sv/Bq)
I-129	9.5×10^{-12}	6	94		1×10^{-11}
I-131	2.0×10^{-14}	6	94	7.3×10^{-12}	1×10^{-11}
I-132	3.2×10^{-18}	6	94		1×10^{-18}
I-133	4.2×10^{-16}	6	94		1×10^{-16}
I-134	4.6×10^{-19}	6	94		1×10^{-19}
I-135	2.9×10^{-17}	6	94		1×10^{-17}
In-111	1.5×10^{-15}	94	6		1×10^{-15}
In-113m	3.6×10^{-18}	94	6		1×10^{-18}
Mn-54	2.7×10^{-14}	41	59	$2.9 \times 10^{-17} – 5.3 \times 10^{-15}$	1×10^{-14}
Mo-99	1.2×10^{-16}	2	98		1×10^{-16}
Na-22	1.4×10^{-13}	3	97		1×10^{-13}
Na-24	2.8×10^{-17}	3	97		1×10^{-17}
Nb-95	6.3×10^{-18}	34	66	1.3×10^{-13}	1×10^{-13}
Ni-59	4.0×10^{-15}	15	85		1×10^{-15}
Ni-63	9.3×10^{-15}	15	85	1.3×10^{-14}	1×10^{-14}
Np-237	4.7×10^{-12}	5	95		1×10^{-12}
Np-239	2.1×10^{-16}	5	95		1×10^{-16}
P-32	3.1×10^{-13}	99	1	$3.3 \times 10^{-13} – 2.3 \times 10^{-12}$	1×10^{-12}
Pa-231	4.7×10^{-11}	2	98		1×10^{-11}
Pa-233	2.0×10^{-15}	2	98		1×10^{-15}
Pb-210	5.2×10^{-11}	34	66		1×10^{-10}
Pd-103	2.3×10^{-16}	2	98		1×10^{-16}
Pd-107	2.0×10^{-15}	2	98		1×10^{-15}
Pd-109	2.2×10^{-17}	2	98		1×10^{-17}
Pm-147	9.9×10^{-15}	5	95		1×10^{-14}
Po-210	1.1×10^{-11}	8	92		1×10^{-11}
Pu-238	3.2×10^{-12}	5	95	3.3×10^{-13}	1×10^{-12}
Pu-239	3.5×10^{-12}	5	95	1.1×10^{-10}	1×10^{-12}
Pu-240	3.5×10^{-12}	5	95		1×10^{-12}
Pu-241	6.0×10^{-14}	5	95	9.0×10^{-14}	1×10^{-13}
Pu-242	3.4×10^{-12}	5	95		1×10^{-12}
Ra-224	2.7×10^{-14}	8	92		1×10^{-14}
Ra-225	1.6×10^{-13}	8	92		1×10^{-13}
Ra-226	2.4×10^{-11}	8	92	$3.7 \times 10^{-10} – 4.0 \times 10^{-10}$	1×10^{-10}
Rb-86	2.3×10^{-14}	77	23		1×10^{-14}
Rh-105	4.7×10^{-17}	2	98		1×10^{-17}
Rh-107	3.1×10^{-20}	2	98		1×10^{-20}

TABLE VII–III. (cont.)

