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# INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION

# COMMITTEE 2

## Supporting Guidance Document
## Interpretation of Bioassay Data

## MAIN TEXT

**Prepared by INDOS and DOCAL Task Groups**

**DRAFT DOCUMENT**

# ICRP SUPPORTING GUIDANCE DOCUMENT
# INTERPRETATION OF BIOASSAY DATA

## CONTENTS

*(cont'd overleaf)*

## Preface

Committee 2 of the International Commission on Radiological Protection (ICRP) has the responsibility for providing dose coefficients for intakes of radionuclides by workers who are occupationally exposed. It also gives information that can be used for assessing intakes and doses from monitoring data. The interpretation of bioassay measurements is not straightforward. The biokinetic and dosimetric models developed and published by ICRP are `almost universally used to calculate intakes and doses from bioassay measurements. However, there are several assumptions that have to be made in terms of the exposure scenario, including pattern and mode of intake, physical and chemical characteristics of the compound and the time delay between the exposure(s) and measurement. Health physicists also have to infer which bioassay results to use and to decide upon the detection capability, quality of the measurements and the choice of best fit for multiple results.

In the late 1970s and early 1980s the International Commission on Radiological Protection (ICRP) issued the various parts of publication 30 which gave dosimetric models for the respiratory and gastrointestinal tracts and systemic biokinetic models to be used in internal dosimetry. Since that time there have been many developments in modelling the behaviour of radionuclides in the human body. Following the issue of Publication 60 in 1991, an further series of publications were developed by ICRP giving a new generation of biokinetic and dosimetric models. They were used to calculate dose coefficients (committed effective dose per unit intake in Sv $Bq^{-1}$) for both workers in Publication 68 and members of the public in Publications 56, 67, 69, 71 and 72. These dose coefficients were also adopted in the IAEA International Basic Safety Standards, in the EURATOM Directive and in the national regulations and codes of practice of many countries.

For consistency, dosimetry services subject to these regulations and standards need to use the current models when assessing doses from bioassay measurements such as whole-body or organ content or daily excreta. The complexity of these models and the efforts required for the gathering of data for the assignment of patterns and time of intake, lung absorption parameters, particle sizes and individual variations have contributed to uncertainties in individual dose assessments for dose record keeping. Some guidance on the interpretation of monitoring data to assess occupational exposures from intakes of radionuclides, based upon the biokinetic models given in Publications 30 and 68, was provided in ICRP Publications 54 and 78 respectively.

Recent interlaboratory comparisons have, however, revealed that different laboratories can obtain quite different estimates of intakes and doses when provided with the same monitoring data. The current generation of models is designed to represent physiological processes more realistically than the previous generation of simple compartment models. As a consequence the new models are generally more complex recycling models, as for the alkaline earths and actinides, and more difficult to work with. ICRP has recognised that it needs to give more guidance on how its biokinetic and dosimetric models can be applied in the interpretation of bioassay data.

Following the expected issue of the 2007 Recommendations of ICRP, Committee 2 intends to issue a series of publications concerned with assessing doses to workers from occupational intakes of radionuclides (OIR). The series of publications will give dose coefficients (Sv $Bq^{-1}$) and supporting biokinetic and dosimetric models for selected elements and radionuclides together with data that can be used for monitoring and dose assessment of people who are occupationally exposed. In addition a Guidance Document is being developed to provide advice on the interpretation of monitoring data and to support the OIR series of publications.
This draft of the Guidance Document is issued by ICRP for formal consultation. It emphasises that the dose assessor needs to have a good understanding of the specific

circumstances of an individual's exposure: radionuclides handled; likely physical and chemical forms; routes of intake; time and pattern of intake. But it also gives advice on assumptions to be made when important information is incomplete. It also proposes a structured approach that can be used for dose assessment and guidance is given on when it is appropriate to use default parameter values and the extent of analysis needed. This guidance is not intended to be prescriptive but rather to provide support to the occupational health physicist and the extent of its application will depend upon the experience available.

**Membership**

Over the period of preparation of the present report, the Task Group on Internal Dosimetry had the following full members (since 2000):

*Members:*

| | | |
|---|---|---|
| J W Stather (Chair) | R A Guilmette | H Métivier |
| E Ansoborlo | J D Harrison | D Noβke |
| M R Bailey | N Ishigure | F Paquet |
| A Bouville | R W Leggett | D M Taylor |
| H Doerfel | J L Lipsztein | |

(W Bolch, as the Chairman of the Task Group on Dose Calculations, is an 'ex officio' member of the Task Group on Internal Dosimetry)

*Corresponding Members:*

| | | | |
|---|---|---|---|
| A Birchall | J Inaba | M-L Perrin | D Whillans |
| X-A Chen | W Kreyling | A Rannou | |
| F A Fry | D Lee | P Roth | |
| O Hoffman | D Newton | T J Smith | |

The membership of the Task Group on Dose Calculations over the period of preparation of the report was:

*Members :*

| | | |
|---|---|---|
| W Bolch E (Chair) | H Menzel | A W Phipps |
| V Berkovski | D Noβke | M Zankl |
| K F Eckerman | M Pellicconi | |
| N Hertel | N Petoussi-Henβ | |

*Corresponding Members:*

| | | |
|---|---|---|
| L Bertelli | R W Leggett | M Stabin |
| T P Fell | I A Likhtarev | A Ulanovsky |
| J G Hunt | G Miller | G Xu |
| L Johansson | M Pelliccioni | |
| G M Kendall | R Richardson | |

The membership of the Working Party on Bioassay Interpretation over the period of preparation of the report was:

*Members:*

| | | |
|---|---|---|
| F A Fry (Chair) | A Bouville | D Lee |
| E Ansoborlo | H Doerfel | J L Lipsztein |
| M R Bailey | R A Guilmette | G Miller |
| V Berkovski | N Ishigure | D Noßke |

During the period of preparation of this report, the membership of Committee 2 was:

(2001-2005)

| | | | |
|---|---|---|---|
| C Streffer (Chair) | K F Eckerman | H-G Menzel | D M Taylor |
| M Balonov | F A Fry | H Métivier | Y Zhou |
| B B Boecker | J Inaba | H Paretzke | |
| A Bouville | I A Likhtarev | A S Pradhan | |
| G Dietze | J L Lipsztein | J W Stather | |

(2005-2009)

| | | | |
|---|---|---|---|
| C Streffer (Chair) | G Dietze | H-G Menzel | Y Zhou |
| M Balonov | K F Eckerman | F Paquet | |

| | | |
|---|---|---|
| V Berkovski | J D Harrison | H Paretzke |
| W E Bolch | N Ishigure | A S Pradhan |
| A Bouville | J L Lipsztein | J W Stather |

The work of the Task Groups was greatly aided by significant technical contributions from:

J W Marsh and G Etherington

**SUPPORTING GUIDANCE DOCUMENT**

# INTERPRETATION OF BIOASSAY DATA

***The Main Commission of ICRP is to issue new Recommendations on the principles of radiological protection. It posted a second draft of these recommendations on its web site for consultation in June 2006 and a third draft in January 2007. The new Recommendations will include advice on protection in relation to occupational exposure and are expected to be published in 2007 after approval by the Main Commission.***

***Following the issue of the 2007 Recommendations, Committee 2 of ICRP intends to issue a series of publications concerned with assessing doses to workers from occupational intakes of radionuclides (OIR). A series of publications will be issued giving dose coefficients (Sv $Bq^{-1}$) and supporting biokinetic and dosimetric models and data for selected elements and radionuclides that can be used for monitoring and dose assessment of people who are occupationally exposed. In addition a Guidance Document is to be issued to provide advice on the interpretation of bioassay data and which will support the OIR series of publications.***

***This draft of the Guidance Document is issued for formal consultation. It follows publication of an earlier informal consultation draft in January 2006. The information provided is a significant development from that given in previous publications by ICRP covering the interpretation of bioassay data (ICRP, 1988a, 1997b) and comments from those involved in occupational health physics will be particularly welcome. This consultation document has been developed from the January 2006 draft posted on the ICRP web site for informal consultation and takes into account the comments received.***

***In addition to the main text and associated tables and figures, an Appendix is included on uranium. This Appendix illustrates the approach that will be used for providing data on the various elements and their radionuclides in what will be a separate series of publications on Occupational Intakes of Radionuclides-Dose Assessment and Monitoring (OIR).***

## 1. INTRODUCTION

### 1.1 Background

In 1988 the International Commission on Radiological Protection (ICRP) issued Publication 54 on *Individual Monitoring for Intakes of Radionuclides by Workers: Design and Interpretation* (ICRP, 1988a). That document gave guidance on the design of monitoring programmes including the interpretation of results of measurements of intakes of radionuclides by workers. It was a companion volume to the various parts of Publication 30 (ICRP, 1979, 1980a,b, 1988b) which gave values of Annual Limits on Intake for radionuclides based on the then current dosimetric models of the respiratory and gastrointestinal tracts and the biokinetic models for systemic behaviour of each element. Publication 54 (ICRP, 1988a) took into account the radiological protection principles in Publication 26 (ICRP, 1977) and the anatomical and physiological data in Reference Man (ICRP, 1975).

Publication 54 was superseded by Publication 78 (ICRP, 1997b) which was consistent with the biokinetic and dosimetric models used to calculate dose coefficients for intakes of radionuclides by workers given in Publication 68 (ICRP, 1994b). Publication 68 took into account the more recent recommendations from ICRP on radiological protection principles given in Publication 60 (1991). It also used the Human Respiratory

Tract Model (HRTM) (ICRP, 1994a) for inhaled radionuclides, the updated basic anatomical and physiological data for the skeleton in Publication 70 (ICRP, 1995b) and a revision of the systemic models for selected nuclides given in Publications 56, 67, 69 and 71 (ICRP, 1989, 1993b, 1995a,c). Publication 68 gave dose coefficients for intakes of radionuclides (doses per unit intake, Sv $Bq^{-1}$) but not ALIs as ICRP now wished to stress the need to take account of all exposures to radiation in the workplace both from external radiation and intakes of radionuclides. Publication 78 provided data for bioassay interpretation of selected radioisotopes of 15 elements.

The Main Commission of ICRP is expected to issue new Recommendations on the principles of radiological protection in 2007 following an extensive period of consultation. The new recommendations will give further advice on protection in relation to occupational exposure. They will also include revised tissue and radiation weighting factors.

There have been further developments in the work of Committee 2 of ICRP that impact on the calculation of doses from intakes of radionuclides:

- ICRP has issued Publication 89 on *Basic Anatomical and Physiological Data for use in Radiological Protection* (ICRP, 2002a) which updates the information on reference man given in Publication 23 (ICRP, 1975);
- for the calculation of most organ and tissue doses ICRP is adopting new human voxel phantoms for the adult male and adult female based upon medical imaging data. The models are to be consistent with the information given in Publication 89.; In particular, the voxel phantoms will be used to calculate the Specific Effective Energy, SEE (T←S), the equivalent dose in target tissue, T per transformation of a given nuclide in source organ, S. Exceptions are the respiratory tract, the alimentary tract and skeletal tissues for which doses to target cells are calculated separately;
- The radionuclide decay data given in Publication 38 (ICRP, 1983) are being updated;
- A new biokinetic and dosimetric model for the Human Alimentary Tract has been issued as Publication 100 (ICRP, 2006) and supersedes that for the gastrointestinal tract given in Publication 30; and
- the biokinetic models for the systemic behaviour of radionuclides are under review.

The main publications by ICRP since Publication 30 that relate to occupational exposure to radionuclides are given in Table 1.1.

Following the issue of the 2007 Recommendations, Committee 2 of ICRP will issue a series of publications concerned with assessing doses to workers from occupational intakes of radionuclides (OIR). Task Groups of Committee 2 are preparing these publications that will take into account the new information from ICRP described above.

For intakes of radionuclides in the workplace a number of routes are possible including inhalation, ingestion, entry through intact skin and through wounds. Dose coefficients for intakes by inhalation and ingestion have been given in previous ICRP publications. Revised dose coefficients for inhalation intakes will apply the Human Respiratory Tract Model (HRTM) (ICRP, 1994a) while ingestion intakes will use the Human Alimentary Tract Model (HATM) (ICRP, 2006). This model will also be applied to material cleared to the throat and swallowed after inhalation. In the HATM fractional absorption of radionuclide is specified by $f_A$ values, instead of $f_1$ values as given for the model for the gastrointestinal tract described in Publication 30 (ICRP, 1979).

ICRP has generally not given information on the transfer of radionuclides from wound sites to blood and other organs and tissues. Information on the transfer of radionuclides from wound sites has, however, being reviewed by a Working Group of NCRP and these data have been used to develop a series of models to describe the transfer of material from wounds after intakes in different physico-chemical forms (NCRP, 2007 in preparation). ICRP considers that these models are a valuable contribution to methods for assessing radiation doses from exposure to radionuclides in the workplace. This publication illustrates the application of the wound model.

Biokinetic models are also used by ICRP to describe the distribution and excretion of the soluble component of material transferred to the systemic circulation from the various sites of intake. As indicated above, the systemic models recommended for the elements are being reviewed and revised as necessary to take account of more recent information and in particular to provide models that are appropriate for both dosimetry and bioassay interpretation.

As a consequence of these developments, dose coefficients for intakes of radionuclides and data for the interpretation of bioassay data will be provided to replace the information given in Publications 30 and 68 (ICRP, 1979; 1980 a,b,c;1994b) and in Publications 54 and 78 (ICRP, 1988a, 1997b). This information will be published in the OIR series of publications. Each publication will bring together both dosimetric information and information for the interpretation of bioassay data for various chemical forms of the elements and radionuclides found in the workplace.

In addition to the need to update these basic data ICRP has recognised the need to provide further advice on the interpretation of bioassay data. Experience from a number of interlaboratory exercises, notably the 2001 European interlaboratory comparison (Doerfel et al, 2003) have demonstrated that there can be wide differences between individuals and laboratories in the approaches used to interpret monitoring data for those occupationally exposed to radionuclides. As a consequence the results obtained from different laboratories can be quite variable.

The variability in the interpretation of bioassay data led to the setting up of a European project in the EC 5$^{th}$ Framework programme to develop guidance on internal dose assessments (Project IDEAS). The project involved examination of a number of case studies and from this guidelines were developed on a structured approach to the interpretation of monitoring data (Doerfel et al, 2002, Doerfel et al 2006). It included an intercomparison exercise held jointly with the IAEA, in which participants were encouraged to follow the guidelines, and to comment on them (Doerfel, 2005; IAEA, in press).

In addition, the EC also supported a project to examine how best to ensure that resources available for internal dose assessment are used to maximum effect (project OMINEX) (Etherington et al, 2003). The project has developed methods for optimising the design and implementation of internal exposure monitoring programmes.

The development of this Guidance Document aims to extend the previous advice given in Publications 54 and 78 and in so doing to take into account the experience gained through the IDEAS and OMINEX projects and also that coming from publications by the IAEA giving advice on assessment of occupational exposure due to intakes of radionuclides (IAEA, 1996, 2000, 2004). It will complement the information given in the OIR series of Publications.

## 1.2 Purpose and Scope

The aim of this document is to provide guidance on the interpretation of bioassay data to allow for a better harmonisation of approaches by individuals and laboratories.

This document gives general advice on the design of individual monitoring programmes and more detailed guidance on the interpretation of the results of monitoring data designed to estimate the intakes of radionuclides by workers and the associated doses. It also includes some illustrative examples. It is to be used in conjunction with the Commission's general principles of monitoring for the radiation protection of workers (ICRP, 1997, 2000, 2007).

This guidance does not give radionuclide specific data as these are to be given in the OIR publications which detail the biokinetic and dosimetric models developed for describing the behaviour of intakes of radionuclides as well as the updated element specific systemic models. Radionuclide specific dose coefficients are also given in these publications together with tabulated data and figures to aid the interpretation of bioassay data. It is the application of these data that is the subject of this guidance document. Some illustrative examples are included as Appendices.

The predicted parameter values to be provided by ICRP in the OIR series of publications have been calculated for a reference worker and for the specified biokinetic models and parameter values. Occupational health physicists in organisations where there is the potential for significant exposures from intakes of radionuclides need to consider carefully the particular working conditions, the physical and chemical nature of the radionuclides handled and the design of monitoring programmes that meet their specific needs (ICRP, 2007).

## 1.3 Dose Per Unit Content

The system of dose assessment from bioassay data that is generally applied relies first on the calculation of the intake of a radionuclide either from direct measurements (e.g. external monitoring of the whole body or of specific organs and tissues) or indirect measurements (e.g. urine, faeces or environmental samples). Predicted values of the measured quantities for unit intake are recommended by ICRP and these values can be used to estimate the intake (ICRP, 1997b). The effective dose resulting from any intake is then calculated from the dose coefficients recommended by ICRP (ICRP, 1994b), and also given in the EU Basic Safety Standards (EU, 1996) and International Basic Safety Standards (IAEA, 1996). Tabulations of data in the OIR series of publications are to be provided for this approach to be followed, as was the case in Publications 54 and 78 (ICRP, 1988; ICRP, 1997b).

A paper by Berkovski *et al* (2003) has indicated that there can be advantages in calculating the committed effective dose directly from the measurements using functions that relate them to the time of the intake. The measurements could be the whole body or organ content, a urine or faecal sample, or even an environmental measurement. In some circumstances this approach can simplify the procedure normally used for assessing bioassay data and reduce the opportunities for misreading tabulations of data. An additional advantage is that in some situations the dose per unit content is less sensitive to the choice of parameter values then the intake. For example, Berkovski *et al* (2003) showed that for a number of chemical forms of radionuclides the "dose per unit content" is largely insensitive to the choice of inhaled particle size for a wide range of measurement times following an intake. The need to make a decision on the appropriate activity median aerodynamic diameter (AMAD) of an aerosol may not therefore arise. Similar considerations can apply to the choice of absorption Type as defined in HRTM. Care is still needed in the choice of the most appropriate measurement data and defining the time of the intake. This is described in Annex D.

The OIR series of publications will therefore include data on dose per unit content. The inverse of the dose per unit intake, "content per unit dose" (usually presented as the measured activity corresponding to a committed effective dose of 1 mSv, as a function of time between intake and measurement), is particularly useful for planning purposes. Comparison can be made with the detection limit of each available technique.

## 1.4 Structure of the Report

Chapter 2 of this Guidance Document sets out the principal recommendations of the Main Commission of ICRP relevant to the monitoring of workers exposed to radioactive materials. It also considers the use of effective dose and the requirement to demonstrate compliance with dose limits when radiation exposures are to both external radiation and intakes of radionuclides.

Chapter 3 gives an overview of those aspects of the biokinetic and dosimetric models that are relevant to an understanding of how radioactive materials behave in the body. This includes a summary of the Human Respiratory Tract Model (HRTM) and the Human Alimentary Tract Model (HATM). It also describes the NCRP wound model. The principles of the models used for describing the behaviour of elements following uptake to blood and their excretion are also summarised. Uncertainties associated with the use of these models are considered as is individual variability. A Foundation Document prepared by Committee 2 on Dosimetry, to be issued as an Annex to the new Recommendations, will provide additional information on dose assessment from intakes of radionuclides (ICRP, 2007).

A description of methods for individual monitoring is given in Chapter 4. The Chapter covers *in vivo* measurements and the analysis of excreta and other biological materials as well as environmental monitoring.

The general principles for design of monitoring programmes, types of programmes and monitoring requirements are summarised in Chapter 5. Also covered briefly are wound monitoring and the potential effects of medical intervention.

The dosimetric data and functions provided by ICRP in the OIR series of publications for dose assessment and the interpretation of bioassay data are summarised in Chapter 6. Apart from updates of the data previously given in Publications 54 and 78 tabulated values of dose per unit content (Sv $Bq^{-1}$) and figures giving content per unit dose (Bq $Sv^{-1}$) are also to be provided to facilitate the interpretation of bioassay data. These data will be summarised in the OIR publications and given in more detail on a CD-ROM. An example is included as the Appendix on Uranium.

General aspects of dose assessment are considered in Chapter 7. The Chapter examines the need to understand the exposure situation and radionuclide(s) being handled as well as their physico-chemical form. It also stresses the need for any assessment to be proportionate to the expected exposure. It discusses the requirements for an effective monitoring programme and summarises approaches to data handling for single or multiple measurements.

Chapter 8 gives practical guidance that aims to help the occupational health physicist step through the various stages of a structured approach needed for an appropriate analysis of the data. The extent of application of this approach will depend upon the radionuclide involved, its physical and chemical form, the extent of any possible exposure and the experience of the occupational health physicist. It is stressed that this is not intended as a prescriptive process but is a guide to help ensure that the most appropriate dose assessments are obtained. The structured approach described is developed in terms of expected levels of exposure with the effort being related to the

likely radiation dose; higher potential exposures requiring a more detailed assessment. The basic approach is described in the Chapter while more details are given in Annex G, which covers situations in which more extensive analysis is required.

A series of supporting Annexes is included. These are:

Annex A - Proposed content of accompanying CD-ROM

Annex B - Examples of the interpretation of monitoring data

Annex C - Direct dose assessment

Annex D - Dose assessments with dose per unit content

Annex E - Data fitting

Annex F - Obtaining an unbiased estimate of intake in routine monitoring when the time of intake is unknown

Annex G - Structured approach to advanced dose assessment

Annex H - Examples of the application of the NCRP wound model (to be developed)

Information given in an Appendix (OIR-27) gives illustrative data for uranium that will be issued in Part 1 of the series of publications on Occupational Intakes of Radionuclides - Dose Assessment and Monitoring. This Appendix will not be included in the published version of this Guidance Document.

# 2 RELEVANT RECOMMENDATIONS OF THE COMMISSION

***The Main Commission of ICRP is expected to issue new Recommendations in 2007. Some changes in terminology were proposed in the 2005 consultation draft that was issued by ICRP (ICRP, 2004) but these were not carried through into the updated 2006 draft (ICRP, 2006). Much more emphasis is now placed on the use of dose constraints although they were described in Publication 60. The published version of this guidance will be consistent with the 2007 Recommendations of ICRP.***

## 2.1 Limits, Constraints and Investigation Levels

For occupational exposure the Commission recommends a limit on effective dose of 100 mSv in a 5 year period, giving an average value of 20 mSv in a single year, with the further provision that the effective dose should not exceed 50 mSv in any single year (ICRP, 1991, 2005). There are additional annual limits on equivalent dose in the lens of the eye, the skin, and the hands and feet, but these are not likely to be relevant in the context of intakes of radionuclides. For internal exposure, the Commission recommends that intakes of radionuclides should be controlled on the basis of a maximum value of committed effective dose of 20 mSv per year (ICRP, 1991). Where workers may be exposed to both external radiation and intake of radionuclides, the annual dose limit applies to the sum of the effective doses from external radiation and the committed effective dose from intakes of radionuclides occurring within the year. Similar advice is given in the new Recommendations (ICRP, 2007).

In the new Recommendations emphasis is placed on the use of *dose constraints*, although the principles were described in Publication 60. A dose constraint is considered to be the most fundamental level of protection for the most exposed individuals from a single source within a class of exposure. The Commission continues to recommend that occupational exposure in planned situations is controlled by the process of optimisation below the constraint and the use of dose limits. For many types of occupation in planned situations, it is possible to reach conclusions about the level of individual doses likely to be incurred in well-managed operations. This information can then be used to establish a dose constraint for that type of occupation that will be less than the dose limit. The Commission considers that it will usually be appropriate for such dose constraints to be fixed at the national or local level. Constraints may be used to set *investigation levels*. ICRP stresses that the chosen value for the dose constraint will depend upon the prevailing circumstances of the exposure under consideration. It does not however, represent a demarcation between 'safe' and 'dangerous' or reflect a step change in the associated health risks for individuals.