Nuclide	Simple method			More complex method range of results	Screening value (man·Sv/Bq)
	Total (man·Sv/Bq)	Contribution (%)			
		Fish ingestion	Drinking water		
Ru-103	2.9×10^{-15}	2	98	4.8×10^{-13}	1×10^{-13}
Ru-106	1.8×10^{-13}	2	98	$1.2 \times 10^{-14} - 6.3 \times 10^{-12}$	1×10^{-11}
S-35	1.3×10^{-14}	58	42	$9.9 \times 10^{-14} - 8.5 \times 10^{-13}$	1×10^{-12}
Sb-124	1.7×10^{-14}	15	85	4.9×10^{-15}	1×10^{-14}
Sb-125	5.9×10^{-14}	15	85	$1.6 \times 10^{-14} - 1.6 \times 10^{-12}$	1×10^{-12}
Se-75	3.8×10^{-14}	25	75		1×10^{-14}
Sr-85	3.8×10^{-15}	11	89		1×10^{-15}
Sr-87m	4.0×10^{-19}	11	89		1×10^{-19}
Sr-89	1.4×10^{-14}	11	89	$1.6 \times 10^{-13} - 2.8 \times 10^{-12}$	1×10^{-12}
Sr-90	2.3×10^{-12}	11	89	$7.5 \times 10^{-13} - 3.5 \times 10^{-11}$	1×10^{-11}
Tc-99	5.4×10^{-14}	3	97		1×10^{-13}
Tc-99m	6.1×10^{-19}	3	97		1×10^{-18}
Te-125m	8.4×10^{-15}	41	59		1×10^{-14}
Te-127m	3.9×10^{-14}	41	59		1×10^{-14}
Te-129m	1.7×10^{-14}	41	59		1×10^{-14}
Te-131m	4.2×10^{-16}	41	59		1×10^{-16}
Te-132	2.2×10^{-15}	41	59		1×10^{-15}
Th-228	2.2×10^{-11}	15	85		1×10^{-11}
Th-230	1.3×10^{-11}	15	85		1×10^{-11}
Th-232	1.5×10^{-11}	15	85		1×10^{-11}
Tl-201	5.5×10^{-17}	63	37		1×10^{-16}
Tl-202	1.0×10^{-15}	63	37		1×10^{-15}
U-232	2.6×10^{-11}	2	98		1×10^{-11}
U-234	4.0×10^{-12}	2	98	1.42×10^{-10}	1×10^{-10}
U-235	3.9×10^{-12}	2	98		1×10^{-12}
U-238	3.7×10^{-12}	2	98	7.7×10^{-11}	1×10^{-10}
Y-87	1.7×10^{-16}	5	95		1×10^{-16}
Y-90	6.7×10^{-16}	5	95	6.2×10^{-17}	1×10^{-15}
Y-91	1.2×10^{-14}	5	95		1×10^{-14}
Zn-65	2.0×10^{-13}	63	37	$6.3 \times 10^{-13} - 5.0 \times 10^{-12}$	1×10^{-12}
Zr-95	8.6×10^{-15}	34	66	2.1×10^{-14}	1×10^{-14}

VII–3. SIMPLE GENERIC MODEL

UNSCEAR has developed a set of simple generic models that can be used to estimate collective doses for the transfer of radionuclides in the environment without using complex models [VII–2]. Generic global parameter values can be used with these models if site specific data are not available.

For atmospheric releases it is assumed that all the activity released will be deposited and that the collective dose is independent of the distribution pattern of the deposited material. Three exposure pathways were considered here: inhalation, ingestion of terrestrial foods and external radiation from deposited material. A population density is required in this calculation of collective dose; for the purposes of calculating screening values a population density of 35 people per km^2 was used [VII–3]. This represents a global average; considerably higher densities can be found in some countries, while lower densities can be found in others. Global average yields of particular foods were also required and were obtained from an FAO compilation [VII–3]. The foods considered were grain, green vegetables and fruit, root vegetables, milk and meat.

Table VII–I gives values of the collective doses for unit releases of various radionuclides to the atmosphere estimated using a simplified method based on the UNSCEAR approach. These are the collective effective dose commitment values calculated using the effective dose coefficients given in Section 6 of this report. In most cases the values are of the same order as, or in the range of, the collective doses obtained from the more complex models, and are also presented in Table VII–I.

For releases of radionuclides in liquid form their dispersion and subsequent transfer to humans will vary considerably depending on the characteristics of the receiving water body. In the general model developed by UNSCEAR [VII–2], simple dilution is assumed. Account is taken of the transfer of radionuclides to sediments through the use of the sediment distribution coefficient K_d. The exposure pathways considered are the consumption of drinking water and aquatic foods; in this application fish and crustacea were considered. It is assumed that the number of people consuming the food divided by the volume of the receiving waters is relatively constant, and generic values of 4.3×10^{-9} and 4.6×10^{-8} people/L for marine and fresh waters, respectively [VII–2], are used. The collective doses obtained using this method are given in Tables VII–II and VII–III and were obtained using the effective dose coefficients given in Section 6. In some cases there are significant differences between the generic results and those obtained using the more complex model. In these cases generally the more complex model was applied to radionuclides discharged into coastal waters, where a much lower value of the ratio between the number of people consuming seafood and the volume of the receiving waters applies.