Investigation levels are defined by ICRP (1997b) as levels above which the cause or the implications of the exposure should be examined. They are therefore used retrospectively. Investigation levels can be set for any operational parameter related to the individual or to the working environment. For individual monitoring of exposure to intakes of radionuclides, they are most likely to relate to a measured body or organ/tissue content, an activity level in excreta, or an air concentration measured by a personal air sampler (this is addressed further in Section 5.6).

The most likely route of intake of radionuclides in the workplace is by inhalation. In most normal situations, however, the extent of any possible intakes should have been established and precautions taken. The use of respiratory protection in situations where there is the potential for inhalation intakes is well established (IAEA, 1999a). In practice exposures to external radiation generally dominate occupational exposure. Thus UNSCEAR noted in its 1993 report that doses from internal exposure varied with the work but were mainly in the region of about 10% of those from external exposure (UNSCEAR, 1993).

In the case of intakes of mixtures of radionuclides, and in the absence of external radiation exposure, the total intake during a year should be controlled so as not to give rise to a committed effective dose greater than 20 mSv. Investigation levels set for individual radionuclides should take account of the presence of other radionuclides in the working environment and also of the contribution to dose from external radiation exposure. Further guidance on demonstrating compliance with dose limits and constraints is given below in Section 2.5.

## 2.2 Objectives of Monitoring

The purpose of monitoring for exposure to radionuclides is to verify and document that the worker is protected adequately against risks arising from intakes, and that the protection afforded complies with legal requirements. Two types of monitoring can be identified: *workplace monitoring* and *individual monitoring* (ISO/FDIS 20553, 2005(draft)).

Workplace monitoring makes use of measurements made in the working environment. An example is the measurement of radionuclide air concentrations using static air samplers. Workplace monitoring can be used to provide an assessment of exposure for groups of workers, but requires assumptions to be made about exposure conditions. It is mainly used where individual monitoring is not justified, or where the sensitivity of individual monitoring is inadequate. It is also of value in demonstrating that working conditions meet safe working criteria and have not changed.

Individual monitoring makes use of measurements made to assess exposure to radionuclides or intakes of radionuclides for a single worker. The principal objectives are:

- to obtain an assessment of the committed effective dose and, where appropriate, the equivalent dose in significantly exposed tissues, so as to demonstrate compliance with managerial and regulatory requirements;
- to contribute to the control of the operation and the design of facilities; and
- to provide valuable information for the initiation and support of any appropriate health surveillance and treatment in the case of accidental exposure.

Usually it is necessary to carry out only a simple assessment of dose to demonstrate compliance with dose limits when doses are expected to be only small fractions of the dose limits (often assuming default parameter values for the dose assessment). In some countries it may be unnecessary to make an assessment of individual dose at all, the measured value being compared with an appropriate reference value. However, there is considerable variation between countries and local guidance or regulations needs to be adhered to. At higher doses more emphasis will need to be placed upon specific dose assessments for the exposed individual and the circumstances of any exposure.

The monitoring programme has to be defined prospectively from a consideration of potential exposure scenarios, not actual exposures (which in general are likely to be much less). If monitoring were to be defined on the basis of exposures actually received then in general very little monitoring would be carried out. Practical experience can also inform how monitoring should proceed.

In contrast, where accidental exposures or other incidents have occurred that could have resulted in an accidental intake, a comprehensive assessment of dose would normally be required.

Individual monitoring for radionuclides in either normal or accident situations may in general employ one or more of the following techniques:

- direct measurement of radionuclides in the whole body or in regions of the body;
- measurements of activity in excreta or exhaled air; and
- measurement of activity concentration in air by means of an air sampler carried on the person.

Measurements, together with information about the workplace, should enable each radionuclide to be identified, its activity quantified, and the measurement result interpreted in terms of intake and/or committed effective dose. There may be some circumstances where individual monitoring techniques, as defined above, are not adequate to assess doses adequately and in these situations it may be necessary to combine individual and workplace monitoring techniques.

The effort involved in monitoring doses needs to be proportionate to the radiation dose which could potentially be received by the worker, as described in Chapters 7 and 8.

## 2.3 Categories of Individual Monitoring Programme

In Publication 60 (ICRP, 1991) the Commission recommended that individual monitoring for intakes of radioactive material should be used routinely for workers who are employed in areas that are designed as controlled areas, specifically in relation to the control of contamination and in situations in which there are grounds for expecting significant intakes. Routine monitoring would be required in conditions of essentially continuous risk of contamination of the workplace as a result of normal operations.

Measurements in a routine monitoring programme are made at pre-determined times not related to known intakes, and therefore it is necessary to make some assumptions about the pattern of intakes. National of local legislation or regulations may also set the requirements for systematic routine monitoring that may be required if exposures could exceed a specified fraction of the dose limit or a dose constraint

Other monitoring programmes may be conducted in relation to a particular task, or to determine intakes in actual or suspected abnormal conditions. In these circumstances, the time of intake, or potential intake, is likely to be known and workplace monitoring programmes may provide some information on the physical and chemical nature of any contamination.

Routine, confirmatory, special and task related monitoring are described further in Chapter 5. Occasional measurements may also be carried out to verify satisfactory working conditions.

## 2.4 Needs for Individual Monitoring

The Commission stated in Publication 60 (ICRP, 1991) that it is necessary to identify groups of workers for whom individual monitoring is needed. The decision to provide individual monitoring depends on many factors. Routine individual monitoring for intakes of radioactive material should be used for workers where there is a reasonable probability that committed doses from intakes of radionuclides in a year will exceed 1 mSv. This would apply for people employed in areas that are designated as controlled areas specifically in relation to the control of contamination and in which there are grounds for expecting significant intakes.

The use of individual monitoring for workers whose doses could exceed 1 mSv is common practice in many organisations although it may not be required by legislation. The 1 mSv criterion is not achievable by individual monitoring in some cases, e.g. exposures to some physico-chemical forms of $^{239}$Pu or other actinides when assessed from urine monitoring. In such situations a combination of monitoring techniques, including measurements on the working environment, may be needed.

There are some other factors, however, such as technical and managerial issues, which could support arguments for the assessment of individual dose at lower levels, at least for those radionuclides for which it is straightforward. The following Guidance therefore considers assessments of doses in situations where committed doses from intakes of radionuclides could exceed 0.1 mSv in a year.

Some working situations where experience has shown that it is necessary to give consideration to routine individual monitoring for internal exposure of workers include the following operations:

- the handling of large quantities of gaseous and volatile materials, e.g. tritium and its compounds in large scale production processes, in heavy water reactors and in luminising;
- reactor operations
- the processing of plutonium and other transuranic elements;
- the processing of thorium ores and use of thorium and its compounds (these activities can lead to internal exposure from both radioactive dusts and thoron [radon-220] and its progeny);
- the milling and refining of uranium ores;
- natural and enriched uranium processing and reactor fuel fabrication;
- work with naturally occurring radioactive materials (NORM);
- the production of radiopharmaceuticals;
- the production of large quantities of radionuclides; and
- the handling of large quantities of iodine-131, e.g. for therapy; and
- exposures to radon and its decay products in mines

The results of monitoring of the workplace may also indicate a need for a temporary programme of special individual monitoring (see Chapter 5) aimed at identifying any need for a routine programme of workplace monitoring.

## 2.5 Effective Dose

The main and primary use of effective dose is to provide a means of demonstrating compliance with dose limits. In this sense effective dose is used for regulatory purposes worldwide.

The calculation of reference dose coefficients and dose conversion factors is based on reference anatomical data for the organs and tissues of the human body together with defined biokinetic and dosimetric models. The general approach for internal dosimetry is to monitor individuals or the environment and from these measurement data to assess the radionuclide intake. The dose coefficients published by ICRP are then used to assess the effective dose. The weighting factors used in the calculation of reference dose coefficients and conversion factors are selected values that apply to a population of both genders and all ages. Thus dose coefficients and the reference models and weighting factors used in their calculation are not individual

specific but apply to a reference person for the purposes of regulatory control. The reference dose coefficients and dose conversion factors provided by ICRP in the OIR series of publications (and here illustrated in Appendix OIR-27 on Uranium) are to be applied in the estimation of effective doses in relation to exposures in the workplace, principally for planning purposes and for assessing occupational exposures in normal situations.

Particularly in retrospective dose assessments for occupational exposures, information may be available that differs from the standard reference parameter values used in the calculation of dose conversion factors and dose coefficients. As discussed in Chapters 7 and 8, in such situations it may be appropriate, depending on the level of exposure, to use specific data in the assessment of exposure or the intake and calculation of doses. It is, therefore, important to distinguish between those parameter values that might be altered in the calculation of effective dose under the particular circumstances of an exposure and those values that cannot be changed under the definition of effective dose.

In the assessment of effective dose in occupational situations of exposure to radionuclides, changes may reasonably be made to the assumed physical and chemical characteristics of inhaled or ingested radionuclides to better assess doses from intakes. In addition, where appropriate information is available, specific biokinetic data may also be used to improve the assessment of the intake. These changes need to be recorded. Examples of the use of material specific data in the calculation of doses from inhaled radionuclides have been given in Supporting Guidance 3 (ICRP, 2002).

For retrospective assessments of occupational doses to specific individuals from external exposure and from intakes of radionuclides where the radiation dose could exceed a limit, it may be considered appropriate to make specific individual estimates of dose and risk. Consideration might then be given to changes in dosimetric assumptions (e.g. body and organ mass) used to calculate absorbed doses, and organ-specific risk estimates relating to the age and gender of the individual and the radiation exposure. Such changes from reference parameter values are not consistent with the definition or intended use of effective dose. They should only be performed by radiation protection specialists, with the level of effort determined by the level of exposure. In such situations any changes to parameter values must be documented.

Effective dose is used to limit the occurrence of stochastic effects (cancer and hereditary effects) and is not applicable to the assessment of the possibility of "tissue reactions" (deterministic effects in Publication 60). In the dose range below the annual effective dose limit, the occurrence of most, and probably all, tissue reactions should be avoided if the effective dose is controlled. Only in a few cases (e.g. an acute localised exposure of a single organ with a low tissue weighting factor such as the skin) could the use of the annual limit on effective dose be insufficient to avoid tissue reactions. In such cases local tissue doses will also need to be assessed.

In cases of incidents and accidents that could give rise to tissue reactions, it is necessary to estimate absorbed dose and dose rates to organs and tissues and to take into account dose-response relationships to assess the potential for radiation effects that are likely to occur above dose thresholds (NCRP, 1990; ICRP, 1989; 1991; 2007). It should also be noted that in cases of accidents involving high-LET radiations (neutrons and alpha particles), radiation weighting factors ($w_R$) applicable to stochastic effects do not apply to tissue reactions; values of Relative Biological Effectiveness (RBE) relevant to tissue reactions should be used to assess the consequences of any exposures and the need for any remediation measures.

Effective dose is a risk related quantity based upon the radiation detriment following whole body exposure at low doses. The $w_T$ values are selected values that are

chosen to take account of the contribution of individual organs and tissues to total radiation detriment from stochastic effects, in terms of cancer and hereditary effects, on the basis of current epidemiological evidence. Furthermore, $w_T$ values are averages applying to both sexes and all ages. Effective dose is not, therefore, an appropriate quantity for use in epidemiological studies of radiation risks. Epidemiological analyses instead require estimates of absorbed doses to tissues and organs, taking full account, to the extent possible, of the circumstances of exposure and the characteristics of the exposed individuals in the study population. Similarly, absorbed doses, not effective doses, are required for calculations of probability of causation of cancer in exposed individuals. These issues are discussed further in the Annex on Dosimetry prepared by Committee 2 to underpin the new Recommendations (ICRP, 2007).

In summary, effective dose should be used for assessing exposure and controlling stochastic effects for regulatory purposes. It can be used to demonstrate compliance with dose limits and for dose records. In situations approaching or even exceeding the dose limits, effective dose provides a convenient quantity for the assessment of overall radiation exposure, taking account of all exposure pathways, internal and external, for dose record keeping and regulatory purposes. Effective dose is not individual-specific but applies to a reference person. In retrospective situations the assessment of effective dose gives an insight into the quality of radiological protection and gives information on whether the dose limits could have been exceeded.

However, there are situations in which the use of effective dose is not appropriate and individual organ and tissue absorbed doses should be calculated. These include epidemiological studies, assessment of the probability of causation of cancer, assessments of the possibility of tissue reactions, or where treatment or medical surveillance are considered.

Because effective dose is calculated using reference parameter values and applies to a reference person, no uncertainty is attached to the reference dose coefficients. However, it is recognised that there may be substantial uncertainties in the dose and risk to any individual, including those contributed by uncertainties in the intake, in biokinetic and dosimetric models and uncertainties in age- and gender- specific risk coefficients. In the context of the assessment of occupational exposures, it is helpful to distinguish between reference values of effective dose which are not regarded as uncertain and individual-specific estimates of equivalent dose and risk to organs and tissues for which an estimate of uncertainty might be required. For example, assessments of uncertainties may be needed in retrospective assessments when effective dose might exceed limits and in epidemiological studies.

## 2.6 Control of Worker Doses

In occupational exposure doses may arise from both external and internal radiation sources. For external exposure individual dose monitoring is usually performed by measuring the personal dose equivalent using personal dosemeters and taking this measured value as an acceptable assessment of the value of effective dose. For internal exposure the committed effective dose values are determined based on measurements of other quantities (e.g. *in vivo* monitoring, urinary excretion, etc.) and the application of conversion coefficients. However, for practical purposes the values from both kinds of quantities can be combined in the assessment of the value of total effective dose for demonstrating compliance with dose limits and constraints.

For practical purposes, the effective dose, $E$, can in most situations of occupational exposure be estimated from operational quantities using the following formula:

$$E \cong H_p(10) + E(50)$$

where $H_P(10)$ is the personal dose equivalent from external exposure, normally defined by the dose equivalent at a depth of 10 mm in the body below the position where the dosemeter is worn, and the committed effective dose from internal exposure is assessed by:

$$E(50) = \sum_{\mathrm{j}} e_{\mathrm{j,inh}}(50) \cdot I_{\mathrm{j,inh}} + \sum_{j} e_{\mathrm{j,ing}}(50) \cdot I_{\mathrm{j,ing}}$$

where $e_j(50)$ is the committed effective dose coefficient per unit intake (Sv $Bq^{-1}$) of a radionuclide integrated over 50 years after intake by inhalation (inh) or ingestion (ing). The intakes, *j* may be for one or a number of radionuclides. The commitment period of 50 years relates to the life expectancy of a young person entering the workforce.

Workplace exposures to radionuclides are usually assumed to occur by inhalation although direct ingestion intakes can occur. In the calculation of the effective dose from specific radionuclides allowance may need to be made for the characteristics of the material taken into the body.

If incorporation of radionuclides through the skin or wounds occurs an additional term for the associated effective dose would have to be included. Exposure to isotopes of radon and their decay products may also need to be taken into account (ICRP, 1993a).

The dose coefficient for intakes of radionuclides (Sv $Bq^{-1}$) is the fundamental quantity recommended by ICRP for protection purposes. In the assessment of committed effective doses from operational data related to an actual intake of specific radionuclide(s) or of radionuclide concentration(s) in the air at a workplace it is often useful to refer these data to derived parameters such as the Annual Limit on Intake (*ALI*) and the Derived Air Concentration (*DAC*).

The *ALI* was defined in Publication 60 (ICRP 1991, para S30) as an intake (in Bq) of a radionuclide in a year which would lead to a committed effective dose of 20 mSv (0.02 Sv). The average annual limit on effective dose for workers is thus:

$$ALI_j = \frac{0.02}{e_j(50)}$$

The *DAC* is the activity concentration in air in $Bq/m^3$ of the radionuclide considered which would lead to an intake of an *ALI* assuming a gender averaged breathing rate of 1.2 $m^3\ h^{-1}$ (ICRP, 1993b, p23) and an annual working time of 2000 h. Then the *DAC* is given by:

$$DAC_j = \frac{ALI_j}{2400}$$

The Commission does not now give ALI values, as it considers that for compliance with dose limits it is the total dose from external radiation as well as from intakes of radionuclides that must be taken into account as indicated above. The ALI is, however, a concept that is useful in various practical situations, e.g. in characterising the relative hazard of radiation sources to ensure that appropriate administrative controls are in place and in determining values of the DAC.

## 2.7 Female Workers

It is the Commission's policy (ICRP, 2007) that the methods of protection at work for women who are or may be pregnant should provide a level of protection for the embryo/fetus similar to that provided for members of the public. The Commission considers that this policy will be adequately applied if the mother is exposed, prior to her declaration of pregnancy, under the system of protection recommended by the Commission. Once pregnancy has been declared, and the employer notified, additional protection of the embryo/fetus should be considered. The working conditions of a pregnant worker, after declaration of pregnancy, should be such as to make it unlikely that the additional equivalent dose to the fetus would exceed about 1 mSv during the remainder of the pregnancy.

ICRP (2001, 2004) has provided information in Publications 88 and 95 on doses to the embryo, fetus and newborn child following intake of radionuclides by female workers either before or during pregnancy or during lactation.

Comparisons of fetal dose coefficients (ICRP, 2001) with corresponding adult dose coefficients show that doses received by a woman from intakes before or during pregnancy will in most cases be substantially greater than doses to her fetus. However, doses to the offspring can exceed doses to the mother for a number of radionuclides. In particular, the requirements of skeletal development during fetal growth, particularly in late pregnancy, can lead to significant uptake of radioisotopes of phosphorus and of calcium and, to a lesser extent, other alkaline earth elements. Thus, offspring : adult dose ratios are up to factors of about 10 – 20 for isotopes of P and Ca and 2 – 6 for isotopes of Sr (Stather *et al*. 2003; ICRP, 2004). Uptake of radioisotopes of iodine by the fetal thyroid can also lead to greater doses to the fetus than to the mother late in pregnancy (dose ratios of up to about 3). Other radionuclides for which doses to the fetus can exceed doses to the mother include tritium as tritiated water, carbon-14 and sulphur-35. Offspring : adult dose ratios are greatest following ingestion or inhalation of soluble (Type F) forms. Offspring doses may also be of concern when the dose ratio is <1 since a dose of 1 mSv might be reached at otherwise acceptable levels of occupational dose (Phipps *et al* 2001).

When a worker has declared pregnancy, possible doses to her child will be taken into account in measures taken to limit exposures. Thus, the high dose ratios discussed above, applying to later pregnancy, may in practice be of less importance that doses occurring as a result of intakes before the declaration of pregnancy, as a result of either acute or chronic exposures. A number of radionuclides of potential significance in this category have been identified, including nickel-63 and iron-55 (Phipps *et al* 2001; Nosske and Karcher, 2003).

In general, doses to the infant from radionuclides ingested in breast-milk are estimated to be small in comparison with doses to the reference adult (ICRP, 2004). On the basis of the models developed in this report, it is only in the cases of tritiated water, $^{45}$Ca, $^{75}$Se and $^{131}$I that infant doses may exceed adult doses, by ratios of between 1 and 3, applying to maximum transfer occurring after maternal intakes by ingestion shortly after birth. Ratios of infant to reference adult doses are generally lower for intakes by inhalation than for ingestion. Comparisons with Publication 88 (ICRP, 2001) doses to the offspring due to in-utero exposures show that in most cases these are more important than doses that may result from breast feeding; exceptions include $^{60}$Co, $^{131}$I and $^{210}$Po.

In addition to the need to consider the assessment of doses to the offspring the biokinetics of radionuclides in the mother may also be different from the reference adult. Dose coefficients for female workers may also be modified during pregnancy or lactation and changes to biokinetics of radionuclides may need to be taken into account for dose assessment purposes.

# 3 MODELS TO REPRESENT THE INTAKE, TRANSFER, AND EXCRETION OF MATERIAL

## 3.1 Introduction

A knowledge of the behaviour of radioactive materials within the human body is essential for the interpretation of measurements of activity in the body or in excreta in terms of intake or committed effective dose or organ doses. This Chapter describes, in general terms, the routes of intake of radionuclides into the body, their subsequent transfers within the body and their distribution and retention. The uncertainties associated with the development of these models is also addressed. This report gives only an overview of the models. The reader is referred to the original publications (ICRP, 1989, 1993b, 1994a,b, 1995a,c, 2005b) for full details.

For intakes of radionuclides in the workplace a number of routes are possible including inhalation, ingestion and entry through intact skin and wounds. Fig. 3.1 summarises the routes of intake, internal transfers, and routes of excretion.

For inhalation, the Human Respiratory Tract Model (HRTM) (ICRP, 1994a) was applied in Publication 68 (ICRP 1994b) and in subsequent publications on dose coefficients. For these implementations of the HRTM, chemical forms of radionuclides that had been assigned to ICRP Publication 30 inhalation Classes D, W, and Y were assigned to HRTM absorption Types F, M, and S respectively. In the new series of publications on *Occupational Intakes of Radionuclides-Dose Assessment and Monitoring* (OIR) information is being reviewed on the lung clearance characteristics of different chemical forms of each element, within the framework of the HRTM. In some cases, (mainly compounds of actinide elements) material specific parameter values for absorption to blood will be given. Guidance on the application of material specific data in the HRTM is described in ICRP Guidance Document 3 (ICRP, 2002b).

For ingestion of radionuclides the new *Human Alimentary Tract Model* (HATM) (ICRP, 2006) will be applied. This is for application both to intakes by ingestion as well as to material cleared to the throat and swallowed after inhalation. In the HATM fractional absorption of radionuclide is specified by $f_A$ values instead of $f_1$ values as given for the gastrointestinal tract (GIT) model described in Publication 30 (ICRP, 1979).

ICRP has generally not given advice on assessing doses from intakes of radionuclides transferred from wounds sites to blood and other organs and tissues as this occurs as a result of an accident and is not subject to environmental controls in the workplace. Information on the transfer of radionuclides from wound sites has, however, being reviewed by a Scientific Committee of NCRP and these data have been used to develop a series of models to describe the transfer of material from wounds after intakes in different physico-chemical forms (NCRP 2007, in preparation). The NCRP model has been applied in the OIR series of publications to describe the transfer of radionuclides from wound sites and to the blood and lymphatic tissue.

A proportion of the intake of activity is absorbed to blood. Activity reaching body fluids (the transfer compartment) in this way is known as systemic material. The activity then undergoes various and often complex transfers which determine its distribution within the body, its retention and its route and rate of elimination. The distribution of systemic activity in the body can be diffuse and relatively homogeneous, e.g. with tritiated water, potassium and caesium, or localised in certain organs or tissues, e.g. with iodine (thyroid), alkaline earth metals (bone), plutonium (bone and liver). Biokinetic models are also used by ICRP to describe the distribution and excretion of the soluble component of material transferred to the blood and systemic circulation from the various sites of intake. The systemic models for the elements are presently being reviewed and

revised as necessary to take account of more recent information and in particular to provide models that are appropriate for both dosimetry and bioassay interpretation.

Removal of deposited material from the body occurs principally by urinary and faecal excretion although radionuclides may also be lost by exhalation or though the skin (e.g. $^{131}I$, $^{3}H_2O$). Urinary excretion is the removal in urine of material from the plasma and extracellular fluid. Faecal excretion has two components: systemic faecal excretion which represents removal of systemic material via the alimentary tract; and direct faecal excretion of the material passing unabsorbed through the alimentary tract after ingestion in the diet or clearance to the throat from the respiratory system after inhalation.

The models outlined in this Chapter can be used to calculate body or organ content and daily urinary or faecal excretion at specified times after intake. However, to calculate dose coefficients, ICRP assigned numerical values to a wide range of model parameters. Thus for inhalation, these include the size of the inhaled particles and the breathing rate of the subjects. These values are known as 'default' or 'reference' values, and are chosen to be typical, representative values. In any particular situation the actual values of many of the parameters may be different from the reference values. Usually, doses from intakes of radionuclides are low compared with the relevant dose limit or constraint, and the resulting difference between the reference value and the individual specific value is unimportant. As described in Section 2.5, however, there are circumstances in which it is feasible, and indeed desirable, to obtain better dose assessments by using information that is specific to the situation. The calculation of organ/tissue equivalent doses may also be necessary as well as effective dose.