VII–4. CHOICE OF SCREENING VALUES

The collective doses given in Tables VII–I to VII–III were used to obtain order of magnitude estimates of collective doses per unit discharge to be used for screening purposes. These estimates are also presented in the tables. In many cases the generic value and the range obtained from the more complex method were of the same order of magnitude, making the choice of screening value straightforward. However, in other cases there was a considerable range in the values obtained; in these cases an order of magnitude at the top of the range was chosen. In addition, where different radionuclides were expected to have collective doses of the same order of magnitude, for example ^{238}Pu and ^{239}Pu, the order of magnitude estimates were made to be the same even if the range available did not indicate this. Because of the methods used to determine the order of magnitude estimates, these should be used with extreme caution and only for screening purposes.

REFERENCES

[VII–1] BEXON, A.P., Radiological Impact of Routine Discharges from UK Civil Nuclear Sites in the Mid 1990s, Rep. NRPB-R312, National Radiological Protection Board, Chilton (1999).

[VII–2] UNITED NATIONS, Dose Assessment Methodologies, Draft R602, Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), UN, New York (1999).

[VII–3] FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, FAO Production Yearbook 1980, Vol. 34, FAO, Rome (1981).

SYMBOLS FOR PARAMETERS USED IN THIS REPORT

SECTION 3

Symbol	Unit	Description
A_B	m^2	Projected cross-sectional area of the building most influencing the flow of the plume.
B	m^{-2}	Gaussian diffusion factor corrected for dispersion in the lee of a building inside the wake zone (use instead of F).
B_0	–	Unitless constant that accounts for the potential increase in air concentration along a vertical wall.
C_A	Bq/m^3	Annual average concentration of radionuclide in air (at ground level).
$\dot{d}_i$	Bq·m^{-2}·d^{-1}	Daily average deposition rate to the ground surface of radionuclide i from both dry and wet processes.
E, G	—	Fit parameters for the equation that defines σ_z.
f	—	Unitless reduction factor accounting for radioactive decay during atmospheric transport.
F	m^{-2}	Gaussian diffusion factor appropriate for the height of release H and downwind distance x.
K	m	Constant with a value of 1 m.
H	m	Height at which the radionuclide is released.
H_B	m	Height of the building that dominates the air flow near the radionuclide release point.
P_p	—	Fraction of time during the year that the wind blows towards the receptor of interest in sector p.
Q_i	Bq/s	Annual average rate of radionuclide emission to the atmosphere.
u_a	m/s	Annual geometric mean of the wind speed at the height of the release.
V	m^3/s	Volumetric flow rate of vent or stack at the point of release.

Symbol	Unit	Description
V_d	m/d	Dry deposition coefficient (deposition on to soil and vegetation).
V_w	m/d	Wet deposition coefficient.
V_T	m/d	Total (dry and wet) deposition coefficient.
x	m	Location of the receptor (downwind distance).
λ_i	s^{-1}	Radioactive decay constant for radionuclide i.
σ_z	m	Vertical diffusion parameter.

SECTION 4

Symbol	Unit	Description
B	m	River or estuary width.
$\bar{B}$	m	River width under a mean annual river flow rate upstream of the tidal flow area.
C_0	Bq/m^3	Radionuclide concentration in water at the effluent discharge outfall.
$C_{s,b}$	Bq/kg	Radionuclide concentration in bottom sediment.
$C_{s,s}$	Bq/m^2	Radionuclide concentration in shore or beach sediment.
$C_{s,w}$	Bq/kg	Radionuclide concentration in suspended sediment.
C_{te}	Bq/m^3	Maximum radionuclide concentration on the river bank opposite to the discharge point.
C_{sludge}	Bq/kg	Annual average radionuclide concentration in sewage sludge.
$C_{w,s}$	Bq/m^3	Radionuclide concentration in filtered water.
$C_{w,\,tot}$	Bq/m^3	Total radionuclide concentration in water.
D	m	Flow depth that corresponds to q_r.
F	m^3/s	Flow rate of the liquid effluent.
K_d	L/kg	Distribution coefficient between the radionuclide on sediment and that dissolved in water.

P_r	—	Correction factor for partial mixing.
q_r	m^3/s	30 year low annual river flow rate.
q_r	m^3/s	Mean annual river flow rate.
Q_i	Bq/s	Average radionuclide discharge rate into river, estuary, lake or coastal waters.
Q_{sludge}	Bq/a	Annual discharge of a radionuclide to a sewage plant.
S_{sludge}	kg/a	Annual sewage sludge production at the relevant sewage treatment works.
S_s	kg/m^3 or g/L	Suspended sediment concentration.
T_e	s	Effective accumulation time for shore or beach sediment.
T_p	s	Tidal period.
U	m/s	Net freshwater velocity.
x	m	Longitudinal distance from the release point to a potential receptor location.
y_0	m	Distance between the release point and the beach.
λ_i	s^{-1}	Radionuclide decay constant for radionuclide i.

SECTION 5

Symbol	Unit	Description
B_p	L/kg	Equilibrium ratio of the concentration of a radionuclide in aquatic food p to its dissolved concentration in water, known as the bioaccumulation factor.
B_v	Bq/kg$_{\text{plant tissue}}$ per Bq/kg$_{\text{dry soil}}$	Concentration factor for the uptake of radionuclide from soil by edible parts of crops; adhesion of soil to the vegetation is not taken into account. For pasture grass the unit of mass is dry matter; for vegetables consumed by humans it is fresh weight.
C_A	Bq/m$^3_{\text{air}}$	Annual average concentration of radionuclide in air.
$C_{a,i}$	Bq/kg$_{\text{dry weight}}$	Annual average concentration of radionuclide in animal feed.

$C_{af,i}$	Bq/kg	Annual average concentration of radionuclide in aquatic foods.
$C_{f,i}$	$\text{Bq/kg}_{\text{meat}}$	Annual average concentration of radionuclide in meat.
$C_{m,i}$	$\text{Bq/L}_{\text{milk}}$	Annual average concentration of radionuclide in milk.
$C_{p,i}$	$\text{Bq/kg}_{\text{dry matter}}$	Annual average concentration of radionuclide in stored animal feed.
C_s	Bq/kg	Annual average concentration of radionuclide in dry soil.
$C_{v,i}$	Bq/kg	Annual average concentration of radionuclide in food crops (expressed in fresh matter for vegetation consumed by humans and in dry matter for vegetation consumed by animals).
$C_{w,i}$	$\text{Bq/L}_{\text{water}}$	Annual average concentration in water.
$\dot{d}_i$	$\text{Bq·m}^{-2}\text{·d}^{-1}$	Annual average deposition rate from wet and dry processes to the ground surface.
F_f	$\text{d/kg}_{\text{meat}}$	Fraction of the animal daily intake that appears in a kilogram of flesh at the time of slaughter.
F_m	d/L_{milk}	Fraction of animal daily intake that appears in a litre of milk at the time of milking.
f_p	—	Fraction of time during a year that an animal consumes fresh pasture.
F_v	$\text{Bq/kg}_{\text{plant tissue}}$ per $\text{Bq/kg}_{\text{dry soil}}$	Concentration factor for the uptake of radionuclide from soil by edible parts of crops, adjusted implicitly to account for adhesion of soil to the vegetation. For pasture grass the unit of mass is dry matter; for vegetables consumed by humans it is fresh weight.
I_w	$\text{m}^3_{\text{water}}\text{·m}^{-2}\text{·d}^{-1}$	Average irrigation rate.
Q_f	$\text{kg}_{\text{dry matter}}/\text{d}$	Amount of feed consumed per day by a meat producing animal.
Q_m	$\text{kg}_{\text{dry matter}}/\text{d}$	Amount of feed consumed per day by a milk producing animal.
Q_w	m^3/d	Amount of water consumed per day by an animal.
t_b	d	Duration of discharge of radioactive material.

t_e	d	Time period that crops are exposed to contamination during the growing season.
t_f	d	Time interval between slaughter and consumption of the meat.
t_h	d	Delay (holdup) time that represents the time interval between harvest and consumption of the food.
t_m	d	Time interval between milking and consumption of the milk.
α	m²/kg	Mass interception fraction for a given food crop.
$\lambda_{E_i^s}$	d⁻¹	Effective rate constant responsible for reductio radionuclide concentration in the root zo
$\lambda_{E_i^v}$	d⁻¹	Effective rate constant responsible for reduction of the radionuclide concentration on vegetation.
λ_i	d⁻¹	Radioactive decay constant of a radionuclide.
λ_w	d⁻¹	Effective rate constant responsible for reduction of the concentration of material deposited on plant surfaces due to processes other than radioactive decay.
λ_s	d⁻¹	Effective rate constant responsible for reduction of the concentration of material deposited in the root zone of soils due to processes other than radioactive decay.
ρ	kg/m²$_{\text{dry soil}}$	Effective surface soil density for the root zone.