Where there is prior knowledge of the likely physical and chemical form of the radionuclide in the working environment, the appropriate aerosol absorption parameter values for application in the HRTM and values for absorption in the alimentary tract for application in the HATM can also be selected. It should also be noted that there is scope for considerable individual variability in results due to differences in body mass, age, and other factors. If the biokinetic behaviour of any specific material is expected to differ significantly from that of the default biokinetic model employed in this report, and if doses are likely to approach dose limits, then consideration needs to be given to whether the model parameters should be modified to take account of the relevant data (see below). Chapter 8 gives detailed guidance on a structured approach to modifying model parameter values according to the situation.

The models for the major routes of intake by inhalation and ingestion are described below. For some radionuclides, it is also necessary to consider direct uptake from contamination on the skin or as a result of intakes through wounds.

## 3.2 Human Respiratory Tract Model (HRTM)

The Human Respiratory Tract Model (HRTM) described in Publication 66 (ICRP, 1994a), was applied to calculate inhalation dose coefficients for workers and members of the public in Publications 68, 71 and 72 (ICRP 1994b, 1995c, 1996), and bioassay functions in Publication 78 (ICRP, 1997b). The HRTM is designed to facilitate the application of material-specific parameter values for inhaled radionuclides as described in the Guidance Document on the Practical Application of the HRTM (ICRP, 2002b). In applying the HRTM in the accompanying series of OIR publications, some updating has been carried out as indicated below.

In the HRTM the respiratory tract is represented by five regions (Fig. 3.2). The extrathoracic (ET) airways are divided into $ET_1$, the anterior nasal passage, and $ET_2$, which consists of the posterior nasal passage, the pharynx and larynx. The thoracic regions are bronchial (BB: trachea and bronchi), bronchiolar (bb), and alveolar-

interstitial (AI: the gas exchange region). Lymphatic tissue is associated with the extrathoracic and thoracic airways ($LN_{ET}$ and $LN_{TH}$ respectively). For consistency with the HATM, the oral passage is not now included in $ET_2$ as it was originally. This does not affect results obtained with the model, because deposition in ET from air entering the mouth was taken to occur only in the larynx.

### 3.2.1 *Deposition*

The deposition model evaluates fractional deposition of an aerosol in each region, for all aerosol sizes of practical interest (0.6 nm – 100 µm). For the ET regions, measured deposition efficiencies are related to characteristic parameters of particle size and airflow, and are scaled by anatomical dimensions to predict deposition under other conditions (eg. gender, ethnic group). For the thoracic airways a theoretical model of gas transport and particle deposition is used to calculate particle deposition in each of the BB, bb, and AI regions, and to quantify the effects of the subject's lung size and breathing rate. To model particle deposition, the regions are treated as a series of filters, during both inhalation and exhalation. The efficiency of each is evaluated by considering aerodynamic (gravitational settling, inertial impaction) and thermodynamic (diffusion) processes acting competitively. Regional deposition fractions are calculated for aerosols having lognormal particle size distributions, with geometric standard deviations ($\sigma_g$) taken to be a function of the median particle diameter, increasing from a value of 1.0 at 0.6 nm to a value of 2.5 above about 1 µm *(*Publication 66, § 170). Deposition parameters are given for three reference levels of exertion for workers (sitting, light exercise, heavy exercise).

For inhalation of radionuclides by workers, the reference subject is taken to be a normal nose-breathing adult male at light work. For occupational exposure the default value now recommended for the Activity Median Aerodynamic Diameter (AMAD) is 5 µm (ICRP, 1994b). Fractional deposition in each region of the respiratory tract of the reference worker is given in Table 3.1 for aerosols of 5 µm AMAD.

### 3.2.2 *Clearance*

The model describes several routes of clearance from the respiratory tract (Fig. 3.3). Material deposited in $ET_1$ is removed by extrinsic means such as nose-blowing. In other regions clearance is competitive between the movement of particles towards the alimentary tract and lymph nodes (particle transport), and the absorption into blood of material from the particles in the respiratory tract. Removal rates due to particle transport and absorption to blood are taken to be independent.

It is assumed that particle transport rates are the same for all materials. A single compartment model is therefore provided to describe particle transport of all materials (Fig. 3.4). Reference values of rate constants were derived, so far as possible, from human studies, since particle transport rates are known to vary greatly among mammalian species. Fig. 3.4 as it stands would describe the retention and clearance of a material. However, as noted above, there is in general simultaneous absorption to blood.

Absorption depends on the physical and chemical form of the deposited material. It is assumed to occur at the same rate in all regions (including the lymph nodes) except $ET_1$, where it is assumed that none occurs. Absorption is a two-stage process: dissociation of the particles into material that can be absorbed into body fluids (dissolution); and absorption into body fluids of soluble material and of material dissociated from particles (uptake). The clearance rates associated with both stages can be time-dependent.

*Dissolution:* the HRTM uses a simple compartment model to represent time-dependent dissolution. It is assumed that a fraction ($f_r$) dissolves relatively rapidly,

at a rate $s_r$, and the remaining fraction $(1 - f_r)$ dissolves more slowly, at a rate $s_s$ (Fig. 3.5). A limitation of the system is that it can only represent an overall dissolution rate that decreases with time. To overcome this, Publication 66 also describes a more flexible equivalent system, described in terms of "particles in initial state" and "particles in transformed state" which was applied in earlier Publications (ICRP, 1994b, 1995c, 1997b). The additional flexibility is, however, rarely required in practice, and is more complex (less intuitive) to present. The simpler approach is therefore adopted now as the default, with the more flexible approach retained as an alternative.

*Uptake:* uptake to body fluids of dissolved material can usually be treated as instantaneous, as in Fig. 3.5. In some situations, however, a significant fraction of the dissolved material is absorbed slowly into body fluids because of binding to respiratory tract components. To enable this to be taken into account, the HRTM includes compartments in which activity is retained in each region in a "bound" state. However, it is assumed by default that uptake is instantaneous, and this is reflected in the reference values.

The system shown in Fig. 3.5 applies to each of the compartments in the particle transport model shown in Fig. 3.4 except $ET_1$ where no absorption occurs.

Material-specific rates of absorption can be used in the model for compounds for which reliable experimental data exist. Values are proposed in the OIR publications for selected compounds of some radionuclides. For other situations, default values of parameters are recommended, according to whether the absorption is considered to be fast (Type F), moderate (M) or slow (S). For gases or vapours instantaneous uptake to body fluids, Type V (very fast), may also be recommended. Reference values for each, given in Table 3.2, are now specified in terms of the parameters $f_r$, $s_r$ and $s_s$ shown in Fig. 3.5. The bound state is not invoked for the default values, i.e., uptake to blood is assumed to be instantaneous. Element-specific values of the rapid dissolution rate $s_r$ are recommended for some elements (such as uranium) in the OIR document.

These absorption rates, expressed as *approximate* half-times, and the corresponding amounts of material deposited in each region *that reach body fluids* can be summarised as follows:

*Type V*: 100% absorbed instantaneously. Regional deposition does not need to be assessed for such materials, because in dose calculations they can be treated as if they were injected directly into body fluids (eg. tritiated water, methyl iodide).

*Type F*: 100% absorbed with a half-time of 10 minutes. There is rapid absorption of almost all material deposited in BB, bb, and AI, and 50% of material deposited in $ET_2$. The other 50% of material deposited in $ET_2$ is cleared to the GI tract by particle transport (eg. most forms of caesium and lead).

*Type M:* 10% absorbed with a half-time of 10 minutes and 90% with a half-time of 140 d. There is rapid absorption of about 10% of the deposit in BB and bb; and 5% of material deposited in $ET_2$. About 70% of the deposit in AI eventually reaches body fluids (eg. iron oxide and hydroxide, plutonium nitrate).

*Type S:* 0.1% absorbed with a half-time of 10 minutes and 99.9% with a half-time of 7000 d. There is little absorption from ET, BB, or bb, and about 10% of the deposit in AI eventually reaches body fluids (eg. plutonium and uranium dioxides).

For absorption Types F, M, and S, all the material deposited in $ET_1$ is removed by extrinsic means. Most of the deposited material that is not absorbed is cleared to the alimentary tract by particle transport. The small amounts transferred to lymph nodes

continue to be absorbed into body fluids at the same rate as in the respiratory tract.

In the accompanying OIR publications, compounds of each element have been assigned to the HRTM default Absorption Types based on current information, and in some cases material specific absorption parameter values are proposed.

### *3.2.3 Gases and Vapours*

For radionuclides inhaled as particles (solid or liquid) the HRTM assumes that total and regional deposition in the respiratory tract is determined only by the size distribution of the aerosol particles. The situation is different for gases and vapours, for which deposition in the respiratory tract depends entirely on the chemical form. In this context, *deposition* refers to how much of the material in the inhaled air remains behind after exhalation. Almost all inhaled gas molecules contact airway surfaces, but usually return to the air unless they dissolve in, or react with, the surface lining. The fraction of an inhaled gas or vapour that is deposited in each region thus depends on its solubility and reactivity.

As for particulate forms of radionuclides, default parameter values are provided for use in the absence of more specific information. Thus the general defaults for gases and vapours are 100% total deposition in the respiratory tract (regional deposition: 20% $ET_2$, 10% BB, 20% bb and 50% AI) with Type F absorption. This classification is somewhat different from that recommended in Publication 66. In particular, no deposition in $ET_1$ is assumed by default.

In the accompanying OIR publications, guidance is given on the selection of parameter values for gaseous and vapour compounds of a number of elements, including hydrogen, carbon, sulphur and iodine. In each case, values are proposed for total deposition, regional deposition (usually in the same proportions as in the default pattern given above) and absorption rate (usually Type V or Type F).

## 3.3 Human Alimentary Tract Model (HATM)

The Publication 30 (ICRP, 1979) model of the gastrointestinal tract has been replaced by the Human Alimentary Tract Model (HATM) described in Publication 100 (ICRP, 2006). The structure of the HATM is shown in Fig 3.6. Its main features can be summarised as follows:

- Inclusion of all alimentary tract regions. Doses are calculated for the oral cavity, oesophagus, stomach, small intestine, right colon, left colon and rectosigmoid (the sigmoid colon and rectum). Colon doses are combined as a mass-weighted mean to include the right colon, left colon and rectosigmoid.
- Gender-dependent parameter values for adults for dimensions and transit times of contents through the regions (age-dependent parameter values are also specified for use in future calculations of doses to members of the public).
- Transit times for food and liquids, as well as for total diet, for the mouth, oesophagus and stomach.
- Default assumption that total fractional absorption, $f_A$, of an element and its radioisotopes to blood occurs in the small intestine, ie. $f_{SI} = f_A$. It is assumed there is no recycling from the wall to blood.
- Model structure to allow for absorption in other regions, where information is available.
- Model structure to allow for retention in the mucosal tissues of the walls of alimentary tract regions, and on teeth, where information is available.

- Explicit calculations of dose to target regions for cancer induction within each alimentary tract region, considering doses from radionuclides in the contents of the regions, and considering mucosal retention of radionuclides when this is taken into account.

The organs and fluids represented in Fig 3.6 by dashed boxes show connections between the HATM and the HRTM and systemic biokinetic models. First-order kinetics is assumed for all transfers in the HATM. This is a considerable simplification of the complex processes involved in transfer of material through the lumen of the alimentary tract but is expected to provide a reasonably accurate representation of the mean residence time of a radionuclide in each segment of the tract.

Mucus and associated materials cleared from the respiratory tract enter the oesophagus via the oropharynx. For ingested food and liquids, the HATM specifies two components of oesophageal transit representing relatively fast transfer of 90% (mean transit time of 7 seconds for total diet) of the swallowed material and relatively slow transit of the residual 10% (40 seconds for total diet). It is assumed that the slower oesophageal transit times apply to all material cleared from the respiratory tract.

The oral cavity and oesophagus will receive very low doses from radionuclides in transit because of their short transit times (ICRP, 2006). However, these regions were included for completeness, because a specific $w_T$ is assigned to the oesophagus (ICRP, 1991, 200x), and because retention in the mouth, on teeth for example, can result in a substantial increase in dose to the oral mucosa. In general, the alimentary tract regions of greatest importance in terms of doses and cancer risk are the stomach and particularly the colon. While the small intestine may receive greater doses than the stomach, it is not sensitive to radiation-induced cancer and is not assigned a specific $w_T$ value. Doses are calculated separately for the right colon, left colon and rectosigmoid. This partitioning of the colon for the purposes of dose calculations is predicated on the availability of transit time data. The rectum is taken to be part of the rectosigmoid, primarily because of difficulties in determining transit times separately. Mean transit times for the stomach and colon are about one-third greater in adult females than males. Slightly smaller masses in females (eg. 10% lower mass of colon tissue) will compound this gender difference.

The accompanying OIR reports specify values for the fractional absorption of elements and their radioisotopes from the alimentary tract to blood ($f_A$) after ingestion or inhalation. In most cases, the values of $f_A$ for ingestion will be the same as the $f_1$ values given previously for use with the Publication 30 model, since in most cases there is unlikely to be sufficient new information to warrant a revision in values. In addition, the general default assumption will be that absorption occurs solely from the small intestine, as in the Publication 30 model; that is, $f_{SI} = f_A$. However, the HATM allows absorption to be specified for other regions as well as the small intestine. As discussed in Publication 100 (ICRP 2006) for the example of isotopes of iodine doses to alimentary tract regions and other tissues will in many cases be insensitive to assumptions regarding the site of absorption.

For inhaled particles reaching the alimentary tract after escalation from the respiratory tract, it is appropriate to take account of solubility in the lungs in specifying $f_A$ values. For elements exhibiting a range in solubility according to their physicochemical form, there is evidence that the reduced solubility of Type M or S materials is also associated with reduced intestinal absorption. As discussed in Publication 71 (ICRP, 1995c), in many cases for which a single $f_A$ value is specified for ingestion of an element, this is taken to apply to inhaled Type F materials and lower default values of 0.1 and 0.01 are applied to Types M and S. However, because of the need for realism in estimates of absorption for application to bioassay interpretation, attempts have been made wherever possible to use available data to specify $f_A$ values for different forms

rather than rely on defaults.

Human and animal data suggesting or showing retention of ingested radionuclides in mucosal tissues of the walls of alimentary tract regions, principally the small intestine, can be used to specify retention in the HATM. The inability of the Publication 30 model to take account of such retention has long been regarded as an important deficiency. However, as illustrated by calculations for examples from the few cases for which quantitative information can be derived (e.g. Fe: ICRP, 2006), inclusion of retention may not result in large increases in doses to the small intestine and increases in committed effective doses are likely to be small.

An important development in the HATM is the methodology used to calculate doses in the various regions from non-penetrating alpha and electron radiations. Thus, while the Publication 30 approach was to assume that the dose to the wall was one half of that to contents of the region, with an additional factor of 0.01 included for alpha particles to allow for their short range, the HATM takes explicit account of the location of the target tissue in the mucosal layer of the wall of each region. The targets relating to cancer induction are taken in each case to be the epithelial stem cells, located in the basal layers of the stratified epithelia of the oral cavity and oesophagus and within the crypts that replenish the single cell layer epithelium of the stomach and small and large intestines.

As discussed in Publication 100 (ICRP 2006), the HATM generally results in substantially lower estimates of doses to the colon from beta-emitting radionuclides than obtained using the Publication 30 model. This is because the HATM takes explicit account of dose to the target region throughout the length of the colon, and of loss of energy in the colon contents and the mucosal tissue overlying the target stem cells (at a depth of 280 - 300 μm). This reduces energy deposition in the target tissue for electrons and results in zero dose in the target tissue from alpha particles. In the absence of retention of radionuclides in the alimentary tract wall, doses from ingested alpha emitters to all regions of the alimentary tract will be solely due to their absorption to blood and subsequent irradiation from systemic activity in soft tissues. For the stomach, the HATM and Publication 30 approaches give more similar estimates of doses from electron-emitting nuclides.

### 3.4 Absorption through Intact Skin

Generally, radionuclides do not cross the intact skin to any significant extent. Exceptions of practical importance are tritium oxide as liquid or vapour, organic carbon compounds and iodine as vapour or in solution. Tritiated water is the only case considered in the OIR publications.

There is no general model of entry of radionuclides through the skin because of the large variability of situations which may occur. Skin can become contaminated by contact with aerosols, liquids or surfaces contaminated with radionuclides. Clothing may be an important source of skin contamination and wet clothing may bring the contaminant into close contact with the skin thereby increasing the possibility of penetration through it. Many factors must be taken into account in dose assessment: the chemical form of the compound, the location and the surface of the contaminated area as well as the physiological state of the skin. Intact skin is a good barrier against entry of a substance into the body.

For skin contamination, both the radiation dose to the area of skin contaminated and the dose to the whole body as a result of absorption need to be considered. ICRP in Publication 60 (ICRP, 1991) has recommended that for skin contamination doses should be calculated to sensitive cells, assumed to be at a depth of 70 μm (as a reasonable average value). For deposited activity doses are to be calculated as an average to each

$cm^2$ of skin tissue. This applies to activity distributed over the skin surface or aggregated in particles. No specific models are recommended by ICRP for calculating doses from β particles deposited on the skin.

### 3.5 Wounds

To provide a means for calculating doses resulting from radionuclide-contaminated wounds, the National Council on Radiation Protection and Measurements, in collaboration with the ICRP, has developed a biokinetic and dosimetric model for exposure to radionuclides from contaminated wounds (NCRP, 2007 in preparation). The wound model (Fig. 3.7) was formulated and parameterized using experimental animal data due to the lack of adequate human information. The model can be used to calculate radiation doses to the wound site from deposited radionuclides, and, when coupled with an element-specific systemic biokinetic model, can be used to calculate committed doses to organs and predict urinary and fecal excretion patterns for bioassay interpretation.

The NCRP Wound Model (Fig. 3.7) was designed to predict the biokinetic behaviour of both soluble and insoluble radioactive materials, regardless of initial physical and chemical state. To do so, five compartments were designated to describe certain physical or chemical states of the radionuclide within the wound site. These comprise Soluble (S); Colloidal and Intermediate State (CIS), Particles; Aggregates and Bound State (PABS); Trapped Particles and Aggregates (TPA); and Fragments. In some cases, the compartments contain the radionuclide in its original physicochemical form. In others, the originally deposited material changes state and moves from one compartment to another with time. Although using five compartments to represent the wound site appears complex, in most cases the model simplifies to two or three compartments depending on the physical and chemical form of the radionuclide specified. This two- or three-compartment representation was shown to be widely consistent with the experimental data describing wound site retention (NCRP, 2007 in preparation).

Four categories of retention were defined for radionuclides present in a wound initially in soluble form: Weak, Moderate, Strong and Avid, which refer generally to the magnitude of persistent retention in the wound. The criteria for categorization were: 1) the amount retained 1 d after deposition and 2) the rate of clearance of the remainder.

Release of radionuclide from the wound site occurs via the blood for soluble materials and lymph nodes (LN) for particulates. Further solubilisation of particles in LN also provides radionuclide to the blood. The blood comprises the central compartment that links the wound model with the respective radioelement-specific systemic biokinetic model. Once the radionuclide reaches the blood, it behaves biokinetically as if it had been injected directly into blood, and in a soluble form. This is the same approach as is taken for the HRTM and HATM.

To illustrate the application of the model for bioassay interpretation, the wound model was coupled to the systemic biokinetic model for $^{137}Cs$ described in ICRP Publication 78 (1997). The principal default for Cs in the wound model is the Weak Category. Accordingly, the parameters for this category were applied to the wound model, and urine and faecal excretion patterns predicted (Fig. 3.8). The patterns show peak excretion of $^{137}Cs$ in urine at 2-3 days after intake, and for faeces at about 5 days. Both patterns reflect the rapid movement of $^{137}Cs$ from the wound site, and its distribution in and excretion from the systemic organ sites.

In comparison, if the $^{137}Cs$ in the contaminated wound site is assumed to be present in particles of irradiated power reactor fuel, then it can be given parameter values of the Particle Category. In this case, dissolution and absorption to blood are

much slower than for the Weak Category, and the urine and faecal excretion patterns exhibit a pseudo-equilibrium pattern after about 10 days, and lasting for several years (Fig. 3.9).

The presence of wounds, abrasions, burns or other pathological damage to the skin may greatly increase the ability of radioactive materials to reach subcutaneous tissues and thence the blood and systemic circulation. Although much of the material deposited at a wound site may be retained at the site, and can be surgically excised, soluble (transportable) material can be transferred to the blood and hence to other parts of the body. These events occur only as a result of accidents, each event will, therefore, be unique and need to be assessed by occupational health physicists and medical staff.

To date, ICRP has not given advice on the interpretation of wound monitoring data following accidents involving radionuclides as each incident will be unique and general advice cannot be given. The biokinetic models that have been developed for various radionuclides are, however, applicable to the soluble component of any deposit in cuts or wounds that enters the blood circulation. With the development by NCRP of the model outlined above together with the comprehensive literature review that it was based upon (NCRP, 2007 in preparation), it is appropriate for ICRP to give dose coefficients for intakes of radionuclides by direct entry into the blood. These dose coefficients can be used in conjunction with the NCRP wound model parameter values to obtain organ and tissue doses and effective dose for radionuclides that have entered the blood from the wound site. Examples of the use of the wound model will be given in Annex H (in preparation).

## 3.6 Biokinetic Models for Systemic Radionuclides

The fraction of an intake of a radionuclide entering the systemic circulation is referred to as the uptake. In Publication 30 ICRP reviewed information on the behaviour of radionuclides that had entered in the body. It recommended biokinetic models for radionuclides that had entered the blood for intakes by inhalation and ingestion. The ICRP 30 models applied specifically to workers and not to members of the public. More recently, Publications 56, 67, 69 and 71 revised the biokinetic models for selected radionuclides of 31 elements and these have been applied in the calculation of dose coefficients for both workers and for infants, children and adult members of the general population (ICRP 1989, 1993b, 1995a,c) (Table 1.1). The models given in these Publications were primarily developed to provide age-dependent dose coefficients but the models for adults were also used in Publication 68 (ICRP, 1994) which gave updated dose coefficients for workers and in Publication 78 (ICRP, 1997b) on the interpretation of bioassay data.

The biokinetic models recommended for adults are presently being reviewed and will be published in the OIR series of Publications. A key feature of the updated models is that they will be applicable for both the calculation of doses and the interpretation of bioassay data.

Radionuclides entering the blood may distribute throughout the body (e.g. $^{3}$H, $^{24}$Na, $^{42}$K, $^{137}$Cs); they may selectively deposit in a particular tissue (e.g. $^{131}$I in the thyroid; $^{90}$Sr in bone) or they may deposit in significant quantities in a number of tissues (e.g. $^{239}$Pu, $^{241}$Am, $^{144}$Ce). If a radionuclide that enters the blood is an isotope of an element that is required by the body then it will follow the normal metabolic pathways for that element (eg. $^{24}$Na, $^{32}$P, $^{42}$K, $^{45}$Ca, $^{59}$Fe). If it has similar chemical properties to an element that is normally present then it will tend to follow the biokinetic pathways of that element, although its rate of transfer between the various compartments in the body may be different (e.g. $^{90}$Sr and $^{226}$Ra behave similarly to Ca, $^{137}$Cs and $^{86}$Rb similarly to K). For other radionuclides their behaviour in the body will depend upon their affinity for biological ligands and other transport systems in the body and, as a result,

the extent of uptake and retention is largely unpredictable and must be assessed from the available human or animal data (eg. $^{95}$Nb, $^{106}$Ru, $^{239}$Pu, $^{241}$Am). Fig. 3.10 illustrates the structure of the systemic model for plutonium and americium.

This draft Guidance Document includes in an Appendix an example of updated reviews of biokinetic data and models covering Uranium (OIR-27).

### 3.7 Excretion Pathways

The biokinetic model adopted for the urinary bladder for members of the public and workers is described in Publication 67 (ICRP, 1993b) and Publication 68 (ICRP, l994b). Although the model was developed for dosimetry it has also been used to predict excretion. The number of voids per day is taken to be six. To represent the kinetics of the bladder in terms of first-order processes, the rate of elimination from the bladder is taken to be 12 $d^{-1}$. There is some degree of approximation in representing discrete events by a continuous process in this way. However, any inaccuracies introduced are likely to be small and will tend to cancel out when averaged over a daily measurement.

The activity present in the alimentary tract includes material which entered it by secretion from the systemic circulation into the upper large intestine as well as material cleared from the respiratory system to the throat or was directly ingested and not absorbed.

For bioassay interpretation it should be remembered that the transit time through the alimentary tract is subject to particularly large inter (and intra-) subject variations. Moreover, while for ease of computation transit through the alimentary tract is represented by a series of compartments that clear exponentially, in practice, the movement is more like “slug” flow. This means that the model will typically underestimate faecal clearance in the first day or two after entry to the alimentary tract and overestimate it in the next few days. It is therefore unlikely that individual daily faecal clearance measurements in the first few days after intake will strictly follow the predicted pattern, and so for assessment purposes it may be best to consider cumulative excretion over the first few days. If this is not practicable then correction for a nominal daily loss could be considered. This is considered further in Chapter 4.

The rate of loss of systemic activity from the body through the various routes of excretion is given explicitly in some of the biokinetic models. For the remainder, it is necessary to partition the excreted systemic activity between urine and faeces according to a constant ratio. In these circumstances, information on the urinary to faecal excretion ratio is given with the element specific models.

### 3.8 Medical Intervention

If medical intervention to prevent uptake or enhance excretion is considered, then any treatment will modify the biokinetic behaviour described by the models summarised above and detailed outlined in the OIR series of publications (NCRP, 1980; Gerber and Thomas, 1992; IAEA, 1996). In addition, when treatment has been administered the data provided cannot be used directly to assess committed effective doses from monitoring information. In such circumstances a programme of special monitoring (Section 5.5) should be undertaken to follow the retention of the particular contaminant in the person, and these data should be used to make a specific assessment of committed effective dose for that person.

### 3.9 Uncertainties in Biokinetic Models

The models developed by ICRP for describing the intake of radionuclides into the body, their transfer to the systemic circulation and subsequent distribution and retention

are largely based on human data and the results of studies in experimental animals. These models include the HRTM, HATM and the element specific systemic models to be given in the OIR series of publications. The more recent, physiologically based biokinetic models, as for example for the alkaline earths and actinides (ICRP, 1993b & 1995a) also take into account information about the age-dependent changes in anatomy and physiology. Biokinetic parameters in the models are mainly derived from:

- direct observations of the time distribution and excretion of the element in humans;
- information on the behaviour of the element in other mammalian species;
- the biokinetics of chemically similar elements in humans and animals; and
- the i*n vitro* behaviour of the element of interest (Tries, 2000).

This information is sometimes supplemented by considerations of mass balance and physiological data (Skrable et al, 2002). Where animal data are used there is some uncertainty in terms of the extent to which it can represent the behaviour of radionuclides in man. Ideally comparative data from a number of animal species are used as the basis for developing parameters for describing the transfer of elements between model compartments.

The model parameters developed by ICRP are given for a reference person and take into account judgements on the appropriate model and model parameters as the best estimate for a population. In general ICRP does not provide individual specific parameter values as these can vary substantially from person to person (see section 2.5). The biokinetic models referred to above and to be described in the OIR series of publications are the most recent models adopted by ICRP.

The reliability of the biokinetic models is associated with the uncertainties on the sources and the quality and completeness of data used in their derivation (Skrable et al, 2002). These uncertainties include the stochastic variability and the lack of knowledge about a single true value or a true but unknown distribution of values. These issues are described in NCRP commentary 15 (Tries, 2000). Papers by Leggett et al (1998, 2001, 2003), Likhtarev et al (2003), Harrison et al (2001) and Skrable et al, (2002) provide detailed discussion on the reliability of biokinetic models. Uncertainties in biokinetic and dosimetric models are also described in the Dosimetry Annex to the new Recommendations (ICRP, 2007).

While there are some elements for which extensive human data are available to develop reasonably reliable models, there are also many elements which have had to be based on much more limited data and for which the confidence in the model is relatively low.

Inhalation is the most important pathway for intake of radionuclides entering the human body as a result of occupational exposure. In spite of the major advance in the model structure for the respiratory tract there remain areas of uncertainty in the application of the HRTM (ICRP, 1994a). The default parameters for the respiratory tract model often do not represent a particular compound accurately and hence the move to use material specific rates of absorption, where reliable experimental data exists, in the calculation of dose coefficients given in the OIR series of publications. Advice on using material specific data is also given in ICRP Guidance Document 3, *Guide for the Practical Applications of the ICRP Human Respiratory Tract Model* (ICRP, 2002b).

For ingestion it is frequently the case that absorbed fractions are only available from animal studies and the data can be quite variable. For example, to determine the value for absorption in the alimentary tract for antimony, there are animal data available that lie within a range from less than 0.01 to 0.2, depending on the chemical form of the

element. An $f_1$ value of 0.1 was chosen by ICRP for ingested antimony (ICRP, 1995a), but clearly there is a large degree of uncertainty attached to this parameter.

Even in cases when extensive human data are available, there may still be much uncertainty in the estimated effective dose if the assumed biokinetic model does not consider all relevant components of the actual biokinetic behaviour of the radionuclide.

The whole body retention of caesium can be well described by the sum of two exponential functions with biological half-lives of 2 and 110 days, respectively, as was done by the ICRP in Publications 30 (ICRP, 1979) and 56 (ICRP, 1989). However, in general the data supporting this model were only collected for up to a few months after intake. Longer-term data obtained following the Goiânia incident (Melo et al 1997, 1998), however, indicated that there is an additional, small, long-term component (about 0.1% of initial systemic activity) with a biological half-time of about 500 days. This third component has little influence on the effective dose per unit intake, but for the interpretation of bioassay measurements at long times after an intake, it may influence the estimated intake of activity, and therefore the resulting effective dose, by an order of magnitude. In addition individual variation in the principle component of for $^{137}$Cs retention can vary between about 50 and 150 days (Rundo, 1964). The biokinetic model for caesium has been modified in the OIR document from that adopted in Publications 30 and 68 and takes into account this long term component of retention.

As well as the uncertainties in the standard biokinetic models discussed above, there are also uncertainties in the models which describe the energy deposition in the target regions. These include the use of simplifying assumptions about organ masses, sizes, and shapes, and the geometrical relationships between internal organs that are implicit in the use of computer phantoms. There are also limitations to the computational procedures for the calculation of specific absorbed fractions for penetrating radiations, and in the simplified assumptions about absorbed fractions in the bone and GI tract for non-penetrating radiations. The adoption of voxel phantoms based upon medical imaging data for dose calculations will help to significantly reduce some of the uncertainties implicit in the use of the MIRD mathematical phantom (see Section 3.11).

As discussed in Section 2.5, because effective dose is calculated using reference parameter values and applies to a reference person, no uncertainty is attached to the reference dose coefficients. However, as described above, there may be substantial uncertainties in the dose and risk to any individual, including those contributed by uncertainties in the intake, in biokinetic and dosimetric models and uncertainties in age- and gender- specific risk coefficients. For an individual-specific estimates of equivalent dose and risk to organs and tissues an estimate of uncertainty might be required. For example, in retrospective assessments when effective dose might exceed limits and in epidemiological studies.

### 3.10 Individual Variability in Biokinetic Parameters

As indicated earlier, the biokinetic and dosimetric models for internally deposited radionuclides are designed for a reference individual, i.e. for an individual representing average values for the group considered and generally based upon Reference Man. There are, however, considerable differences between individuals of such a group. Variations in anatomical and physiological factors influence the tissue concentration, distribution and excretion of radionuclides in the body and hence doses to individuals. There are, for example, differences in the genetic constitution, age, gender, breathing patterns, lung, renal, liver, alimentary tract and cardiovascular functions, pregnancy or lactation. Environmental factors, such as exercise, disease, stress, infection, smoking, alcohol intakes, dietary factors, barometric pressure, exposure to sunlight, may interact with biological factors, producing sizeable variations in the behaviour of radionuclides among individuals (Tries, 2000).

Thus, the iodine uptake by an individual's thyroid is largely dependent on the thyroid content of stable iodine, which is influenced by the amount of stable iodine in the individual's diet. In countries with low levels of iodine in typical foodstuffs a higher radioiodine uptake (about 50%) is observed rather than about 30% as given in the ICRP model (ICRP, 1989). Because this higher uptake is frequently correlated with a higher thyroid mass, this does not necessarily influence the thyroid dose appreciably and hence does not have a significant effect on the effective dose. However, this higher uptake by the thyroid can greatly influences the estimated intake and effective dose based on bioassay data.

There can be very large variations in the excreted activity from one day to another for the same individual, which cannot easily be interpreted by a biokinetic model. These variations will always introduce an easily identified source of uncertainty in a model to represent any particular individual. In addition, the nominal daily urinary excretion from reference man is 1.6 L, but this depends strongly on physiological and environmental conditions. For the interpretation of excretion measurements using the values of the measured value per unit intake, m(t) (see Section 6.3 (iii) ) given in the OIR publications, and with the exception of tritiated water, it is preferable to have a 24-hour sample. This cannot be assured in all cases, however, and the normalization of data to 24 hours excretion will be another source of uncertainty.

Faecal samples from individual voidings vary widely in mass, composition and transit time through the alimentary tract. In addition activity concentrations can be very difficult to interpret, since they contain materials cleared from the lung, systemic material excreted into the alimentary tract and material passing unabsorbed through the alimentary tract following ingestion. Each of these variables are sources of uncertainty, when using the values of m(t) for faecal excretion to derive the intake. Many bioassay monitoring programmes require a three days' sample collection to estimate the daily excretion rate. Often the complete 24 hours' excretion is not received or the worker cannot provide samples for a period of several days and normalization to the reference man excretion rate is required, introducing another source of uncertainty.

An individual specific analysis is only necessary for the few situations when the worker's dose approaches the dose limit. Even when specific analysis is conducted, the day-to-day variation, the behaviour of the individual in relation to environmental factors and the limitations of measurements will introduce uncertainties that are not easily quantified.

### 3.11 Uncertainties in Dosimetric Models

The dosimetric models are formulated to compute the physical quantity, mean absorbed dose, due to radiations incident on the body or emitted by nuclear

transformations of radionuclides present in the body. The absorbed dose is computed in target regions (organs, tissues, or regions of tissues) indicated to be radiosensitive. Radiation and tissue weighing factors are applied to the mean absorbed dose to define the protection quantities equivalent and effective dose. As the weighing factors are based on judgement, no uncertainties are associated with these factors. Thus the uncertainties in the protection quantities arise from the uncertainties in the mean absorbed dose. The physical and anatomical parameters contributing to uncertainties in the mean absorbed dose for internal emitters are:

- Energy and intensity of the nuclear and atomic radiations emitted by the radionuclide;
- Interaction coefficients of the emitted radiations in tissues;
- Elemental composition of the tissues of the body;
- Volume, shape, density of the organs of the body; and
- Spatial relationship of the source regions (regions containing the radionuclide) and the target regions (radiosensitive organs and tissues for which dose values are desired).

Limitations are present in the computational model representing the anatomy and in the numerical procedures used to calculate the energy absorbed in the target tissues. The magnitudes of these uncertainties vary with radiation type, the energy of the radiation, and the specific source-target pair. The adoption of computational phantoms based upon medical imaging data (often referred to as voxel phantoms) has significantly reduced the uncertainties associated with photon and neutron radiations. For source and target regions that can not be resolved in the medical image data; e.g., target regions of the airways and intestines and source regions in trabecular bone, uncertainties are associated with the computational models used to represent these regions. The anatomical models are static and thus do not address uncertainties in the spatial position of the organs due to breathing and posture other than reclining.

Reference values for the masses and elemental composition of the organs of the body have been defined in ICRP Publication 89 (ICRP, 2002a) and used in the reference computational models of the anatomy (computational phantoms) noted above. Individual variability in the anatomical parameters is discussed below.

The parameters of the dosimetric model contributing to uncertainties in the absorbed dose are those physical parameters associated with the nuclear transformation processes which determine the energy and intensity of the emitted radiation and parameters which govern the transport radiations in the body. Uncertainty less than 10% has been assigned to attenuation and absorption coefficients for photons with some what higher uncertainties ascribe to soft tissue stopping power values for alpha and electron particles. Improvement in the basic nuclear data has reduced the uncertainty in the physical half-life of the radionuclide and the branching fractions of its decay modes. The simplified procedures used in the dosimetric calculations to address the delayed beta and gamma radiations of spontaneous fission can contribute substantial uncertainties in the mean absorbed dose in some tissues.

Some uncertainties also arise in the manner in which the biokinetic models are implemented in the dosimetric calculations. The biokinetic models are presented as compartment models which in a dosimetric evaluation are further extended to include the kinetics of radioactive decay and ingrowth of radioactive daughter products. A number of numerical methods are capable of solving the set of potentially large (100s) of coupled differential “stiff” equations that describe the kinetics although frequently one has to balance the demands of numerical accuracy and computational time.

Compartment- model issues contributing to uncertainties in the mean absorbed dose include:

- Biokinetics of members of decay chain (independent or shared kinetics);
- Representation of 'Other' compartment in the dosimetry.

The behaviour of gaseous radioelements and radioiodines formed within the body by the decay of solid parent radionuclide are assigned kinetics independent of the parent radionuclide. Independent kinetics has also been used with some decay chains of heavy metals (e.g., thorium). However for most decay chains the kinetics applied to the daughters are those of the parent; i.e., so called shared kinetics. The uncertainties associated with the treatment of the kinetics of the decay chain can potentially be substantial when the daughter members emitted greater energy than the parent.

The dosimetric calculations must associate an anatomical region (source region) with each biokinetic compartment. Many biokinetic models partition the systemic activity among a few identified organs/tissues and include a compartment referred to as 'Other tissue' which represents the residual. The dosimetric procedure distributes the activity in the 'Other tissue' compartment uniformly among all tissues not explicitly noted in the model. Substantial uncertainty may be associated with the mean absorbed dose for tissues that are members of 'Other tissue.' Frequently 'Other tissue' includes tissues assigned an explicit tissue weighting factor. For example, breast tissue is rarely if ever explicitly noted in biokinetic models and thus its mean absorbed dose is often based on its membership in the 'Other tissue'.

# 4 METHODS OF INDIVIDUAL MONITORING

## 4.1 Introduction

The purpose of this Chapter is to describe briefly the main measurement techniques, their advantages and their limitations for individual monitoring. In most cases, assessment of intakes of radionuclides may be achieved by body activity measurements, excreta monitoring, air sampling with personal air samplers, or a combination of these techniques. The choice of measurement technique will be determined by a number of factors including the radiation emitted by the radionuclide; the likely radiation dose; the biokinetic behaviour of the contaminant and the availability of equipment. This is covered in Chapter 5.

Routine monitoring programmes usually involve only one type of measurement if adequate sensitivity can be achieved. For some radionuclides, only one measurement technique is feasible, e.g. urine monitoring for intakes of tritium. For radionuclides, such as plutonium isotopes, that present difficulties for both measurement and interpretation, a combination of techniques may have to be employed. If different methods of adequate sensitivity are available, the general order of preference in terms of accuracy of interpretation is:

- body activity measurements;
- excreta analysis;
- personal air sampling; and
- environmental measurements.

These techniques are, however, complementary and not mutually exclusive.

Results of monitoring of the working environment (area monitoring) may provide information that assists in interpreting the results of individual monitoring, e.g. information on particle size, chemical form and solubility, time of intake. The results of workplace monitoring for air contamination may sometimes be used to estimate individual intakes. However the interpretation of the results of measurements from air sampling in terms of intake is not simple and may be misleading. The most common form of representative sampling is by using static fixed air samplers at a number of selected locations intended to be reasonably representative of the breathing zone of the worker. When such a method is routinely used for quantitative determinations of intake, the appropriateness of the results should be assessed using a special monitoring programme, often involving personal air samplers.

Monitoring in relation to a particular task or event may often involve a combination of techniques so as to make the best possible evaluation of a novel or unusual situation. For example, a programme of both body activity and excreta measurements and, in some circumstances, personal air sampling may be used in combination with solubility studies on samples of airborne activity or source material from the workplace. Solubility studies will not necessarily give a good indication of the solubility characteristics of material in the lung but can provide valuable guidance for use in determining the most appropriate monitoring procedure (ICRP 2002). In some cases of suspected incidents, screening techniques (such as measuring nose blow samples or nasal smears) may be employed to give a preliminary estimate of the seriousness of the incident. In these cases the regional deposition in $ET_1$ in the nose (given in Table 3.2) can be used to confirm that an intake has occurred and to give a rough estimate of the intake.

## 4.2 *In Vivo* Measurements

The IAEA (1996) has given guidance on the direct measurement of body content of radionuclides. Advice has also been issued by ICRU (2002a). Direct measurement of body or organ content provides a quick and convenient estimate of activity in the body. It is feasible only for those radionuclides emitting radiation that can be detected outside the body. In principle, the technique can be used for radionuclides that emit: x or $\gamma$ radiation; positrons, since they can be detected by measurement of annihilation radiation; energetic $\beta$ particles that can be detected by measurement of bremsstrahlung (e.g. $^{90}$Y for $^{90}$Sr); and the $\alpha$-emitters such as $^{235}$U and $^{241}$Am that can be detected by measurement of their characteristic 186 keV and 60 keV $\gamma$ rays respectively, or $\alpha$-emitters such as $^{239}$Pu that can be detected by measurement of its characteristic 13, 17 and 20 keV x rays.

Many facilities for the measurement of radionuclides in the whole body or in regions of the body consist of one or a number of high efficiency detectors housed in well-shielded, low-background environments (IAEA, 1996). The geometrical configuration of the detectors is arranged to suit the purpose of the measurement, e.g. the determination of whole-body activity or of activity in a region of the body such as the thorax or the thyroid. The skull or knees may be used as a suitable site for measurement of radionuclides deposited in the skeleton and some radionuclides deposit preferentially in the liver where they can be detected.

Care must be taken to remove contamination on clothing or hair on the body surface before activity is measured. For routine measurements, determination of whole-body content is often adequate for radiological protection purposes. Total body activity will then consist of systemic activity and activity in the alimentary and respiratory tracts. However, in special investigations, or in interpretation of unusual measurements, it may be advantageous to determine the distribution within the body either by profile scanning or by analysis of the relative response of detectors placed at different positions along the body.

Commonly encountered fission and activation products, such as $^{131}$I, $^{137}$Cs and $^{60}$Co, can be detected with comparatively simple equipment at levels that are adequate for radiological protection purposes. Such simple equipment may consist of a single detector, viewing the whole body or a portion of the body, or, for iodine isotopes, a small detector placed close to the thyroid. The advantage of simple equipment is that it may be operated at the place of work, thereby avoiding the time required to visit a remote whole-body monitoring facility. Measurements may then be made more frequently so that any unusually large intake would be recognised soon after it had occurred and any necessary follow-up readily organised.

In contrast, high sensitivity techniques are needed for monitoring a few radionuclides at the levels that are required for protection purposes. Examples are the low energy photon emitters such as $^{210}$ Pb, $^{241}$Am and isotopes of Pu. In all situations when Pu isotopes are unaccompanied by $^{241}$Am they are not detectable at the levels required for radiation protection purposes. If $^{241}$Am is present then this can provide a valuable tracer for plutonium if the Pu: $^{241}$Am ratio is known.

Up to the mid-1990s most body activity measurement facilities, whether high-sensitivity or simple systems, used thallium-activated sodium iodide detectors. These have the advantage that crystals of large volume can be manufactured and so provide high efficiency for detection of $\gamma$-rays. Interpretation of a $\gamma$-ray energy spectrum obtained from a mixture of radionuclides may, however, raise some difficulties. The components of the spectrum can be resolved by a multiple linear regression analysis technique, but this requires previous calibration of the detection equipment with standard sources of the required radionuclides dispersed in a matrix in such a way as to

simulate the distribution and attenuation within the body. The increased availability of high-efficiency germanium detectors has lead to their increasing use, particularly in situations where workers may be exposed to mixtures of unknown $\gamma$-ray emitting radionuclides. The superior energy resolving power of these detectors simplifies the interpretation of spectra obtained from complex mixtures of radionuclides although calibration is still needed.

The activity present in a wound can be detected with conventional $\gamma$ detectors if the contaminant emits energetic $\gamma$-rays. In the case of contamination with $\alpha$-emitting radionuclides, detection is much more difficult since the low energy x-rays that follow the
$\alpha$-decay will be severely attenuated in tissue; this effect is more important the deeper the wound. It is often necessary to localise the active material and this requires a well-collimated detector. Wound monitors must have an energy discrimination capability if a good estimate is to be made of contamination with mixtures of radionuclides.

The technology on which *in vivo* measurement systems are based is well-established. Nevertheless, there have been a number of recent developments that offer the promise of improved capabilities. Development work is being carried out on the optimisation of the area and thickness of detectors, with particular emphasis on the use of large detector arrays. Room temperature semiconductor arrays utilising either silicon or the compound semiconductor CdZnTe offer the possibility that bulky liquid nitrogen or electrical cooling systems may no longer be necessary (Franck *et al*., 2000; Webb *et al*., 2000; Genicot, 2000; Genicot *et al*., 2003; Wahl *et al*., 2006).

Almost all laboratories continue to use physical phantoms such as the Bottle-Mannikin-ABsorption (BOMAB) or Lawrence Livermore thorax phantoms for activity calibrations, but this approach has significant limitations with respect to the body size, body shape, and radionuclide distribution that can be modelled. These limitations could in principle be overcome using the numerical calibration techniques which have been developed over recent years. Mathematical voxel phantoms are constructed using data from computed tomography (CT) or magnetic resonance imaging (MRI scans on real subjects). Monte-Carlo simulations are then used to model photon transport from the phantom and the detection of photons by a simulated detector (Franck *et al*., 2003; Hunt *et al*., 2003).

### 4.3 Analysis of Excreta and Other Biological Materials

In some cases, excreta monitoring may be the only measurement technique for those radionuclides which have no $\gamma$-ray emissions or which have only low energy photon emissions. Excreta monitoring programmes usually involve analysis of urine, although faecal analysis may also be required if the material is relatively insoluble. Other samples may be analysed for specific investigations. Examples are the use of nose blow or nasal smears as routine screening techniques. Blood can be sampled in the case of suspected high level contamination, although activity concentrations are generally difficult to relate to body content or intakes (see also IAEA, 1999b, 2000, 2004).

The collection of urine samples involves three considerations. Firstly, care must be taken to avoid adventitious contamination of the sample. Secondly, it is usually necessary to assess or estimate the total activity excreted in urine per unit time from measurements on the sample provided. For most routine analyses, a 24 h collection is preferred but, if this is not feasible, it must be recognised that smaller samples may not be representative. Where a 24h sample is not easily collected then the first morning voiding is preferable for analysis (IAEA, 2000). The total daily excretion of creatinine, produced as a metabolic product in muscle metabolism, may be less variable than the volume of fluid lost in urine, although some individuals may still exhibit wide daily variations. Measurements of creatinine concentration in urine has therefore been

commonly used to estimate 24 h excretion of radionuclides from urine samples collected over part of a day. Tritium is an exceptional case for which it is usual to take only a small sample and to relate the measured activity concentration to the concentration in body water. Thirdly, the volume required for analysis depends upon the sensitivity of the analytical technique. For some radionuclides, adequate sensitivity can be achieved only by analysis of several days' excreta (e.g see Duke, 1998)

The analysis of faecal samples for routine monitoring involves uncertainty in interpretation owing to daily fluctuations in faecal excretion. Ideally, therefore, collection should be over a period of several days. However, this may be difficult to achieve in practice and interpretation may need to be based on a single sample. Faecal monitoring is more often used in special investigations, particularly following a known or suspected intake by inhalation of moderately soluble, Type M or insoluble, Type S compounds. In these circumstances measurement of the quantity excreted daily may be useful in the evaluation of clearance from the lungs and in the estimation of intake. Early results may be useful in identifying exposed individuals.