SECTION 6

Symbol	Unit	Description
C_A	Bq/m³	Annual average concentration in air.
C_{gr}	Bq/m²	Ground surface concentration resulting from a 30 year discharge to the atmosphere.
$C_{p,i}$	Bq/kg	Concentration of radionuclide i in foodstuff p at the time of consumption.
$C_{s,s}$	Bq/m²	Annual average sediment surface concentration.

C_{sludge}	Bq/kg	Mass concentration deposited in sewage sludge.
C_{sludge}	Bq/m^2	Surface concentration deposited in sewage sludge.
DF_s	Sv/a per Bq/m^3	Dose coefficient for skin from air immersion.
DF_{gr}	Sv/a per Bq/m^2	Dose coefficient for exposure to contaminated ground or sediment surfaces.
DF_{im}	Sv/a per Bq/m^3	Immersion dose coefficient.
DF_{ing}	Sv/Bq	Dose coefficient for ingestion of radionuclides.
DF_{inh}	Sv/Bq	Dose coefficient for inhalation of radionuclides.
DL	kg/m^3	Dust loading factor.
E_{gr}	Sv/a	External dose rate from ground deposits after a 30 year discharge to the atmosphere.
E_{inh}	Sv/a	Annual inhalation dose from the concentration in air.
E_{im}	Sv/a	External dose rate from immersion in a discharge cloud.
$E_{im,\,s}$	Sv/a	External dose rate to skin from immersion in a discharge cloud.
$E_{ing,\,p}$	Sv/a	Annual dose from ingestion of foodstuff p.
E_m	Sv/a	External dose rate from exposure to contaminated sediment.
E_{res}	Sv/a	Annual inhalation dose from resuspension of radionuclides deposited in sewage sludge.
E_s	Sv/a	External dose rate from concentration in sewage sludge.
H_p	kg/a	Annual average consumption rate for foodstuff p.
O_f	—	Fraction of time in a year when an individual is exposed to a particular exposure pathway.
R_{inh}	m^3/a	Annual average inhalation rate.
ρ	kg/m^3	Density (in Section 6 the density of sewage sludge is referred to).

GLOSSARY

critical group. A group of **members of the public** which is reasonably homogeneous with respect to its **exposure** for a given radiation source and given **exposure pathway** and is typical of individuals receiving the highest **effective dose** or **equivalent dose** (as applicable) from the given **source**.

decay constant, λ. For a radionuclide in a particular energy state the quotient of dP by dt, where dP is the probability of a given nucleus undergoing a spontaneous nuclear transition from that energy state in the time interval dt.

$$\lambda = \frac{dP}{dt} = -\frac{1}{N}\frac{dN}{dt} = \frac{A}{N}$$

discharge. Planned and controlled release of (usually gaseous or liquid) radioactive material to the environment.

  authorized discharge: discharge in accordance with an authorization.

dispersion. The spreading of radionuclides in air (aerodynamic dispersion) or water (hydrodynamic dispersion) resulting mainly from physical processes affecting the velocity of different molecules in the medium.

dose.
  A measure of the energy deposited by radiation in a target.
  - For definitions of the most important such measures, see **dose quantities** and **dose concepts**.

dose concepts.

  annual dose: the **dose** due to **external exposure** in a year plus the **committed dose** from **intakes** of radionuclides in that year.

  collective dose: the total radiation dose incurred by a population.
  - This is the sum of all of the **individual doses** to members of this population. If the **doses** continue for longer than a year, then the annual **individual doses** must also be integrated over time. Unless otherwise specified, the time over which the **dose** is integrated is infinite; if a finite upper limit is applied to the time integration, the collective dose is described as truncated at that time.

  committed dose: the lifetime dose expected to result from an **intake**.

individual dose: the **dose** incurred by an individual.
- Contrast with **collective dose**.

dose constraint.