Radionuclides that emit $\gamma$-rays may be determined in biological samples by direct measurement with scintillation or semiconductor detectors. Analysis of $\alpha$- and $\beta$-emitting radionuclides requires chemical separation followed by appropriate measurement techniques. Measurement of so-called total $\alpha$ or $\beta$ activity may occasionally be useful as a simple screening technique, but there is no single method that will determine accurately all the $\alpha$ and $\beta$ activity in the sample. The total activity technique may be used in routine monitoring situations where intakes are expected to be very low compared with annual limits. The results could not be interpreted quantitatively, but would be useful for providing confirmation of satisfactory working conditions, an unusual result indicating the need for further investigation which would include radiochemical analysis. Total activity measurements may also be useful following a known contamination event or to identify those samples that merit early attention. Measurements of total $\alpha$ or $\beta$ activity cannot generally be used in quantitative evaluations of intake or committed effective dose as the radionuclide composition is not known, as it must be when using dose coefficients to assess effective dose.

Measurement of activity in exhaled breath is a useful monitoring technique for some radionuclides such as $^{226}$Ra and $^{228}$Th since the decay chains of both these radionuclides include gases which may be exhaled. (Youngman *et al*., 1994; Sathyabama *et al*., 2005). It can also be used to monitor $^{14}CO_2$ formed *in vivo* from the metabolism of $^{14}$C-labelled compounds (Leide-Svegborn et al, 1999; Gunnarsson et al, 2003)

Increasing use is being made of mass spectrometric techniques for the analysis of excreta samples. Inductively Coupled Plasma - Mass Spectrometry (ICP-MS) can achieve much lower detection limits for long-lived radionuclides than is possible with alpha spectrometry. For example, for $^{238}$U and $^{232}$Th, detection limits are in the region of 5-10 μBq per sample (Hurtgen C and Cossonnet C, 2003). After radiochemical separation, where appropriate, measurement times are in the region of a few minutes, whereas an alpha spectrometry measurement typically takes several days. The more advanced mass spectrometric techniques such as multiple collector ICP-MS or sector field ICP-MS have the capability to detect very small changes in isotopic ratios and so can detect small amounts of depleted or enriched uranium in urine samples containing natural uranium (Parrish *et al*, 2006). The more complicated technique of accelerator mass spectrometry (AMS) can be used to measure $^{14}$C in small samples, mg-size, with low activities down to ~0.1mBq/ml with high accuracy (<2%). The AMS technique may need to be considered for routine monitoring in situations where exposure to [$^{14}$C]-compounds that deposit extensively in adipose tissue, or other tissues with very slow metabolic turnover, biological half-times of 60->150 d, is likely, since it must be expected that the elimination of $^{14}$C from such tissues will occur by metabolism to [$^{14}$C]$CO_2$ and exhalation

in the breath.

### 4.4 Environmental Monitoring

A Personal Air Sampler (PAS) is a portable device specifically designed for the estimation of intake by an individual worker from a measurement of concentration of activity in air in the breathing zone of the worker. A sampling head containing a filter is worn on the upper torso close to the breathing zone. Air is drawn through the filter by a calibrated air pump carried by the worker. Ideally, sampling rates would be similar to typical breathing rates for a worker (~1.2 $m^3$ $h^{-1}$). However, sampling rates of current devices are only about 1/10 of this value. The activity on the filter may be measured at the end of the sampling period to give an indication of any abnormally high exposures. The filters can then be retained, bulked over a longer period, and the activity determined by radiochemical separation and high sensitivity measurement techniques. An estimate of intake during the sampling period can be made by multiplying the measured average air concentration by the air volume estimated to be inhaled by the worker during the period of intake. The difficulties in assessing intakes from PAS measurements were considered by Whicker (2004). Breathing zone measurements can vary significantly as affected by measurement conditions such as orientation of the sampler with respect to source, on which lapel (right or left) the sampler is worn, design of the air sampling head, particle size, local air velocities and directions, and sharp gradients in and around the breathing zone of workers.

There are thus three important requirements for a PAS device:

- First, the sampler should collect sufficient material for the activity corresponding to a significant intake to be measurable in a reasonable counting time. This will depend mainly on the lowest committed effective dose that the PAS is required to detect. Typically, in a routine monitoring programme, the requirement will be to detect annual intakes that, in total, give rise to committed effective doses greater than 1 mSv.
- Second, the volume of air aspirated by the sampler should be sufficient to provide a statistically accurate representation of the activity concentration in the breathing zone of the worker. PAS monitoring is most often used for radionuclides such as plutonium, for which a very small number of particles may contain activities that would correspond to a significant intake. The statistics of sampling small numbers of events then becomes the critical factor in determining sampling accuracy.
- Third, the particle collection characteristics of the sampler should be known. These depend on the aspiration efficiency of the sampling head and the collection efficiency of the filter. The aspiration efficiency is the ratio of the particle concentration in the air entering the sampler to that in the ambient air. It is usually close to unity for particles of aerodynamic diameter less than about 1 $\mu$m, but the inertia of larger particles will give a tendency to under- or over-sample according to conditions. Similar effects apply to particles entering the nose and mouth and are taken into account in the ICRP Human Respiratory Tract Model (ICRP, 1994b). The aspiration efficiency of the respiratory tract is termed inhalability.

A PAS does not usually provide information on particle size. Nevertheless, it is important either to determine the particle size distribution of the inspirable material or to make realistic assumptions about it, since it can have a marked effect on deposition fractions in the respiratory tract, and hence on dose estimates. This is particularly important as the recommended default AMAD of 5 $\mu$m is intended to be realistic rather than conservative in terms of dose estimation (Dorrian and Bailey, 1995, Ansoborlo et al, 1997). All samplers are size selective to a greater or lesser extent, under- or over-

sampling at particular particle sizes, and this can result in errors in intake estimation. The aspiration efficiency of a PAS should therefore be determined to indicate whether corrections are necessary. An investigation of the aspiration efficiency of a PAS gave values close to unity up to an aerodynamic diameter of 30 $\mu$m under workplace conditions (Mark et al, 1986). It has been suggested that samplers should be designed to collect the inspirable fraction rather than the total aerosol (Vincent and Armbruster, 1981). Use of such samplers would be acceptable, but would require modification of analysis procedures, since the ICRP Respiratory Tract Model implicitly assumes that the *total* aerosol concentration is known (ICRP, 1994a and 2002b – Annex B).

A large and extensive PAS monitoring programme has been ongoing since 1986 at the BNFL Sellafield nuclear fuel reprocessing site in the United Kingdom. This programme was designed to address the acknowledged limitations at the time of their urinalysis-based bioassay programme for Pu internal dosimetry (Strong and Jones, 1989). In the first year of usage, 500,000 PAS were worn by 3500 regular users of PASs, with < 0.1% of the resulting estimates of intake exceeding 10% of the ALI for plutonium (alpha) for a single work period of 8 h. In analysing results from 228 assessments occurring during 1986, for which there were suspected single acute exposures to various actinide mixtures greater than 0.1 ALI, Strong and Jones (1989) concluded that the protracted use of PAS does provide a useful indicator of general environmental airborne contamination levels and a convenient means of identifying particular tasks or particular working methods which apparently give rise to localised enhanced levels of airborne contamination. However, no clear relationship was evident between significant PAS results and the evidence from biological sampling. This illustrates the limitations in using PASs to assess individual inhalation intakes and exposures.

A uranium exposure study was also conducted by Eckerman and Kerr (1999) to determine the correlation between uranium intakes predicted by PASs and intakes predicted by bioassay at the Y12 plant in Oak Ridge, USA. This study concluded that there was poor correlation between the two measurements.

Britcher and Strong (1994) extended the experience in using PAS as part of the internal dosimetry monitoring programmes for the Calder Hall reactors and the Sellafield nuclear fuel reprocessing facility. Looking principally at workers from the Magnox plant, in which low-level chronic exposures were most common, they concluded that over prolonged periods of time (several years) there was no significant bias between intake assessments from PAS, or SAS (using a PAS:SAS ratio of 10, see below), and from biological samples. It was concluded that samplers can therefore be used to obtain satisfactory estimates of intake for groups of workers. However, for individuals, the correlation between assessments using PAS and biological samples was poor and the authors cast doubt on the adequacy of PAS for estimating annual intakes of individual employees at the levels of exposure encountered in operational environments. The authors also questioned whether, for environmental monitoring, PAS offered any advantages over static air sampling programmes. The same lack of correlation between PAS and bioassay sample-based intake estimates was also seen for known acute exposures (Britcher *et al*. 1998).

Snapp et al (2004) summarised the use of PASs at the Y12 site in the USA. The occupational health staff at the plant make use of the PAS data in the following ways:

- as an early indicator or workplace incidents to trigger special bioassays that include faecal sampling, urinalysis, and lung counting;
- as a tool to indicate the potential for acute exposures superimposed on top of chronic exposures when modelling bioassay date; and
- to identify potential areas with increased airborne activity through review and trending of the applicable PAS data.

Static air samplers (SAS) are commonly used to monitor workplace conditions, but can underestimate concentrations in air in the breathing zone of a worker. Marshall and Stevens (1980) reported that PAS:SAS air concentration ratios can vary from less than 1 up to 50, depending on the nature of the work. Britcher and Strong (1993/1994) concluded from their review of monitoring data for Magnox plant workers that intakes assessed from PAS data were about an order of magnitude greater than those implied by SAS data. Nevertheless, if SAS devices are sited appropriately, a comparison of PAS and SAS measurements can be used to define a fixed PAS:SAS air concentration ratio which can be used in the interpretation of SAS measurements for dose assessment purposes. It should, however, be recognised that the use of SAS is a relatively indirect method for assessing doses, and use of the results to estimate individual dose requires a careful assessment of exposure conditions and working practices. Apart from their potential use for dose estimation, SAS devices can also provide useful information on radionuclide composition, and on particle size if used with a size analyser such as a cascade impactor.

Overall, however, experience of the use of PASs and SASs indicates that body activity measurements and/or excreta analysis are to be preferred for the assessment of individual intakes of airborne radionuclides and doses.

# 5 MONITORING PROGRAMMES

## 5.1 Introduction

The design and management of monitoring programmes was considered in ICRP Publication 78 (1997b). It was recommended that the emphasis in any particular monitoring programme should be on the formal assessment of doses to those workers who are considered likely to receive routinely a significant fraction of the relevant dose limit or who work in areas where exposures could be significant in the event of an accident.

The results of workplace monitoring should give an indication of the likelihood of doses from intakes exceeding 1 mSv a year. In such circumstances it is likely that individual monitoring will be required. As described in Section 2.4, experience has shown that workers involved in various operations would normally require individual monitoring.

The use of individual monitoring for workers whose doses could exceed 1 mSv is common practice in many organisations although it may not be required by legislation. The 1 mSv criterion is not achievable by individual monitoring in some cases, e.g. exposures to some physico-chemical forms of $^{239}$Pu or other actinides when assessed from urine monitoring. In such situations a combination of monitoring techniques, including measurements on the working environment, may be needed.

For workers who are not routinely employed in areas that are designated in relation to the control of airborne contamination and who are unlikely to have significant intakes of radionuclides, routine monitoring of the workplace will usually be sufficient to provide assurance that intakes are adequately controlled. Other types of monitoring programmes may also be needed depending on the circumstances of exposure as described below.

## 5.2 General Principles for the Design of Individual Monitoring Programmes

Four categories of monitoring programme can be defined: routine, special, confirmatory and task-related:

*Routine monitoring* is performed where intakes could arise as a result of an essentially continuous risk of contamination of the workplace from normal operations, or where undetected accidental intakes could occur.

*Special monitoring* is performed after actual or suspected abnormal events.

*Confirmatory monitoring* is carried out to demonstrate that working conditions are satisfactory, and that there is no need for routine individual monitoring. It could consist of occasional individual monitoring measurements.

*Task-related monitoring* is carried out to provide information about a particular operation.

A specification for an individual monitoring programme includes the monitoring method (or methods) to be employed (measurement of activity in the body, in excreta samples, and in air), the measurement technique used (eg photon spectrometry, alpha spectrometry, mass spectrometry), monitoring intervals for routine monitoring, and measurement or sample collection times for special monitoring.

It is important to consider the monitoring programme design as an integral part of the overall radiation protection programme as well as the dose assessment process. An appropriately-designed monitoring programme should provide the data necessary to enable a dose assessment to meet the specified need; even the most sophisticated dose assessment calculations cannot, however, compensate for inadequate monitoring data.

Guidelines for dose assessment should, therefore, always allow for additional monitoring data to be requested where this is feasible and appropriate for improving the dose assessment.

Many factors need to be taken into consideration when designing an individual monitoring programme. These include the purpose of the monitoring (e.g. whether it is carried out to demonstrate compliance with legislative requirements, or simply to confirm that doses are very low), local factors such as the number of workers to be monitored and the availability of particular measurement methods, and economic factors. The main factors that determine the dosimetric performance of the monitoring programme relate to the characteristics of the material to which a worker may potentially be exposed (normally by inhalation). These are:

- the radiation emitted by the radionuclide and its progeny
- the half-life of the radionuclide
- the respiratory tract deposition characteristics of the aerosol
- the respiratory tract and alimentary tract absorption characteristics of the material
- the retention in the body or the excretion rate from the body as a function of the time between intake and measurement
- any preferential deposition in particular body organs and tissues after systemic uptake, and subsequent retention in those organs
- any significant differences between the biokinetic behaviour of a parent radionuclide and its progeny
- the excretion pathway (i.e. urine, faeces)
- the technical feasibility of the measurement

The dosimetric performance of the monitoring programme may be assessed by considering the effect of these factors on the accuracy of assessed doses and on the sensitivity associated with the monitoring programme, which can be quantified in terms of the assessed minimum detectable dose (Carbaugh, 2003; Etherington *et al*, 2004a, 2004b).

A simple example of the use of the minimum detectable dose concept is shown in Fig. 5.1. This shows the $^{241}$Am activity that would be measured in the lungs following an intake of a Type M $^{241}$Am compound resulting in a committed effective dose, E(50), of 20 mSv. A typical detection limit for direct measurement of $^{241}$Am in the lungs is 10 Bq, and the figure shows that an intake giving rise to an E(50) of 20 mSv could be reliably detected up to about 160 d post exposure.

The CD-ROM that will accompany the OIR series of publications will give Figures of content per unit dose as illustrated in the OIR-27 Uranium Appendix. These can be used to demonstrate the period of time after the intake for which an intake of a radionuclide that will give a dose of 1 mSv can be detected.

Specific advice on the design of individual monitoring programmes for a range of radionuclides and compounds was developed in the EC 5$^{th}$ Framework Programme project OMINEX (Etherington *et al*., 2004). Radionuclides considered were tritium, $^{60}$Co, radioiodine, $^{137}$Cs, uranium, plutonium and thorium; the materials considered were those most commonly encountered in the nuclear industries.

### 5.2.1 Direct Methods of Monitoring

Direct (*in vivo*) bioassay is likely to be the monitoring method of choice if the radionuclide is a high yield, high energy gamma-ray emitter, unless the material is excreted rapidly from the body. The gamma-radiation emitted by such radionuclides is strongly penetrating, and so is readily detected using scintillation or semiconductor detectors positioned close to the body. If the material is absorbed rapidly from the respiratory tract, and is then either distributed uniformly in body tissues (e.g. $^{137}$Cs in most common chemical forms), or is distributed preferentially among a number of organs, (e.g. $^{55}$Fe) then whole body monitoring should be chosen. If the material deposits preferentially in a single organ such as the thyroid (e.g. $^{125}$I,$^{131}$I), then partial body monitoring of the relevant organ should be chosen. In the case of materials that are absorbed less rapidly from the respiratory tract (e.g. insoluble forms of $^{60}$Co oxide), lung monitoring is preferable to whole body monitoring soon after the intake, as it gives a more accurate measure of lung deposition and retention than a whole body measurement.

Direct bioassay is also useful for some radionuclides that emit photons (X- or gamma-rays) at lower energies and/or with lower yields (e.g. $^{241}$Am, $^{210}$Pb, $^{144}$Ce). However, in the extreme case of radionuclides that mainly emit X-rays below 25 keV with low yields (notably, the alpha-emitting isotopes of plutonium) direct bioassay cannot achieve the sensitivity required for radiological protection purposes.

Special considerations apply when direct bioassay measurements of radioactive progeny are used to determine the body content of the parent radionuclide. Significant errors can arise if it is assumed that the progeny are always in secular equilibrium. For example, the activity of $^{232}$Th in the lungs can be underestimated when determined from direct measurements of its $^{228}$Ac, $^{212}$Pb, $^{212}$Bi and $^{208}$Tl progeny, because of the more rapid absorption of these radionuclides from lungs to blood. For the same reason, activity of $^{232}$Th in the lungs can be underestimated when determined from measurements of $^{220}$Rn in breath.

If direct bioassay monitoring is available and meets the dosimetric requirements for a particular monitoring application, it should be chosen in preference to indirect (*in vitro*) bioassay. The information provided by measurements of retention in organs of the body or in the whole body is closely related to the information required to determine absorbed dose to organs, particularly if measurements can be made over extended periods of time. The reason for this is that integration over time of a curve describing activity in an organ gives the total number of disintegrations in the organ during that period, which is proportional to the absorbed dose in the organ. While biokinetic models are still required in order to determine initial intake, distribution among organs, and the time course of retention after measurements are discontinued, a significant part of the information required for a dose calculation can be provided by the measurement data set.

### 5.2.2 Indirect Methods of Monitoring

Indirect bioassay measurements are not directly related to retention or to the number of disintegrations in organs or whole body. Biokinetic models are generally needed to determine this relationship, and the modelling can be subject to significant uncertainties. This generally leads to significantly larger uncertainties in dose estimates derived from indirect bioassay measurements as compared to direct bioassay.

Nevertheless, indirect bioassay methods do have important areas of application. For radionuclides that do not emit penetrating radiation with sufficient yields, (e.g. the pure beta-emitter $^{90}$Sr, the alpha-emitting isotopes of plutonium), indirect bioassay usually has to be chosen as the primary monitoring method.

Urine monitoring provides a measure of systemic uptake to organs and tissues,

and so is most useful for materials that are absorbed relatively rapidly from the respiratory tract (that is, materials that have absorption characteristics within the range defined by default Types F and M) or gastrointestinal tract. It can also be used to determine the fraction of activity deposited in a wound site that transfers to the systemic circulation. When used in conjunction with direct bioassay, monitoring data can also be used to derive or refine estimates of biokinetic model parameter values, particularly values for the absorption parameters $f_r$, $s_r$, and $s_s$ in the HRTM.

Caution needs to be exercised in using urine monitoring for materials that are absorbed relatively slowly from the respiratory tract (i.e. 'insoluble' materials). In these circumstances, it is usually the lung dose that makes the greatest contribution to effective dose, and uncertainties or lack of knowledge of the absorption characteristics of the material can then result in significant errors in assessed dose.

For insoluble materials, significant improvements in sensitivity can be achieved by using faecal monitoring in preference to, or in addition to, urine monitoring. This is because significant fractions of insoluble material deposited in both the extrathoracic airways and the lungs are cleared *via* the gastro-intestinal tract to faeces. Interpretation of faecal monitoring data needs to take account of a number of factors that are specific to the faecal excretion pathway. Excretion of faeces is a discrete process (even though it is usually modeled using first-order kinetics), and so it is advisable to sum the amounts excreted over a 3-day period to obtain a daily excretion rate. Although faecal excretion rates after an acute inhalation are typically highest in the 12 - 72 hour period post-intake, they may be subject to the highest levels of intra- and inter-subject variability during this interval, making it advisable to extend the period for special monitoring to later times. Another important consideration that applies to both faecal and urine monitoring is that many of the materials for which indirect bioassay is useful are naturally-occurring (e.g. uranium and thorium oxides). For such materials, it is necessary to quantify and take account of natural background excretion levels and their variability.

### 5.2.3 Workplace Monitoring

For some actinide compounds, there are circumstances where individual monitoring by direct or indirect bioassay cannot reliably quantify doses below a few mSv. This could be the case where the biokinetic behaviour of the material results in urine excretion rates that are so low that alpha spectrometry does not have adequate sensitivity, more sensitive mass spectrometric techniques for urine measurements are not available, and faecal monitoring is not feasible. It could also arise where background excretion levels of naturally occurring radionuclides require that recording levels are set well above the detection limit of the measurement. In such circumstances, workplace monitoring may be useful for triggering bioassay measurements as well as being employed for the purpose of individual dose assessment. If it replaces individual bioassay measurements, cautious assumptions should be made about exposure conditions and material characteristics, and the results should be checked using a confirmatory monitoring programme that could consist of periodic personal air sampling and/or individual bioassay monitoring carried out at suitable intervals.

## 5.3 Routine Monitoring

The required frequency of measurements in a routine monitoring programme depends upon the retention and excretion of the radionuclide and the sensitivity of the measurement techniques available. Selection of monitoring intervals should also take into account the probability of occurrence of an intake; where the risk of intake is high, the frequency of monitoring may need to be increased. The measurement technique should be selected so that uncertainties in the measured value are small in relation to the major sources of uncertainty. Two general approaches are available when making decisions on the choice of monitoring method and interval.

The approach adopted in ICRP Publication 78 (ICRP, 1997) defines a simple rule that limits the possible error on the estimate of intake arising from the unknown time of exposure. Monitoring intervals are selected so that any underestimation introduced by the unknown time of intake is no more than a factor of three. In practice, this is a maximum underestimate because the actual distribution of the exposure in time is unknown. The error in assessed intake can take on both positive and negative values, depending on the probability distribution of the exposure over the monitoring interval, with the result that the mean value of any underestimate is less than a factor of three. However, if a substantial part of the intake occurs just before sampling or measurement, the intake could be overestimated by more than a factor of three. This may be particularly important in the case of excreta monitoring, since the fraction excreted each day may change rapidly with time in the period immediately following the intake. If an unexpectedly high result is found in a routine monitoring programme, it would be appropriate to repeat the sampling or measurement a few days later, and adjust the estimate of intake accordingly. If appropriate and if convenient, the sample could be collected or the measurement made after a period of non-exposure, for example after a weekend or holiday.

In practice the intake is often taken to occur at the mid-point of the sampling interval. The approach was used in ICRP Publication 78 and for consistency is also adopted in the OIR series of publications. It has been reported by Puncher et al that this can lead to a biased estimate of intake (Punchner et al 2006) and the assumption of a "Constant Chronic" method has been proposed which has been shown to give unbiased estimates of a workers intake. However this approach differs very little from the mid-point method (see Annex F).

A shortcoming of these methods is that no account is taken of uncertainties other than in the time of intake. These additional sources of uncertainty include lack of knowledge of, or variability in, absorption parameter values and particle size. Recent work (Etherington *et al*., 2004) has used Monte-Carlo simulation techniques to evaluate uncertainties in intakes and doses taking account of uncertainties in these factors. An alternative, graphical approach has been developed by Stradling *et al.* (2004). Here the criterion employed is based on a consideration of minimum detectable dose, rather than uncertainty in assessed dose. The method involves the generation of curves showing values of the measured quantity (lung retention, urine excretion, etc.) per unit effective dose (e.g. 1 mSv) for a range of elapsed times between measurement and intake of a particular radionuclide / compound. These curves are generated using different combinations of values of parameters that have a strong influence on effective dose, such as inhaled particle size and the HRTM absorption parameters $f_r$, $s_r$, and $s_s$. Parameter value ranges are chosen to be appropriate for the particular exposure scenario. For each elapsed time, the minimum and maximum values of the measured quantity per unit dose are chosen from the set of curves generated, and the corresponding curves are plotted. These curves are labelled "1" and "2" in the example shown in Fig. 5.2. If an individual monitoring measurement lies on or below the lower curve (i.e. at position A' or A), then the dose can be reported with high confidence as being below 1 mSv. If the measurement lies on or above the upper curve (ie B' or C),

then the dose can be reported with high confidence as being above 1 mSv. Other dose levels, D (mSv), can be employed by simple scaling. Information on the minimum detectable amount for a particular measurement technique can then be used to determine a monitoring interval appropriate for the dose level of interest. The information on content per unit dose can be used in this way see (Appendix on Uranium).

The choice of approach depends on the purpose of monitoring. If a quantitative assessment of dose is required, then the mid-point method is appropriate. On the other hand, if the main purpose is to demonstrate that a dose is below a certain recording level (or investigation level), then the method of Stradling et al (2004) has the advantage that it avoids the need for a detailed dose assessment for the majority of cases where doses are below the relevant level.

### 5.4 Confirmatory Monitoring

One method of confirming that working conditions are satisfactory (typically for doses less than 1 mSv $y^{-1}$) is to carry out occasional individual monitoring. Unexpected findings would give grounds for further investigation. Confirmatory monitoring of this type is most useful for those radionuclides that are retained in the body for long periods, and occasional measurements provide a check on the lack of build-up of the activity within the body.

### 5.5 Special or Task-Related Monitoring

Since both special and task-related monitoring relate to distinct events, either real or suspected, one of the problems encountered in interpretation of routine monitoring results does not apply, viz. the time of intake is known. Furthermore, there may be more specific information about the physical and chemical form of the contaminant. If therapeutic procedures have been applied to enhance the rate of elimination of a radionuclide from the body then special monitoring may be needed to follow its retention in the body and to provide the basis for a dose assessment.

A possible approach to optimising the design of such a monitoring programme is to assess how different choices for the number and time period of measurements affect uncertainties in assessed dose. Where assessed doses could be significant, there is much to be gained from using a combination of different monitoring methods (e.g. lung, urine and faecal monitoring), since they provide complementary information. For instance, direct bioassay measurements provide information on deposition and retention in organs, urine measurements can provide a measure of systemic uptake to organs, while early faecal measurements can provide a measure of the amount initially deposited in the respiratory tract for materials that are absorbed relatively slowly.

In cases where treatment has been given as for example Prussian Blue to enhance the faecal elimination of radioisotopes of caesium, or chelating agents such as DTPA to enhance the rate of urinary excretion of some actinides, care must be taken in assessing intakes from the rate of loss in faeces or urine respectively .

### 5.6 Investigation Levels

In many situations of potential exposure to radionuclides, it is convenient to set investigation levels for the quantities that are measured in monitoring programmes, i.e. whole body content, organ content, daily urinary or faecal excretion, activity concentration in air. The chosen value for the investigation level may be directly related to the dose limit or constraint, or to the intake. The data given for each radionuclide in the OIR Publications can be used to calculate the value of the measured quantity that corresponds to the chosen level of annual intake or committed dose for the appropriate

monitoring programme. For example, an investigation could be based on a reference level that would correspond to an intake of a radionuclides that would give a committed dose of 1 mSv (or 1/20$^{th}$ of an ALI). Thus in a routine monitoring programme for a single radionuclide and with a period of T days, an investigation level could be based on the body content that would give a committed dose of 1 mSv. The value corresponding to the investigation level can be obtained directly from the relevant graphs or calculated from the tables of dose per unit content in the data sets given in the OIR Publications. The use of constraints as described in the new Recommendations (ICRP, 2007) could be used as a basis for setting investigation levels. In setting such investigation levels, due attention must be given to other sources of exposure, i.e. other radionuclides and external irradiation. In situations where intakes and doses are known to be low and there is considerable experience of the processes being undertaken, it may be possible simply to set investigation levels for the measured quantities on the basis of experience. A measurement result in excess of the investigation level would indicate a departure from normal and the need to investigate further.

If the investigation level is set at a level corresponding to a very low dose and intake, a measurement result less than the investigation level may require no action other than to record the fact that a measurement was made and the result was less than the investigation level. If, however, the investigation level corresponds to a significant fraction of the annual dose limit, measured values should be interpreted in terms of intake or dose.

The nature of any investigation will depend upon the circumstances and the extent to which the investigation level is exceeded. The following should be considered:

- repeated measurements to confirm or refine the initial evaluation;
- the use of additional monitoring techniques;
- review of the working conditions and the circumstances of the exposure;
- if default parameter values were used in the original assessment, investigation of the particle size and chemical form of the actual contaminant and selection of more appropriate values, if necessary; and
- in cases of substantial intakes, removal of the contaminated person from work with radioactive materials and investigation of the actual retention and excretion characteristics, in order to refine the dose assessment.

## 5.7 Record Keeping and Reporting

Dose record keeping is the making and keeping of individual dose records for radiation workers. It is an essential part of the process of monitoring the exposures of individuals to both external radiation and to intakes of radionuclides radiation and for demonstrating compliance with dose limits and constraints. Formal procedures need to be established for record keeping and these have been described in publications by the IAEA (IAEA 1999b, 2004). The procedures and criteria for reporting individual and workplace monitoring results should also be clearly specified by the management or regulatory authority. Information reported should be clearly identifiable and understandable. For normal operations only final results are reported. In accident situations interim information will be needed to judge the need for management actions and the need for follow-up monitoring.

## 5.8 Wound Monitoring

Following a cut or wound radioactive material may penetrate to subcutaneous tissue and thence be taken up by body fluids and distribute around the body (Section 3.5). Depending upon the radionuclide(s) and the amount of activity it may be necessary to undertake a medical investigation and a programme of special monitoring. In these circumstances, the amount of radioactive material at the site of the wound should be determined taking into account self-attenuation of the radiation in the foreign material and in tissue, as an aid to decisions on the need for excision. If an attempt is made to remove material from the wound, measurements should be made of the activity recovered and remaining at the wound site, so as to maintain an activity balance. Subsequently, a series of measurements should also be made to determine any uptake to body tissues from which the committed effective dose can be calculated. These measurements may consist of *in vivo* measurements, or urine or faecal excreta monitoring, as appropriate for the particular radionuclides. If whole-body measurements are made, it may be necessary to shield any activity remaining at the wound site. Further information on the behaviour of radionuclides at wound sites and their medical management is given in the NCRP report on the behaviour of radionuclides at wound sites (NCRP, 2007 in preparation).

# 6 DATA TO BE PROVIDED FOR ELEMENTS AND RADIONUCLIDES

## 6.1 Introduction

For the assessment of doses to workers exposed to radionuclides ICRP is planning a series of publications on Occupational Intakes of Radionuclides-Dose Assessment and Monitoring (OIR). These will be based upon updated biokinetic and dosimetric models, as described in Chapter 3 and take into account the new Recommendations by the Main Commission (ICRP 2007) and recent updates from Committee 2 on biokinetic and dosimetric models. The series of OIR publications will provide, for each element, relevant biokinetic data and the parameter values for the biokinetic models adopted together with data for interpreting bioassay measurements from individual monitoring. They will thus contain updates of information that was previously given in Publications 30 and 68 (ICRP 1979, 1980a,b and 1988b) as well as in Publications 54 and 78 (ICRP 1988a, 1997).

In addition to updates of the information for the assessment of bioassay data previously provided in Publications 54 and 78, information will also be included on doses per unit content of radionuclide activity in the whole body, in selected organs or in excreta. It is the view of Committee 2 that in many cases these tabulations should facilitate the interpretation of measurement data for dose assessment purposes.

**An Appendix, OIR-27 gives an example for uranium.**

The information planned to be given in the OIR series of publications is summarised below. Additional information will be provided on CD-ROM as described in Annex A.

## 6.2 Dosimetric and Biokinetic Data

For each element the following information is to be provided:

### *(i) Chemical forms in the workplace*

Summary information on chemical forms likely to be found in the workplace.

### *(ii) Physical data*

The half-life, mode of decay for radionuclides with half-lives greater than 10 minutes. Energy and intensity of emitted radiations that can be used for individual monitoring

### *(iii) Intakes*

Information on the behaviour of various chemical forms of the element after entry into the body by inhalation or ingestion. Data on entry into the body through the intact skin or through wounds is provided where data are available. For inhalation, lung absorption Types and material specific data where they have been developed are tabulated for application in the HRTM. For ingestion, absorbed fractions, $f_A$ for the HATM are given.

### *(iv) Biokinetic models for systemic activity*

The information available on the behaviour of radionuclides after entry into the blood is reviewed. The biokinetic model adopted is described, including information on excretion pathways. Information on any significant gender related differences in biokinetic behaviour of radionuclides and the behaviour of decay products is included.

Where appropriate, information is given on the effect of medical treatment to reduce uptake or to enhance excretion as this can influence dose assessment.

### *(v) Dose coefficients*

Committed effective dose coefficients, $e(50)$, for inhalation of a 5 μm AMAD aerosol, ingestion, and direct transfer to body fluids are given.

For inhalation, dose coefficients are given for appropriate default Types (F, M and S) using the HRTM (ICRP 1994a). Where there are sufficient supporting data material specific dose coefficients are also provided. The aerosols are assumed to be log-normally distributed with an AMAD of 5 μm and geometric standard deviation $\sigma_g$ of approximately 2.5 (ICRP, 1994*, Paragraph 170*) inhaled by a Reference Worker at Light Work. They are assumed to have density 3.00 g $cm^{-3}$, and shape factor 1.5 (ICRP, 1994*, Paragraph 181*).

For ingestion, dose coefficients are calculated using the HATM (ICRP 2006) using the specified values of $f_A$.

For direct entry into the blood the systemic model is used.

Further information will be given on an accompanying CD-ROM. Thus will include:

- Organ and Tissue specific dose coefficients, committed doses to 7 d, 30 d, 1 and 10 years as well as 50 years. This illustrates the build up of dose with time;
- Dose coefficients for isotopes not given in text; and
- Dose coefficients for additional aerosol sizes.

## 6.3 Interpretation of Monitoring Data

The information to be provided for the interpretation of measurement data will be updates of that given in Publications 54 and 78 together with additional data related to the calculation of doses per unit content (see Annex D). This is provided to facilitate the interpretation of measurement data as described in Chapter 1.

### *(i) Principal emissions and detection limits*

Methods of individual monitoring with typical detection limits for the principal emissions that can readily be achieved. Comments on preferred measurement techniques and the adequacy of the detection limits are given where appropriate. For those radionuclides that present difficulties for measurement and interpretation, a later section gives some information on monitoring programmes, with a selected bibliography;

### *(ii) Critical monitoring quantity*

This Guidance Document gives advice on the evaluation of low exposures from routine measurements. Level 0 (Chapter 7) refers to cases where it is expected that the annual dose (committed effective dose from intakes of radionuclides that occur in the accounting year) is likely to be below 0.1 mSv ($10^{-4}$ Sv). If a measurement, $M$ is below the critical monitoring quantity $M_c$, then the annual dose is likely to be below 0.1 mSv, even if there have been similar intakes in each and every monitoring interval during the year. The determination of $M_c$ is detailed in Section 7.2.

### *(iii) Tissue retention and excretion*

Predicted values of activity in tissues or excreta as a function of time after a single intake are tabulated. For single intakes by inhalation, ingestion, or direct uptake to blood, the predicted values of the measured parameter (body content, organ content, or daily excretion) are given for various times after the intake, i.e. $m(t)$ where $m$ is the measured quantity at time $t$ days after intake of 1 Bq. In these tabulations, body or organ content for day 1 means the content at the end of day 1 etc; for excreted activities, the value at day 1 represents the activity excreted during the first day after intake, corrected for radioactive decay to the end of day 1. *These values apply to either special monitoring or routine monitoring.* In the context of *in vivo* measurements, the following definitions are relevant. Whole-body content is the sum of systemic material (including that in the urinary bladder) and material retained within the respiratory and alimentary tracts. The content of the lung is taken to be the sum of the contents of the thoracic lymph nodes and the bronchial, bronchiolar, and alveolar-interstitial regions. The content of the skeleton is taken to be the content of the bone compartment in simple models and to be the sum of all compartments of both the cortical and trabecular bone and bone marrow in the more complex models of iron, uranium, the alkaline earth metals, and actinides (ICRP 1997).

### *(iv) Figures of tissue retention and excretion*

Graphs of the predicted activity of the radionuclide in selected body tissues, urine (daily excretion) or faeces (daily excretion), at various times after the intake, are given for an acute intake of the radionuclide which corresponds to a committed effective dose of 1 mSv. Figures are given for intakes by inhalation, ingestion, and uptake to blood. The values are for use in the assessment of the initial intake of the radionuclide and hence for the determination of the committed dose from the tabulated values of dose coefficients.

These values can also be used to facilitate decisions about monitoring programmes and the extent of the assessment required, as described in Chapter 5. They will also be of value in determining the requirement for medical treatment in the case of accidental exposure.

### *(v) Figures of measured quantities per unit intake*

Graphs of predicted values of measured quantities (body content, organ content, daily urinary or faecal excretion per Bq intake) are also given as a function of time following a single intake by inhalation, ingestion, wounds, and direct transfer to blood. Data are given in the form of fractional activity related to the intake, i.e. Bq per Bq intake for retention and daily excretion. One exception to this is for intake of tritiated water where data are given in Bq $l^{-1}$ per Bq intake since this is directly related to the dose rate. Data are given for time periods up to $10^4$ days after intake or until the fractional activity is less than $10^{-8}$ of the intake.

### *(vi) Choice of monitoring intervals*

For each radionuclide, the monitoring periods have been selected (as in Publication 78, paragraph 91) so that any underestimation introduced by an unknown time of intake is no more than a factor of three. The frequency of monitoring, determined from the models that have been applied, is determined both by the behaviour of the radionuclide in the body and its physical half-time. Within any workplace it should also be determined by the probability of an intake occurring.

For each monitoring interval, T, the predicted value of the measured quantity is the predicted value at the end of the monitoring interval assuming a constant chronic

intake (1Bq) spread throughout the monitoring interval. For routine monitoring, only intake by inhalation is considered.

In OIR-27 on Uranium information is also given on the assumption of a chronic intake over the period of the exposure.

### *(vii) Continuous intakes*

In Publications 54 and 78 information was tabulated on the build-up of activity levels in the body, in organs, or in excreta, for the hypothetical case of continuous daily intake by inhalation for unit intake (1/365 Bq per day) and for intake leading to a committed effective dose at the annual dose limit of 20 mSv. These data cannot be used directly to evaluate intakes and doses in any particular monitoring period. They were provided to enable an appreciation to be gained of the sensitivity of various monitoring techniques in relation to the quantities predicted at annual dose limits. They could also be used in situations where workers are unlikely to be exposed to significant intakes of radionuclides; comparison of the results of occasional screening measurements with these data may provide reassurance that intakes are indeed low. As these data are not seen as useful for interpreting monitoring information they are not included in the OIR series of publications. If needed, they can be calculated from the detailed information on activity concentrations in organs and tissues following acute intakes that are given on the accompanying CD-ROM.

### *(viii) Dose per unit content*

Values of dose per unit content have been provided to allow a more straightforward assessment of committed dose from bioassay measurements without the need to first determine the intake (Section 1.3 and Annex D). For measurements of activity in body tissues and excreta predicted values of committed effective dose are tabulated at various times after radionuclide intake following inhalation, ingestion, entry through wounds or uptake to blood. The use of these tables ensures that current ICRP models are used at all stages in the dose assessment process.

## 6.4 Quality Assurance

The Commission attaches particular importance to the question of quality assurance. The Task Group of Committee 2 on Dose Calculations has arranged for the quantities given to be calculated independently at different laboratories, using different computer codes. Any discrepancies in these calculations were investigated and resolved before publication.

## 6.5 Example for Uranium Provided in Appendix for February 2007 Consultation Draft

The Appendix included in this Consultation Draft gives an illustrative example of the section from the OIR series of publications on uranium (OIR-27). The tables are illustrative and not complete.

For intakes by inhalation the human respiratory tract model (HRTM) is applied to calculate particle deposition and respiratory tract clearance of the deposited particles (ICRP, 1994a). For intakes of different chemical forms of the elements they are either designated as an inhalation Type or, where sufficient data are available, material specific data are given.

For ingestion the human alimentary tract model (HATM) has not yet been implemented and the gastrointestinal tract model given in Publication 30 (ICRP, 1979) was applied. Values for the amount absorbed from the alimentary tract, $f_1$ (which will be

replaced by $f_A$ values in the HATM) are used for materials that are either ingested or passed to the alimentary tract after inhalation. The HATM will be applied in the final publication.

For elements/radionuclides that have entered the systemic circulation biokinetic models are given. These continue to be developed and it is possible that the information provided for U could be modified before final publication.

# 7 GENERAL ASPECTS OF DATA ASSESSMENT

## 7.1 Introduction

For routine operations, where doses are likely to be small, the generalised biokinetic and dosimetric models that have been recommended by ICRP are usually sufficient to provide a basis for the estimation of intakes and the determination of doses. When operations could result in doses approaching regulatory limits or in the case of accidents there may be a need for dose estimates that are more specific to the individuals involved and the exposure situation, both for dose record keeping and for the effective management of those exposed.

The Chapter examines the need to understand the exposure situation and radionuclide(s) being handled as well as their physico-chemical form, their retention characteristics in the body, the anatomical and physiological characteristics of the individual(s) involved and whether any treatment. It also stresses the need for any assessment to be proportionate to the expected exposure. It discusses the requirements for an effective monitoring programme and summarises approaches to data handling for single or multiple measurements.

The effort needed for the evaluation of monitoring data for intakes of radionuclides should broadly correspond to the anticipated level of exposure. A Working Party of Committee 2 that was set up to consider what further advice might be given to occupational health professionals proposed that guidance should be targeted at three levels of use, depending upon the potential doses incurred by workers, the ease or difficulty of interpretation and the expertise in dose assessment required. This division took into account the consideration that it is not always necessary to assess doses using the same level of effort or expertise (Fry et al 2003). Level 1 covered situations were doses and intakes are expected to be low (around 1 mSv $y^{-1}$) and standard default parameters are appropriate. At level 2 doses might be a few mSv $y^{-1}$ and specific information from the workplace should be used. Level 3 is really for the expert user and most of the time would be for exceptional situations such as for exposures near dose limits and after accidents. Committee 2 and the Tasks Groups that developed this Guidance Document have accepted this proposal and included a fourth Level 0 where no dose evaluation is needed when doses are expected to be less than 0.1 mSv $y^{-1}$. This structured approach to dose assessment is described below.

## 7.2 Levels of Task

The effort needed for the evaluation of monitoring data for internal exposure from intakes of radionuclides should correspond to the anticipated level of exposure in the particular facility area or group of workers. The expected annual dose to workers (committed effective dose from intakes of radionuclides that occur during the accounting year) assessed prospectively may be used as a quantitative criterion for planning the scope of the procedures needed for individual monitoring and interpretation of monitoring data. A European project in the EC 5$^{th}$ Framework programme was established set up to give guidance on internal dose assessments from monitoring data (Project IDEAS). The project developed a structured approach to the interpretation of monitoring data (Doerfel et al, 2002; Doerfel, 2005) and developed the proposals by the ICRP Working Party on Dose assessment (Fry et al, 2003). The structured approach was widely discussed through open consultation on the IDEAS web site. It is considered that this approach can be of general value for dose assessment purposes and its key features are described below. More detailed information is given in Chapter 8. It is stressed that this is not intended as a prescriptive process but is a guide to ensuring that appropriate dose assessments are obtained. The structured approach is given in terms of levels of task that can be chosen depending upon the circumstances of any exposure. It would be

appropriate for the occupational health physicist to consider whether circumstances have changed that would require any re appraisal of the level on annual basis.

With respect to operational radiation protection the following structure of "Levels of Task" are proposed:

- Level 0: Potential intakes that could result in an annual dose (committed effective dose from intakes of radionuclides that occur in the accounting year) <0.1 mSv. No evaluation of dose needed.
- Level 1: Simple, "reference" evaluation, with ICRP defaults used for all parameter values, except where there is better *a priori* information available, e.g. for inhalation intakes information on the particle size distribution (committed dose from the intake typically 0.1 – 1 mSv). At Level 1 the user will be generally concerned with radionuclides that are straightforward to measure, e.g. high-energy gamma emitters, that can be measured at levels of activity that would correspond to small intakes and doses and for which there are unlikely to be real problems of data handling.
- Level 2: Sophisticated evaluation generally using additional information from the workplace to give a more realistic assessment of dose. Level 2 users might be concerned with radionuclides that are difficult to measure at levels that would correspond to small doses. Examples are isotopes of uranium, thorium and plutonium and $^{241}$Am with for inhalation intakes the AMAD and absorption Type likely to be inhaled. Level 2 might also be used for an accidental intake. Comparisons would be made of the model predictions ("the fit") with the data, to choose between alternative parameter values, or to find optimum parameter values (*a posteriori*). At this Level, the parameters adjusted typically relate to the material, and the time of intake if unknown (committed dose from the intake typically 1 – 6 mSv).
- Level 3: More detailed evaluation for the expert, which applies to cases where there are comprehensive data available, as would be the case for exposures near the dose limit and probably relating to an accident. The evaluation is an extension of Level 2, typically to parameters relating to the subject (e.g. for inhalation intakes the HRTM particle transport rates). The fundamental approach at this Level is to adjust the model parameter values systematically, in a specific order ("step-by-step" approach), until the goodness of fit is acceptable (i.e. the fits obtained to all the data are not rejected by the specified criteria) (committed dose from the intake typically > 6 mSv). If any parameter values in the ICRP models are changed from the defaults the contributions of organ/tissue equivalent doses to effective dose will be changed and at higher doses may need to be recorded, depending upon national regulations.

Level 0 is the lowest level and it refers to cases where the effective annual dose would be most likely below 0.1 mSv ($10^{-4}$ Sv), even if there should be similar intakes in each monitoring interval of the year. At this level there is *generally* no need to evaluate the measured values explicitly, and the effective dose can be set to zero in analogy to the rounding of doses in external dosimetry. However, the measured value should be recorded with respect to further assessments in the future.

According to the above definition a measured quantity M can be allocated to Level 0, if

$$M \le M_C = \frac{10^{-4} \cdot m(T/2)}{e(50)} \cdot \frac{T}{365} \qquad (7.1)$$

with

$M_c$ "critical" monitoring quantity for Level 1 (Bq)

$T$ monitoring interval for the monitoring quantity considered (d)

$M(T/2)$ corresponding measurement or excretion function for the monitoring quantity at time $t$ = T/2 (Bq per Bq intake). It is assumed that the intake occurs at the mid-point of the monitoring interval.

$e(50)$ effective dose coefficient (Sv $Bq^{-1}$)

Values of the critical monitoring quantity $M_c$, defined by the equation above, are given in the OIR documents, as described in Section 6.3. Table 7.1 gives illustrative values of $M_c$ for some selected radionuclides. $M_c$ is typically above, or close to, the lower limit of detection (LLD) for the fission and activation products whereas it is below the LLD for the actinides considered. So in the case of the actinides, any significant monitoring value is likely to result in a dose of more than 0.1 mSv and thus has to be evaluated. In the case of the fission and activation products, however, there might be significant monitoring values, which result in a dose of less than 0.1 mSv. Thus, Level 1 applies typically to those radionuclides, which are easy to measure and which have low effective dose coefficients (i.e. e.g. $^{3}H$, $^{137}Cs$).