A prospective and **source** related restriction on the **individual dose** delivered by the **source**, which serves as a bound in the **optimization** of protection and safety of the **source**. For **public exposure** the dose constraint is an upper bound on the **annual doses** that **members of the public** should receive from the planned operation of any controlled **source**. The **exposure** to which the dose constraint applies is the **annual dose** to any critical group, summed over all exposure pathways, arising from the predicted operation of the controlled **source**. The **dose constraint** for each **source** is intended to ensure that the sum of **doses** to the critical group from all controlled **sources** remains within the dose limit.

dose quantities.

absorbed dose, D: the fundamental dosimetric quantity D, defined as

$$D = \frac{d\bar{\varepsilon}}{dm}$$

where $d\bar{\varepsilon}$ is the mean energy imparted by ionizing radiation to matter in a volume element and dm is the mass of matter in the volume element [1].
- The energy can be averaged over any defined volume, the average **dose** being equal to the total energy imparted in the volume divided by the mass in the volume.
- Absorbed dose is defined at a point; for the average dose in a tissue or organ see **organ dose.**
- Unit: J/kg, termed the gray (Gy) (formerly the rad was used).

collective effective dose, S: the total **effective dose** S to a population, defined as

$$S = \sum_i E_i N_i$$

where E_i is the average **effective dose** in the population subgroup i and N_i is the number of individuals in the subgroup. It can also be defined by the integral

$$S = \int_0^\infty E \frac{dN}{dE} dE$$

where $\frac{dN}{dE} dE$ is the number of individuals receiving an **effective dose** between E and $E + dE$.[1]

The **collective effective dose** S_k committed by an event, a decision or a finite portion of a **practice** k is given by

$$S_k = \int_0 \dot{S}_k(t)dt$$

where $\dot{S}_k$ is the collective effective dose rate at time t caused by k [1].

committed effective dose, $E(\tau)$: the quantity $E(\tau)$, defined as

$$E(\tau) = \sum_T w_T H_T(\tau)$$

where $H_T(\tau)$ is the **committed equivalent dose** to tissue T over the integration time t and w_T is the tissue weighting factor for tissue T. When τ is not specified, it will be taken to be 50 years for adults and to age 70 years for **intakes** by children [1].

committed equivalent dose, $H_T(\tau)$: the quantity $H_T(\tau)$, defined as

$$H_T(\tau) = \int_{t_0}^{t_0+\tau} \dot{H}_T(t)dt$$

where t_0 is the time of intake, $\dot{H}_T(t)$ is the **equivalent dose rate** at time t in organ or tissue T and τ is the time elapsed after an **intake** of radioactive substances. When τ is not specified, it will be taken to be 50 years for adults and to age 70 years for **intakes** by children [1].

effective dose, E: the quantity E, defined as a summation of the tissue **equivalent doses**, each multiplied by the appropriate tissue weighting factor

---

[1] Although the upper limit for the integral may in principle be infinite, most assessments of collective dose would address separately the component associated with individual dose rates higher than the thresholds for induction of deterministic effects.

$$E = \sum_T W_T H_T$$

where H_T is the **equivalent dose** in tissue T and w_T is the tissue weighting factor for tissue T. From the definition of **equivalent dose**, it follows that

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

where w_R is the radiation weighting factor for radiation R and $D_{T,R}$ is the average **absorbed dose** in organ or tissue T [1].

- The unit of effective dose is J/kg, termed the sievert (Sv). The rem, equal to 0.01 Sv, is sometimes used as a unit of **equivalent dose** and effective dose. This should not be used in IAEA documents except when quoting directly from other documents, in which case the value in sieverts should be added in parentheses.
- Effective dose is a measure of **dose** designed to reflect the amount of radiation detriment likely to result from the **dose**.
- Values of effective dose from any type(s) of radiation and mode(s) of exposure can be compared directly.

equivalent dose, H_T: the quantity $H_{T,R}$, defined as

$$H_{T,R} = w_R D_{T,R}$$

where $D_{T,R}$ is the **absorbed dose** delivered by radiation type R averaged over a tissue or organ T and w_R is the radiation weighting factor for radiation type R. When the radiation field is composed of different radiation types with different values of w_R, the equivalent dose is

$$H_T = \sum_R w_R D_{T,R} \ [1]$$

- The unit of equivalent dose is J/kg, termed the sievert (Sv). The rem, equal to 0.01 Sv, is sometimes used as a unit of equivalent dose and **effective dose**. This should not be used in IAEA documents except when quoting directly from other documents, in which case the value in sieverts should be added in parentheses.
- A measure of the **dose** to a tissue or organ designed to reflect the amount of

harm caused.