Note that there is growing interest in the application of the "dose per unit content" function, $z(t) = e(50)/ m(t)$, which represents the committed effective dose per unit organ (body) radionuclide content or per unit radionuclide content in the 24-hour excreta sample at time $t$ after an acute intake (Section 1.1). Thus $E(50) = M z(t)$, where $E(50)$ is the committed effective dose, and $M$ is the measured value. Its use simplifies the dose evaluation to a single step, instead of the traditional method of first applying the retention or excretion function $m(t)$ to calculate the intake, and then the dose coefficient $e(50)$ to calculate the resulting effective dose. Hence in equation (7.1), $m(chronic)/ e(50)$ could be replaced by 1/z(t') where t' is defined such that m(t')=m*(chronic)*.

### 7.3 Understanding Exposure Situations

Workplace information should be gathered in order to understand the exposure situations, e.g., radionuclides that may have been incorporated (including equilibrium assumptions for the natural series), chemical form, presumed particle size (typically 1 or 5 μm), likely time, pattern and pathway of any intake.

If no special information is available, the following default parameter values could be used:

- Mode of intake: Single intake
- Time of intake: Intake is delivered at a constant chronic rate throughout the monitoring interval, where the monitoring interval is defined as being between the date of the measurement being considered and the date of either the

Inhalation:

- Absorption Type and $f_A$ value: defaults according to OIR publications.
- Particle size: 5 μm AMAD

Ingestion:

- $f_A$ value: defaults according to OIR.

In the case of exposures that lead to effective dose estimates higher than about 0.1 mSv (i.e. above Level 0, see Section 7.2), it is desirable to use parameter values in the calculation of tissue and organ equivalent dose that are more specific to the conditions of exposure and to the individual. By using such workplace specific parameters a more realistic dose assessment can be obtained.

For the interpretation of direct and indirect measurements in terms of the intake and resulting effective dose, data on the time pattern and pathway of intake, the chemical and physical form of the radionuclides and on previous intakes are needed. In many cases however the information may not be available.

The time pattern is a main source of uncertainty in the interpretation of bioassay data. Assumptions about the time of intake and of whether the intake was acute, lasted for a short period of time or extended for a long time is a major point in the reliability of the interpretation of the bioassay data. For example, in some cases the retention and excretion functions diminish by orders of magnitude within a few days, therefore the choice of the time pattern of intake can influence the assessed dose within the same range.

Inhalation is the main pathway of intake in the workplace. The characterisation of the intake in terms of aerosol size and absorption type are needed for the application of the *m(t)* values to estimate the intake. The *m(t)* values are the calculated values of the measured quantities for unit intake at time t after the intake. The aerosol size will influence deposition in the HRTM and as a consequence the transfer of unabsorbed particles to the GI tract. In some working environments more than one particle size is detected. Values of m(t) are given for a range of aerosol sizes in the OIR document and accompanying CD. The rate of absorption of a radionuclide to blood is very important for interpreting bioassay data. It is a critical parameter in interpreting urine excretion data. The differences between the true absorption rates and the default parameters which have been assigned to the compound being inhaled is a source of errors that can be very large, specially when deriving intakes from urinary excretion bioassay data.

Further uncertainty is added when the activity of a radionuclide in the body could not be measured directly but is derived from progeny radionuclides.

Contributions from intakes from natural sources, especially in the diet, may also contribute to the uncertainty of a bioassay result.

### *7.3.1 Knowledge of Radionuclides Handled*

For many elements there are a number of radionuclides that could be present in the workplace with quite different physical characteristics. At the same time their behaviour after entry into the body can also be very variable depending upon the physical and chemical form present. Some examples are given below for uranium and plutonium to illustrate the potential for exposure to complex mixtures.

*Uranium:* Tables 7.2-7.5 show the composition of natural, enriched (3.5% and 92.8%) and depleted uranium in terms of activity. Note that the composition in terms of mass is completely different. The specific activities of the different uranium isotopes are given in Table 7.6 and the mass. Appendix OIR-27, Table 2.7.2 also illustrates the diversity and complexity of uranium compounds found in the workplace.

*Plutonium:* The Pu composition of some materials encountered in the nuclear industry are given in Tables 7.7 and 7.8 which show the composition of Pu and Am radionuclides in the reprocessing of spent fuel (Pu-nitrate, Pu oxides) and fuel from a light water reactor (LWR). Again there are widely different chemical characteristics and composition (Hurtgen, 2004, Puncher, 2004).

### *7.3.2 Time(s) and Pattern of Intake*

A principal source of uncertainty in the interpretation of bioassay data is the determination of the time of intake. In general, the time will not be known beforehand, especially in the case of routine monitoring when it is required to estimate an intake from a measurement made at the end of a monitoring interval. If an unusual occurrence triggered special bioassay monitoring, then the time of that occurrence is usually taken as the time of intake.

Since the bioassay function that gives the predicted measurement depends on the time since the intake it follows that the estimate of intake will vary, depending on when it is assumed the intake took place. If the time of the intake is known then the assessment is straightforward. However, if the time of intake is unknown then a judgement has to be made when it occurred. In Publications 54 and 78 (ICRP 1988a, 1997b) it is argued that in the absence of any information, the time of intake is equally likely to have occurred before the mid-point of the monitoring interval, than after it, and therefore suggests that in these situations, a value of $t$=T/*2* should be used, i.e. the intake is assumed to have occurred at the mid pint of the monitoring interval.

If a significant intake and effective dose is calculated, using this assumption, then a more realistic determination may required. Sometimes a review of workplace monitoring data, such as airborne or surface contamination levels can indicate a likely time for the intake to have occurred. Similarly, if other workers in the same workplace have exhibited positive routine bioassay samples, a review of the data and monitoring schedules for the individual workers may help determine the time of intake for all. Of course, an individual worker may be able to recall the incident that led to the intake. In addition, if several bioassay results are available, perhaps including different types of measurement, a comparison of these results with the m(t) tables may help in narrowing the choice of the time the intake occurred.

While the mid point assumption is a pragmatic and simple approach and for consistency with previous advice in publications 54 and 78 is also applied in the OIR series of publications, it does not result in an unbiased estimate of the intake. Appendix F demonstrates that if this assumption is applied regularly, to an individual worker, it is subject to some bias and will tend to overestimate the worker's real intake. It is always preferable to avoid bias, and it is demonstrated that this can be avoided by assuming a constant chronic intake. If this is of concern and a software program is being used to estimate intake, then the same result can be obtained by setting the intake to be constant and chronic throughout the monitoring interval.

### *7.3.3 Intake Pathways*

Although intakes by inhalation alone are the most frequent in the workplace, intakes by ingestion and uptake through wounds and intact skin cannot be excluded. Sometimes the worker touches the mouth with contaminated hands and ingestion occurs. If the pathway of intake is not known and several bioassay results are available, including different types of bioassay measurements, a comparison of these results with the *m(t)* tables may help in determining the pathway of intake. In some facilities simultaneous intakes by inhalation and ingestion can occur. In principle, results from the ingestion and inhalation tables can be combined to give predicted values of *m(t)*. Alternatively, it can be modelled by assuming inhalation with a large AMAD (> 5 μm).

If the radionuclide activity can be assessed by direct measurements, lung counting can be used to differentiate between inhaled and ingested material. However, if this is not possible and the radionuclide is in an insoluble form, interpretation of activities excreted in faecal and urine samples in terms of intake is quite problematic. Both the ingested material and the inhaled material deposited in the upper respiratory tract will clear through the faeces in the first few days after intake. Consequently, it is important to initiate excreta sampling as soon as possible after the intake, continuing for an extended period. Material in the faeces after the second week will be exclusively from the respiratory tract, and can be used, together with the appropriate values of *m(t)*, to correct the earlier faecal samples for this component. In the monitoring of workers chronically exposed to long–lived, insoluble radionuclides, activities in the faeces after a 15 days' absence from work will mostly reflect the delayed clearance from inhaled material (IAEA, 1999, 2004).

### *7.3.4 Particle Size/Chemical Composition*

Although recent reviews of reported measurements of AMAD in workplaces (eg. Dorrian and Bailey 1995) support the ICRP publication 66/68 default value of 5 μm for occupational exposure, they also show that a wide range (about 1–20 μm) has been observed (see Table 7.1). If the airborne contamination in the workplace has been well characterised, it may be possible to use a more realistic value based on measurements of the activity size distribution. Alternatively, if there are suitable early measurement data available, an "effective" AMAD can be inferred a posteriori from the measurements. The main effect of the aerosol AMAD is to determine the relative amounts deposited (i) in the upper respiratory tract (extrathoracic airways, ET, bronchi, BB, and bronchioles, bb, in the HRTM), which (if not absorbed into blood) is mainly cleared rapidly to the alimentary tract and hence to faeces within a few days, and (ii) in the lower respiratory tract (alveolar-interstitial, AI, region in the HRTM), which is mainly cleared slowly from the lungs. ICRP Supporting Guidance 3 (Fig. 7.1) showed that for a relatively insoluble (Type M or S) material inhaled by a Reference Worker, the ratio of cumulative faecal excretion over the first 3 days to lung activity on day 3 increased almost linearly from about two at 1 μm AMAD to twelve at 10 μm AMAD. Hence the observed ratio could be used to infer the "effective" AMAD. It is referred to as "effective", because the ratio will be determined not only by the aerosol size, but also by the subject's breathing pattern (especially if it involves mouth-breathing) and inter-subject variation in deposition under any given set of conditions. Because it takes account of these, it is preferable for dose assessment than *a priori* measurements of the AMAD.

## 7.4 Measurements

### *7.4.1 Need for Quality Assurance Programme*

The need for a quality assurance (QA) programme within an overall radiation protection programme has been discussed in a recent ISO standard (ISO, 2006). Reference should be made to the ISO standard for a complete account, but some of the more important issues are:

- In deciding on the nature and extent of the QA programme, consideration should be given to the number of workers monitored, and the magnitude and probability of exposures expected
- Assumptions on factors such as radionuclide composition, inhaled particle size, identity of chemical compounds, absorption behaviour, etc., should be verified by appropriate measurements
- Reviews or audits should be conducted at appropriate times (eg. when a new monitoring programme is implemented, or when a significant change to a programme is made)

Laboratories should participate in national or international intercomparisons of measurements and/or dose assessments at appropriate intervals. Such participation enables the determination of the accuracy of measurement and dose assessment procedures, improves reliability, and facilitates harmonisation of methods.

### *7.4.2 Background Subtraction*

Radionuclides from the three natural radioactive decay series are present in all environmental media, and thus are also contained in foodstuffs, drinking water and in the air, leading to intakes by human populations. Their presence needs to be taken into account in any analysis.

ICRP Publication 23 on Reference Man gives data on the daily intake and losses for different elements. For thorium, daily losses are 0.1 µg (0.4 mBq) and 2.9 µg (12 mBq) in urine and faeces respectively. For uranium these losses range from 0.05 – 0.5 µg (1.25 – 12.5 mBq) more recent measurements suggest a value of 0.1 may be more realistic (refs)in urine and from 1.4 – 1.8 µg (35 – 45 mBq) in faeces. For radium these losses are 3 mBq in urine and 80 mBq in faeces.

Similar results were found by Spencer et al (1990). In their study of intake and excretion patterns of naturally occurring isotopes in human to calculate the uptake factor of uranium. In a study of the uranium in excreta of mill crush-men, Fisher et al (1983), found in three control subjects, similar levels of uranium in urine but much higher levels of uranium in faecal sample. They attributed these higher levels to ingestion of dissolved uranium in local water supplies. Julião et al (2003) observed higher levels of thorium in excreta samples of inhabitants of high natural background area. Naumann et al (1998) measured U, Th and Ra in faecal samples from a person living in the Berlin area and found similar results as those given for the reference man. Hurtgen (2001) found comparable results in urine and faeces samples from workers potentially exposed to actinides. The $^{234}U$ /$^{238}U$ activity ratio was found to be 1.7 in urine and faeces; Also for thorium, $^{228}Th$ is always found in excess to its parent $^{232}Th$. The $^{228}Th$ /$^{232}Th$ activity ratio was 2.3 and 10 respectively in urine and faeces. Unexpectedly high activity of $^{228}Th$ in excretion samples have been found following consumption of Brazil nuts. (Bull R.K. et al. 2006)

In a recent study of residents of Finland, Karpas et al (2005) has shown that drinking water containing elevated levels of uranium is reflected in high concentration of

uranium in urine, hair and nail samples. The isotopic ratio $^{234}U$ /$^{238}U$ in excreta sample can be directly correlated to the ratio in the drinking water consumed by individuals from whom the sample were collected. This provides a direct link between the source of exposure to uranium and the bioassay analysis.

A knowledge of the natural background activity found in bioassay samples is thus essential if occupational intakes have to be assessed. "Blank" bioassay sample should be obtained prior to the assessment work for individuals in potentially contaminated areas to allow natural or non occupational intakes and occupational intakes to be distinguished.

### *7.4.3 Data Collection and Processing Before Use*

Some types of measurement data may need processing before use. Examples include:

- Lung. Generally, the combined activity in lungs and thoracic lymph nodes is referred to as 'lung' activity, and it is this quantity that is calculated by internal dosimetry software. Where estimates of lung and lymph activity are given separately, they should be summed. "Chest" measurements may also include counts from activity in liver and skeleton for radionuclides that concentrate in these tissues, and their contributions will be need to be subtracted.
- Faeces. The transit time through the alimentary tract is subject to large inter (and intra-) subject variations. Moreover, while for ease of computation transit through the alimentary tract is represented by a series of compartments that clear exponentially, in practice, the movement is more like "slug" flow. It is therefore unlikely that individual daily faecal clearance measurements in the first few days after intake will follow the predicted pattern, and so it is best to consider cumulative excretion over the first few days.
- Urine and faecal samples collected over periods less than 24 hours should be normalized to an equivalent 24 hour value. This is achieved by multiplying by the ratio of the reference 24 hour excretion volume or mass to the volume or mass of the sample. The reference volumes, for males and females respectively, are: for urine 1.6 litres and 1.2 litre; and for faeces 150 g and 120 g (ICRP, 2002a). For urine the recommended method is to normalise to the amount of creatinine excreted per day; 1.7 g and 1.0 g for males and females respectively [ICRP, 2002a]. If the 24 hour sample is less than 500 ml for urine or less than 60g for faeces, then it is doubtful that it has been collected over a full 24 hour period and normalization should be considered.

## 7.5 Assessment of Uncertainty on Data

The uncertainties on the data are of great importance for the evaluation for several reasons:

- They enable an objective decision to be made on whether a measured value is due to a new intake, or due to previous intakes that already have been evaluated.
- They enable an objective decision to be made on whether a measured value is consistent with previous evaluations, or if it indicates the previous evaluations to be wrong.
- They can have a strong influence on all evaluations using weighted fitting procedures (i.e. where there is more than one data point).

- They enable rogue data to be identified objectively.
- They enable objective (statistical) criteria (goodness-of-fit) to be calculated, which are used to determine whether the predictions of the biokinetic model (with a given set of parameter values) used to assess the intake and dose are inconsistent with the data.
- They enable statistics, such as the $\chi^2$, to be calculated, which are used to compare the fits to the data of different models/parameter values.

Uncertainties in measurement results have been discussed in several IAEA publications on direct (IAEA, 1996) and indirect (IAEA, 2000) methods to measure radionuclides within the body. There are no standard procedures for indirect or direct bioassay measurements, although some examples of bioassay methods are given in these publications. The choice of the procedure, detector or facility will depend on the specific needs such as the nuclides of interest, minimum detectable activities, budget, etc. All procedures used to quantify the activities of a radionuclide are sources of random and systematic errors. Uncertainties in measurements are mainly due to counting errors, validity of the calibration procedures, possible contamination of the source or the measurement system, and random fluctuations in background.

Typically, the components of uncertainty are grouped into two categories: Type A and Type B uncertainties. ISO's Guide to the Expression of Uncertainty in Measurement (BIPM *et al*., 1995) discriminates between the Type A evaluation of uncertainty - that based on statistical means - and the Type B evaluation of uncertainty - that based on non-statistical means. However, as noted in a recent publication of the UK National Physical Laboratory (Cox and Harris, 2004), it is sometimes more useful to make a distinction between effects that can be regarded as random, and those that can be regarded as systematic. Cox and Harris note that the subdivision into Type A and Type B evaluations of uncertainty will correspond in some instances to random and systematic effects, respectively, but not in all circumstances. In this publication, Type A uncertainties are taken to arise only from counting statistics, which can be described by the Poisson distribution. Type B components are due to all other sources of uncertainty.

Examples of Type B components for *in vitro* measurements include the quantification of the sample volume or weight; errors in dilution and pipetting; evaporation of solution in storage; stability and activity of standards used for calibration; similarity of chemical yield between tracer and radioelement of interest; blank corrections; background radionuclide excretion contributions and fluctuations; electronic stability; spectroscopy resolution and peak overlap; contamination of sample and impurities; source positioning for counting; density and shape variation from calibration model and assumptions about homogeneity in calibration (Skrable et al, 1994). These uncertainties apply to the measurement of activity in the sample. With excretion measurements, the activity in the sample is used to provide an estimate of the subject's average excretion rate over 24 hours for comparison with the model predictions. If the samples are collected over periods less than 24 hours then they should be normalised to an equivalent 24 hour value. This introduces additional sources of Type B uncertainty relating to biological (inter-and intra-subject) variability and sampling procedures, which may well be greater than the uncertainty in the measured sample activity.

Examples of Type B components for i*n vivo* monitoring include counting geometry errors; positioning of the individual in relation to the detector and movement of the person during counting; chest wall thickness determination; differences between phantom and individual or organ being measured, including geometric characteristics, density, distribution of the radionuclide within the body and organ and linear attenuation coefficient; interference from radioactive material deposits in adjacent body regions; spectroscopy resolution and peak overlap; electronic stability; interference from other radionuclides; variation in background radiation; activity of the standard radionuclide

used for calibration; surface external contamination of the person; interference from natural radioactive elements present in the body; and calibration source uncertainties (IAEA,1996, Skrable et al, 1994).

For partial body measurements it is difficult to express the result in terms of organ activities. For the determination of lung activity by measurement over the chest, for example, not only individual calibration problems (such as the thickness of the subject's chest wall) must be considered, but also radiation from various other body regions, and not only from the lungs, may be detected. So additionally some assumptions must be made about the biokinetic behaviour of the radionuclide. An example for the case of $^{241}$Am is given in the IAEA Safety Practice on Direct Methods for Measuring Radionuclides in the Human Body (IAEA, 1996).

The Type A uncertainties (i.e. counting statistics) decreases with increasing activity and/or with increasing counting time whereas the Type B components can be considered to be independent of the activity or the counting time. Therefore, when activity levels are low and close to the limit of detection, the total uncertainty is governed by the Type A component (i.e. by counting statistics). For radionuclides that are easily detected and present in sufficient quantity, the total uncertainty is governed by the Type B components (i.e. by uncertainties other than counting statistics).

The Type B components cannot be expressed in terms of Poisson statistics, and thus there is a problem in combining the Type B and the Type A components in order to derive the total uncertainty of the data point.

Table 7.9 lists typical values for the various components of uncertainty of *in vivo* measurements. The uncertainty is given in terms of the scattering factor SF assuming that the distributions of the measurements can be a log-normal distribution. The SF is the geometric standard deviation of the distribution. For example, the scattering factor due to counting statistics is given as SF = 1.5 for low photon energy counting. This means that the scattering of the measured values due to counting statistics would result in 67% of the values to be in between $x_{50}/1.5$ and $x_{50}*1.5$, where $x_{50}$ is the median of all measured values.

Based on the experience gained in the IDEAS project (Work Package 3: Evaluation of Incorporation Cases), as well as on general considerations, the following general approach for the calculation of the total uncertainty may be applied

$$SF = \exp\left[\sqrt{\sum_i \ln^2(SF_i)}\right] \qquad (7.2)$$

with $SF$ total scattering factor
$SF_i$ scattering factor due to component i

When applying this approach on the SF values given in Table 7.9, the values in Table 7.10 are derived for the total scattering factors.

Typical values for Type B scattering factors are given in Table 7.11. In practice routine urinary excretion data from plutonium workers is often found to have a log-normal distribution with a SF ranging from 1.3 to about 2.0 (Moss et al, 1969 and Riddell et al, 1994). However, Moss et al. (1969), showed that when the sampling method and analytical procedures are carefully controlled for true 24-h urine samples, over 5 days, then the SF is significantly less (1.1).

The use of scattering factors is under development and the literature should be consulted for the most up to date information.

## 7.6 Processing of Measurement Data

### *7.6.1 Introduction*

Direct and indirect measurements result in data about the amount(s) of radionuclides present in the body, in parts of the body including specific body organs or tissues, in a biological sample or in a sample from the working environment. The first approach to interpretation of these data is likely to be an estimation of the intake of the radionuclide by the worker. The biokinetic models (see Chapter 3) which describe body and organ contents, and activity in excreta, as a function of time following intake, and exposure models which relate intake to workplace conditions, are used for this purpose. These models are used to calculate values of the measured quantities for unit intake, *m(t)*, at a time *t* after the intake. Values of m(t) are given in the OIR documents and these values can be used to estimate the intake. Once the intake is estimated, the committed effective dose is then computed from the product of the intake and the appropriate dose coefficient. Alternatively, measurements of activity in the body can be used to estimate dose rates directly, if a sufficient number of measurements are available to determine retention functions (Annex C).

When only a single bioassay datum is available, a point estimate of the intake is made. If multiple measurements are available, a best estimate of intake may be obtained by applying a statistical fitting method.

When significant intakes may have occurred, more refined calculations based on individual-specific parameters (special dosimetry) should be made.

The text below gives basic information on data handling. Further information is given in Annex E.

### *7.6.2 Single Measurements*

#### 7.6.2.1 Special monitoring

For special or task-related monitoring when the time of intake is known, the intake can be estimated from the measured results using the m(t) values given in the OIR publications. If only a single measurement is made, the intake, *I*, can be determined from the measured quantity, *M*, by:

$$I = \frac{M}{m(t)} \qquad (7.6)$$

The intake can be multiplied by the dose coefficient to give the committed effective dose; this can then be compared with the dose limit or any pre-determined investigation level based on dose. If the measurement indicates that an investigation level has been exceeded, further investigation is required.

Care must be taken to ensure that the measurement result, *M*, and *m(t)* are comparable; for example, in the case of urinalysis, the bioassay result must be expressed as the total activity in a 24-hour urine sample at the end of collection (not at analysis).

#### 7.6.2.2 Routine monitoring

For routine monitoring, it is assumed that the intake occurred at the mid-point of the monitoring interval of T days. For a given measured quantity, M, obtained at the end of the monitoring interval, the intake is:

$$I = \frac{M}{m(T/2)} \qquad (7.7)$$

where m(T/2)is the predicted value of the measured quantity for a unit intake assumed to occur at the mid-point of the monitoring interval. The dose from the intake in the monitoring interval is obtained by multiplying the intake by the dose coefficient. The dose or intake can be compared with the pro-rata fraction of the dose limit or of the activity corresponding to that limit. Alternatively, the dose or intake can be compared with predetermined investigation levels.

It is normally assumed that in the absence of specific knowledge about the time of the intake it occurred at the mid-point of the monitoring interval. A recent paper has, however, noted that this can lead to some bias in the estimated dose and in some circumstances a more appropriate assumption would be to assume a constant chronic intake. This is described further in Annex F.

An intake in a preceding monitoring interval may influence the actual measurement result obtained. For a series of measurements in a routine monitoring programme, the following procedure may be followed:

- Determine the magnitude of the intake in the first monitoring interval.
- Predict the contribution to each of the subsequent measurements from this intake.
- Subtract the corresponding contributions from all subsequent data.
- Repeat above for the next monitoring interval.

Further details of this procedure are given in stage 2 of the flow charts (Chapter 8).