- Values of equivalent dose to a specified tissue from any type(s) of radiation can therefore be compared directly.

organ dose: the mean **absorbed dose** D_T in a specified tissue or organ T of the human body, given by

$$D_T = \frac{1}{m_T} \int_{m_T} D \, dm$$

where m_T is the mass of the tissue or organ and D is the **absorbed dose** in mass element dm.

exemption. The determination by a regulatory body that a **source** or **practice** need not be subject to some or all aspects of regulatory control on the basis that the **exposure** (including potential exposure) due to the **source** or **practice** is too small to warrant the application of those aspects.

exposure. The act or condition of being subject to irradiation.

external exposure: exposure due to a **source** outside the body.

internal exposure: exposure due to a **source** within the body.

exposure, types of.

medical exposure: exposure incurred by patients as part of their own medical or dental diagnosis (**diagnostic exposure**) or treatment (**therapeutic exposure**); by persons, other than those occupationally exposed, knowingly exposed while voluntarily helping in the support and comfort of patients; and by volunteers in a programme of biomedical research involving their **exposure**.

occupational exposure: all **exposure** of workers incurred in the course of their work, with the exception of excluded exposures and **exposures** from exempt **practices** or exempt **sources**.

public exposure: exposure incurred by **members of the public** from radiation sources, excluding any **occupational** or **medical exposure** and the normal local natural background radiation but including **exposure** from authorized **sources** and **practices** and from intervention situations [1].

exposure pathway. A route by which radiation or radionuclides can reach humans and cause **exposure**.

- An exposure pathway may be very simple, for example **external exposure** from airborne radionuclides, or a more complex chain, for example **internal exposure** from drinking milk from cows that ate grass contaminated with deposited radionuclides.

half-life, $T_{1/2}$.

1. For a radionuclide the time required for the activity to decrease by half by a radioactive decay process.
   - Where it is necessary to distinguish this from other half-lives (see (**2**)), the term radioactive half-life should be used.
   - The half-life is related to the decay constant λ by the expression

$$T_{1/2} = \frac{\ln 2}{\lambda}$$

2. The time taken for the quantity of a specified material (e.g. a radionuclide) in a specified place to decrease by half as a result of any specified process or processes that follow similar exponential patterns to radioactive decay.

   effective half-life, T_{eff}: the time taken for the activity of a radionuclide in a specified place to halve as a result of all relevant processes.

$$T_{\text{eff}} = \frac{\prod_i T_i}{\sum_i T_i} \left(or \ \frac{1}{T_{\text{eff}}} = \sum_i \frac{1}{T_i} \right)$$

   where T_i is the half-life for process i.

   radioactive half-life: for a radionuclide the time required for the activity to decrease, by a radioactive decay process, by half.
   - The term physical half-life is also used for this concept.

intake. The act or process of taking radionuclides into the body by inhalation or ingestion or through the skin.

level.

   reference level: an action level, intervention level, investigation level or recording level.

limit. The value of a quantity used in certain specified activities or circumstances that must not be exceeded.

member of the public. In a general sense any individual in the population except, for protection and safety purposes, when subject to **occupational** or **medical exposure**. For the purpose of verifying compliance with the annual dose limit for **public exposure** the representative individual in the relevant **critical group**.

monitoring. The measurement of **dose** or contamination for reasons related to the assessment or control of **exposure** to radiation or radioactive substances, and the interpretation of the results.

naturally occurring radionuclides. Radionuclides that occur naturally in significant quantities on Earth.
- The term is usually used to refer to the primordial radionuclides ^{40}K, ^{235}U, ^{238}U and ^{232}Th (the decay product of primordial ^{236}U) and their radioactive decay products, but may also include ^{3}H and ^{14}C, low concentrations of which are generated by natural activation processes.

optimization. The process of determining what level of protection and safety makes **exposures**, and the probability and magnitude of potential **exposures**, 'as low as reasonably achievable, economic and social factors being taken into account' (ALARA), as required by the ICRP System of Radiological Protection.

practice. Any human activity that introduces additional **sources** of **exposure** or **exposure pathways** or extends **exposure** to additional people or modifies the network of **exposure pathways** from existing **sources**, so as to increase the **exposure** or the likelihood of **exposure** of people or the number of people exposed.

source. Anything that may cause radiation exposure — for example by emitting ionizing radiation or by releasing radioactive substances or materials — and can be treated as a single entity for protection and safety purposes.

natural source: a naturally occurring source of radiation, such as the sun and stars (sources of cosmic radiation) or rocks and soil (terrestrial sources of radiation).

CONTRIBUTORS TO DRAFTING AND REVIEW

Avetisov, G.	Institute of Biophysics, Russian Federation
Barry, P.	Atomic Energy of Canada Limited, Canada
Beninson, D.	Comisión Nacional de Energía Atómica, Argentina
Bergström, U.	Studsvik Eco and Safety AB, Sweden
Blaylock, G.	SENES Oak Ridge, Inc., United States of America
Boge, R.	National Institute for Radiation Protection, Sweden
Brenk, H.D.	Brenk Systemplanung, Germany
Bucina, I.	Institute of Public Health, Czech Republic
Casimir, R.	CEA/Institut des protection et de sûreté nucléaire, France
Cooper, J.	National Radiological Protection Board, United Kingdom
Crick, M.	International Atomic Energy Agency
Edvarson, K.	National Institute for Radiation Protection, Sweden
Fayers, C.	National Radiological Protection Board, United Kingdom
Frissel, M.	Netherlands
Gerzabek, M.	Austrian Research Centre, Austria
Govaerts, P.	CEN/SCK, Belgium
Grupa, J.	ECN, Netherlands
Haywood, S.	National Radiological Protection Board, United Kingdom
Hoffman, F.O.	SENES Oak Ridge, Inc., United States of America
Homma, T.	Tokai Research Establishment, Japan
Kanyár, B.	National Research Institute for Radiobiology and Radiohygiene, Hungary
Laitano, F.	Ente per le Nuove Tecnologie, l'Energie e l'Ambiente, Italy
Lewyckyj, N.	CEN/SCK, Belgium
Linsley, G.	International Atomic Energy Agency
Malátová, I.	Institute of Public Health, Czech Republic
Miller, C.	Centers for Disease Control and Prevention, United States of America

Monte, L.	Ente per le Nuove Tecnologie, l'Energie e l'Ambiente, Italy
Onishi, Y.	Pacific Northwest National Laboratory, United States of America
Parker, T.	British Nuclear Fuels plc, United Kingdom
Pietrazak-Flis, Z.	Central Laboratory for Radiological Protection, Poland
Pröhl, G.	Institut für Strahlenschutz, Germany
Robinson, C.	International Atomic Energy Agency
Rowan, D.	Atomic Energy of Canada Ltd, Canada
Santucci, P.	CEA/Institut des protection et de sûreté nucléaire, France
Simmonds, J.	National Radiological Protection Board, United Kingdom
Sjoeblom, K.-L.	International Atomic Energy Agency
Subbaratnam, T.	Bhabha Atomic Research Centre, India
Thiessen, K.	SENES Oak Ridge, Inc., United States of America
Torres, C.	CIEMAT, Spain
Tranteeva, M.	Institute of Nuclear Research and Nuclear Energy, Bulgaria
Tscherlovits, M.	Atominstitut der Österreichischen Universitäten, Austria
Vidláková, J.	Institute of Public Health, Czech Republic

Consultants Meetings

Vienna, Austria: 16–20 March 1987, 3–7 December 1990, 31 January – 4 February 1994
Sequim, Washington, United States of America: 21–25 August 1995
Vienna, Austria: 7–11 June, 1–5 November 1999

Advisory Group Meetings

Vienna, Austria: 1–5 June 1987, 14–18 November 1988, 17–21 October 1994