If a measured value in a routine monitoring programme exceeds a pre-determined investigation level (or dose level), special monitoring is started so that the intake and the dose can be assessed more accurately.

### *7.6.3 Multiple Measurements*

Usually, the bioassay data for an intake estimate will consist of results for different measurements performed at different times, and even from different monitoring techniques, eg. direct and indirect measurements.

To determine the best estimate of a single intake, when the time of intake is known, it is first necessary to calculate the predicted values, $m(t_i)$, for unit intake of the measured quantities. It is then required to determine the best estimate of the intake, $I$, such that the product $I\ m(t_i)$ "best fits" the measurement data $(t_i, M_i)$. In cases where multiple types of bioassay data sets are available, it is recommended to assess the intake and dose by fitting predicted values to the different types of measurement data simultaneously. For example, if urine and faecal data sets are available then, the intake is assessed by fitting predicted values to both data sets simultaneously (Annex E).

Numerous statistical methods for data fitting are available [IAEA, 2004]. The two methods that are most widely applicable are the maximum likelihood method and the Bayesian approach (Annex E). Other methods such as the mean of the point estimates and the least-squares fit can be justified on the basis of the maximum likelihood method for certain assumptions on the error associated with the data. For example, the least square method can be derived from the maximum likelihood method if it is assumed that the uncertainty on the data can be characterised by a normal distribution. The assumed distribution (e.g. normal or lognormal) can have a dramatic influence on the assessed intake and dose if the model is a poor fit to the data. However, as the fit of the model to

the data improves, the influence of the data uncertainties on the assessed intake and dose reduces.

In these guidelines it is assumed that the uncertainty on the data can be characterised by a lognormal distribution with a given SF. This restricts the choice of the fitting method. The maximum likelihood method is recommended in these guidelines as it can be applied to such cases where it is assumed the measurements are lognormally distributed. The following section gives a simple equation for estimating the intake for the case when the SF is constant for each data point. Annex E derives equations for more general cases where the SF is different for each data point (equation E.16) or when different types of measurement data are available (equation E.18).

The Bayesian approach is not applied in the guidelines but has been discussed by Miller et al. (2002a and 2003) and is also discussed in Annex E.

#### 7.6.3.1 Maximum likelihood method

Using the maximum likelihood method, the "best fit" value of the intake, *I*, is the intake for which the probability of obtaining the measurement data is a maximum. To apply the maximum likelihood method the probability distribution of the measurements (i.e. measurement error) is required. The type of distribution assumed can effect the estimated intake. Section 7.5 proposes that the probability distribution of measurements can be approximated by a lognormal distribution with a given SF. This is a reasonable assumption if the counts are relatively large (i.e. the SF for Type A uncertainties is < 1.4).

If it is assumed that the overall SF for each measurement value is constant then the best estimate of the intake, I is simply the geometric mean of the point estimates, $I_i$. Therefore, the best estimate of intake is given by the following equation.

$$\ln(I) = \frac{1}{n}\sum_{i=1}^{n}\ln(I_i) = \ln\left[\left(\prod_{i=1}^{n} I_i\right)^{\frac{1}{n}}\right]$$

That is:

$$I = \sqrt[n]{\prod_{i=1}^{n} I_i} \qquad (7.8)$$

where $I_i$ is the intake calculated from the $i^{th}$ measurement and is given by:

$$I_i = \frac{M_i}{m(t_i)}$$

In such a case the best estimate is independent of the value of SF.

Generally, the SF can be assumed to be constant for each measurement (within the same type of monitoring data) if the scattering factor is dominated by Type B uncertainties (i.e. uncertainties other than counting errors such as calibration errors or errors related to sampling procedures as for excretion data).

### *7.6.4 Extended Exposures*

One of the factors that influence the interpretation of bioassay results is the temporal variation of the intakes of radioactive material. The pattern of intake, although often poorly characterized, is an important factor in the correct interpretation of measurements and thus for dose assessment. In general, the amount of activity present

in the body and the amount excreted daily depend on the length of time the individual has been exposed. Consequently, the correct interpretation of bioassay measurements requires information on the complete exposure history of the worker to the particular radionuclide of interest. The bioassay result obtained, e.g. the amount present in the body, in body organs, or in excreta, will reflect the super position of all the previous intakes, whether isolated or persistent.

Therefore, any previous intakes that influence the actual measurement result need to be taken into account. It is proposed to calculate the net value of the activity of the radionuclide, $N_i$ by subtracting the contributions from previous intakes, $P_i$ from the measurement value (i.e. $N_i = M_i - P_i$). For simplicity, ignoring the uncertainty in $P_i$, equation 7.8 can be applied to determine the best estimate of intake but with:

$$I_i = \frac{N_i}{m(t_i)} \qquad (7.9)$$

In applying equation 7.8 to such cases, it is assumed that the net values of the activity are lognormally distributed with a given SF (Sections 7.5.3.1 and 7.5.3.2). It is acknowledged that the actual distribution of the net values is not lognormal because subtracting a value ($P_i$) from lognormally distributed values ($M_i$) does not result in another lognormal distribution.

An alternative approach is to fit the previous intakes as well as the intake of interest to all the data simultaneously using the maximum likelihood method (Annex E). However, this requires appropriate software tools to do this.

***Exposures over a time period***

When exposure is known to extend for several days, perhaps as a result of an undetected incident, bioassay results may be interpreted as containing an independent contribution from each day's intake.

For example, consider the case where a subject has be exposed at a constant chronic rate of intake over a period of T days (i.e. from 0 to T days) and a measurement is carried out at a time $t_i$ after the start of the chronic period. The calculate value of the measured quantity for unit intake arising from an intake rate of 1/T Bq $d^{-1}$ over a period of T days is approximated by:

$$m_c(t_i) = \frac{1}{T}\sum_{j=1}^{T} m(t_i - j) \quad if\ \mathrm{T} < t_i$$

or (7.10)

$$m_c(t_i) = \frac{1}{T}\sum_{j=0}^{t_i - 1} m(t_i - j) \quad if\ \mathrm{T} > t_i$$

Again equation 7.8 can be applied to determine the best estimate of the total intake, *I* but with:

$$I_i = \frac{M_i}{m_c(t_i)} \qquad (7.11)$$

Equation 7.10 only gives approximate values for $m_c(t)$ and is not very accurate if

m(t) varies a great deal over the period of summation. In such cases appropriate software tools are required to improve the accuracy of the numerical integration over the exposure period.

***Chronic and intermittent exposures***

In routine monitoring of workers, especially for long-lived radionuclides, it is highly desirable to produce a scheme in which the workers' realistic exposure (e.g. a weekly cycle) is considered. The schedule of work may differ for individual workers and modifications should be introduced as necessary. The use of an input function that represents the worker's routine intake permits the interpretation of bioassay results according to the day of the week on which samples are taken. In this way the short-term components associated with lung clearance will be better accounted for, since the early clearance component(s) of excretion may introduce a significant difference before and after an interruption in exposure, e.g. the weekend. The interpretation of these data requires, in most cases, appropriate software tools and is beyond the scope of this report.

For long-lived radionuclides, chronic exposures will eventually produce an equilibrium value of activity in the body. Equilibrium values for selected radionuclides will be provided in the OIR Publications.

### *7.6.5 Number and Type of Data Required for Assessment of Dose*

The reliability of the dose assessment depends on the number and type of the monitoring data. Thus, there are minimum requirements for the type and number of monitoring data, depending on the involved radionuclide and the dose range. Table 7.12 shows the requirements for some selected radionuclides, as suggested by IDEAS (Doerfel et al. 2005a). Ideally the measurements should be distributed appropriately over the relevant time range given in Table 7.12.

### *7.6.6 Special Aspects*

***Handling of data below limits of detection***

If data are reported as being below the lower limit of detection (LLD) and only the LLD value is recorded then it is recommended to use the maximum likelihood method to obtain the best estimate of intake (Annex E). It can be shown that this method leads to an unbiased estimate of the intake (Marsh 2002). However, if this is not possible, because appropriate software is not available, then it is recommended to set the LLD data to LLD/2 value.

***Handling of data influenced by decorporation therapy***

Generally, it can be assumed that urine data for actinides and lanthanide radionuclides are affected by DTPA. If DTPA has been effective in reducing systemic uptake then systemic organ retention and systemic faecal will also be affected. Excretion rates may be influenced for up to around 100 days after cessation of treatment.

The method of Jech et al (1973) is proposed here: exclude urinary excretion data that have been affected by DTPA. Following La Bone (1994, 2002) it is proposed that data up to 100 days following chelation should be excluded.

The alternative approach is to use a model for the urinary excretion of the chelated actinide, to compensate for the enhanced excretion (Hall method, La Bone, 1994). This is preferable, when an early assessment is required, because it makes more use of the available information, but the IDEAS partners were unable to propose a

suitable formula at this time.

***Identification of rogue data***

A systematic basis is needed to identify outliers, and criteria to exclude them. Outliers above and below the trend of the other data have different significance. A point above the trend might indicate another intake. A point below is more likely to result from a transcription or measurement error.

The problem of deciding how to identify outliers is not straightforward. Ideally, outliers should be identified before fitting model predictions to the data. If not, then the assessor faces a dilemma when the model does not fit the data: should the model parameters be varied to obtain a fit, or should the data that does not fit be rejected. So ideally, the trend of the data should be obtained first by, for example, fitting a sum of exponentials to the data and then using a statistical test to reject the data. In practice, it is realised that this procedure could be time consuming, and many assessors will rely on judgement when deciding to reject certain data. Specifically, care must be taken in excluding data, particularly if a group of data at early or late times does not appear to be predicted by the model, then model parameters should be varied in preference to excluding data.

For measurement data suspected of being "rogue" a check should be made on whether inclusion or exclusion significantly affects the intake and dose. If it does not, there is no point in expending effort on justifying excluding it: it should be included. If it does have an effect, then a statistical test should be carried out to determine if it is an outlier. If it is an outlier then it should be excluded.

To identify outliers the following statistical test is proposed. A measurement value is an outlier if it is more than a factor of $SF^3$ away from the trend of the other data, where SF is the scattering factor.

If the data set is limited after excluding outliers, then further measurements maybe required for the assessment of dose (Section 7.6.5).

***Criteria for rejecting fit***

In assessing intakes and doses, the underlying starting assumption is that:

- the structure of the biokinetic model is a realistic representation of the physical and biological processes, and
- the model parameter values are correct.

Estimates of bioassay quantities will be unbiased only if these conditions are met. These assumptions are analogous to the null hypothesis in classical statistics. In cases where the model predictions are inconsistent with the data (i.e. fits are inadequate) this indicates that either the model parameter values, or the structure of the model is incorrect. The classical statistical approach is to reject the model and to repeat the assessment with different model parameter values or with a new model structure so that the predictions are not inconsistent with the data. Before the model structure itself can be rejected, it is necessary to first consider changes to the model parameter values. In these guidelines only changes to the parameter values are considered, not to the model structure.

It is important to remember that it is not possible to prove that the null hypothesis is true. Test statistics are used to indicate that the null hypothesis is false. The criteria for rejecting the null hypothesis, (i.e. stating the fit is inadequate), needs to be defined before the assessment is carried out.

A comprehensive discussion of all the possible statistics that can be used to quantify whether a fit is inadequate is beyond the scope of this document. Only the *chi-squared* test statistic, $\chi_O^2$, is considered here.

If it is assumed that each measurement, $M_i$, is taken from a lognormal distribution with a scattering factor of $SF_i$ then for $n$ measurements, $\chi_O^2$ is defined as:

$$\chi_o^2 = \sum_{i=1}^{n} \left( \frac{ln(M_i) - ln[I\, m(t_i)]}{ln(SF_i)} \right)^2 \qquad (7.15)$$

The product $I\, m(t_i)$ is the predicted value.

The above formulae do not apply to data that are reported as below the lower limit of detection (<LLD).

When fitting predicted values to different types of data simultaneously, the overall $\chi_O^2$ is equal to the sum of the calculated $\chi_O^2$ values for each data set.

If the predictions are inconsistent with the data, then the calculated value of $\chi_O^2$ is inconsistent with the theoretical *chi-squared* ($\chi^2$) distribution with (n-1) degrees of freedom. The expected value of $\chi^2$ is equal to the number of degrees of freedom (i.e. $n$-1).

The actual number degrees of freedom when varying $I$ parameters for a linear model (with respect to its parameters) is $n$-$I$. In this case the biokinetic model is not linear with respect to most of its parameters, other than the intake. If the fit is rejected assuming $n$-1 degrees of freedom then the fit would also be rejected if the actual number of degrees of freedom is less. For cases where there is comprehensive data so that $n>>I$) it is proposed to assume $n$-1 degrees of freedom for each step of the procedure given in the flow charts (Chapter 9).

The probability of observing a larger $\chi^2$ value than $\chi_O^2$ for ($n$-1) degrees of freedom is given by the p-value, which can be obtained from Statistical Tables. The p-value is the fraction of the theoretical $\chi^2$ distribution that lies above the calculated $\chi_O^2$ value. So if the p-value is very small, the calculated $\chi_O^2$ value is very much larger than expected and therefore it can be concluded that the predictions are likely to be inconsistent with the data and the assumed uncertainties.

The $\chi^2$ test uses the assumed uncertainties. If the assumed uncertainties are overestimated then $\chi_O^2$ is too small. The converse is also true; if the assumed uncertainties are underestimated then $\chi_O^2$ is too large. This is one of the reasons why it is important to assess realistic uncertainties (Section 7.5.6).

It is proposed that the fits to the data are judged to be inadequate if:

- the probability that $\chi^2$ is greater than $\chi_O^2$ is 5% or less (i.e. if p-value < 0.05).

In other words the fit is inadequate at the 5% level of significance, or if

- the fit displayed graphically looks unreasonable by eye.

It is also acknowledged that whether or not the fit displayed graphically looks unreasonable by eye is a subjective judgement. Generally, however, a fit would be considered unreasonable if all, or a long series, of data were systematically underestimated or overestimated.

If a series of data were systematically underestimated or overestimated then this can be quantified objectively by the use of other test statistics such as the auto correlation coefficient (Chatfield, 2004) and the Durbin Watson statistic (Durbin and Watson 1970). The auto correlation coefficient statistic has the advantage that it is relatively insensitive to the magnitude of the assumed measurement uncertainties.

# 8 Dose Assessment

## 8.1 Introduction

In carrying out the assessment (evaluation) of committed doses from monitoring data following intakes of radionuclides, the assessor may well have to make assumptions about factors such as the pattern of intake and properties of the material. When more than one measurement is available, issues such as the weighting applied to the different data can substantially affect the result. Recent intercomparison exercises (eg. Doerfel et al, 2000) have shown the wide range in doses that can be obtained from the same data set as a result of such factors, and hence the need for guidance on harmonising evaluations.

The procedures proposed in this chapter are based on the following principles:

- Harmonisation: by following the procedures any two assessors should obtain the same estimate of dose from a given data set or at least understand why differences have occurred
- Accuracy: the "best" estimate of dose should be obtained from the available data
- Proportionality: the effort applied to the evaluation should be proportionate to the dose – the lower the dose, the simpler the process should be.

### 8.1.1 *Harmonisation*

A well-defined procedure is needed and for this reason the process is defined here primarily by means of a series of flow-charts. So far as possible, the structured process has been made widely applicable, i.e., it does not assume that the assessor has the use of sophisticated bioassay interpretation software. For routine monitoring situations, where typically there is only one measurement relating to each intake, it is reasonably straightforward to define a procedure. However, in special monitoring situations, where typically there is more than one measurement and quite possibly more than one type of measurement (urine, faeces…) different options for data handling can easily lead to different evaluated doses, even when the same model, parameter values and software are used. Another range of options, and opportunities for different evaluated doses, arises in situations where it is appropriate to consider changing parameter values from the ICRP defaults. Proposals are made here for a systematic approach to dose assessment in all these situations.

### 8.1.2 *Accuracy*

It is recognised that the uncertainties associated with assessed internal dose can be considerable, especially for actinides which are difficult to detect in the body and have relatively high dose coefficients (Sv $Bq^{-1}$). If the initial estimate of dose exceeds 1 mSv, it could well be that the possibility of a substantially higher dose (eg. 6 mSv) cannot easily be excluded. It is then important to make best use of the available information. To do so may well involve changing parameter values from their ICRP default values and guidance is therefore needed on which parameter values might reasonably be varied according to the circumstances.

### 8.1.3 *Proportionality*

The effort applied to the evaluation of incorporation monitoring data should broadly correspond to the expected level of exposure, and the complexity of the case. On the one hand, if the exposure is likely to be very low with respect to the dose limits, simple evaluation procedures with a relatively high uncertainty may be applied. On the

other hand, if the monitoring values indicate the exposure to be close to or even above the dose limits, much more sophisticated evaluation procedures will need to be applied. These take account of any case-specific information available, so that the uncertainty and bias on the best estimate are as low as reasonable achievable.

### 8.1.4 *Levels of task*

With respect to operational radiation protection the following structure of "Levels of Task" are proposed and were discussed in Section 7.2.

- Level 0 (effective dose < 0.1 mSv/a)
- Level 1 (0.1 mSv/a < effective dose < 1 mSv)
- Level 2 (1 mSv < effective dose < 6 mSv)
- Level 3 (effective dose > 6 mSv)

Level 0 is the lowest level and it refers to cases where the effective annual dose would be most likely below 0.1 mSv ($10^{-4}$ Sv), even if there should be similar intakes in each monitoring interval of the year.

## 8.2 Structured Approach to Dose Assessment

### 8.2.1 *Introduction*

In the following Chapter the structured approach to the assessment (evaluation) of internal doses from monitoring data is described. It consists of a series of "Stages", broadly corresponding to the Levels of task given above. Each Stage consists of a series of "Steps", and is presented diagrammatically in a flow chart, with a brief explanation of each Step in the text. Detailed descriptions of aspects of the evaluation process are given. Consideration is also given to the quantity and quality of monitoring data needed for the assessment of doses greater than 1 or 6 mSv. The Stages are shown diagrammatically in Figs 8.1-8.7. For more complex assessments a more detailed approach is described in Annex G.

### 8.2.2 *Stage 1. Level 0, and for higher exposures [Fig. 8.1]*

Level 0 (Steps 1.1 and 1.2) refers to cases where it is expected that the annual dose (committed effective dose from intakes of radionuclides that occur in the accounting year) is likely to be below 0.1 mSv, even if there were similar intakes in each and every monitoring interval during the year. At this level there is no need to evaluate the intake or dose from the measured values explicitly. The effective dose can be reported as zero, by analogy with the rounding of doses in external dosimetry. However, the measured value should be recorded, because it may provide information useful for further assessments in the future.

#### *Step 1.1: Identify monitoring value (M) and duration of monitoring interval (T)*

Some treatment of the data may be required before an evaluation can be made. In particular consideration should be given to the presence of other radionuclides, as well as that measured (indicator) nuclide, which may add significantly to the dose, or even exceed that from the radionuclide measured (see Chapter 8).

#### *Step 1.2: Compare measurement with critical monitoring quantity $M_c$*

If M < $M_c$ then the annual dose is probably less than 0.1 mSv. The evaluation stops and the measured value M is recorded together with all relevant information (radionuclide, activity, type of measurement, type of monitoring etc).

***Step 1.3: Exposure above Level 0***

Since M > $M_c$ the annual dose could be more than 0.1 mSv. Go to Stage 2 to check on the statistical significance of the measurement.

### 8.2.3 *Stage 2. Level 1, and for higher exposures: Check on significance of new measurement and consistency with previous evaluations [Fig. 8.2]*

Level 1 refers to cases where it is expected that the annual dose (committed effective dose from intakes of radionuclides that occur in the accounting year) is likely to be above 0.1 mSv. At this level the intake or dose from the measured values should be calculated explicitly. Before starting the assessment of intake and dose, however, the statistical significance of the measured value M should be estimated. This includes the assessment of uncertainty on M (Step 2.1) as well as the calculation of the contributions from previous intakes to M (Step 2.2) in order to decide whether M is:

- due to a new intake, or
- due to a previous intake, or
- if it is in contradiction to previous assessments (Steps 2.3 – 2.7).

***Step 2.0: Understanding the case***

Plot the data (including those from previous measurements if available) and do some simple hand calculations.

***Step 2.1: Assessment of the uncertainty on M***

Realistic estimates of the overall uncertainty on each data point are required. Here they are expressed as a total "**scattering factor" (SF)** (see how to assess uncertainty on data in Chapter 7).

***Step 2.2: Calculation of the contributions P from previous intakes***

The contributions (P) from all previous intakes of the radionuclide considered are calculated, taking into account all pathways of intake, and all intakes of mixtures where the radionuclide was involved.

***Step 2.3: New intake confirmed***

If M > $SF^2$ * P, then there is a 95% probability for a new significant intake. Calculate the net value (N) of the radionuclide by subtracting P from the measured value M and go to Stage 3, in order to check whether the next stage of the task is Level 2 or Level 3.

***Step 2.4: New intake not confirmed***

If $P/SF^2$ < M < $P*SF^2$, then the measured value M is consistent with the intakes assessed previously, and there is most probably no new intake (i.e., there is no evidence for a new intake). The evaluation stops and the measured value M is recorded together with all relevant information (radionuclide, activity, type of measurement, type of monitoring etc).

***Step 2.5: Discrepancy with the previous evaluations***

If $M < P/SF^2$, then there is a discrepancy with the previous assessments. The reason for the discrepancy could be (i) the measured value M is not reliable and/or (ii) the previous assessments are wrong. For example, an intake occurring near the end of the previous monitoring interval is likely to have been overestimated based on an assumed intake at the mid-point.

***Step 2.6: Check on the reliability of M***

General possibilities for errors could be transcription errors, wrong date of measurement and/or wrong units. For whole body counting special reasons for errors include: external contamination, mismatching of calibration and actual activity distribution (i.e lung activity calculated with whole body efficiency etc.). For excretion measurements possibilities include incomplete collection of the sample etc.).

***Step 2.6.1: Reassess previous intakes***

If it cannot be demonstrated that M is unreliable, then reassess the previous intake(s), i.e. check the previous data and go to the appropriate "Special procedure" at Stage 4.

***Step 2.7: Correct the measurement M***

If it can be demonstrated that M is wrong, make corrections (eg. repeat the measurement) and return to Step 2.1.

### 8.2.4 *Stage 3. Standard evaluation procedure at Level 1 [Fig. 8.3]*

Having determined the measured value (M) to be due to a new intake, the intake and dose are evaluated from the net value (N) using a priori parameters. This standard evaluation procedure should be applied only for routine monitoring.

***Step 3.1*** : If the measured value is not due to routine monitoring, special evaluation procedures (Stage 4) are needed anyway.

***Step 3.2*** : The pathway of intake is identified. In routine monitoring situations the pathway will most likely be inhalation, but it could also be ingestion or a combination of inhalation and ingestion. However, ingestion should be assumed only in those cases where

there is clear evidence for this pathway (well established and documented). Otherwise the inhalation pathway should be assumed.

***Step 3.3*** : Case or site specific parameter values should be assigned as far as they are available. Such a priori information needs to be well established and documented. Examples might include the Activity Median Aerodynamic Diameter (AMAD) – if it has been determined by appropriate air sampling (eg., cascade impactor), or the time of intake, if potential exposure was limited, or an incident was known to occur. Otherwise the following default parameter values should be used:

- Mode of intake: Single intake
- Time of intake: Mid-point of the monitoring interval, i.e. the mid-point of the time range between the date of the measurement being considered and the date of either the previous measurement or the beginning of monitoring
- Inhalation:

- Absorption Type and $f_A$ value: defaults according to OIR Publication (in preparation), Appendix OIR-27 (for examples, see Table 27.2). If the compound is unknown, then the Type for "unspecified compounds" should be used. For uranium, "unspecified compounds" are not listed, and it is proposed that Type M is assumed in the absence of specific information, as in ICRP Publication 71.
- Particle size: 5 um AMAD

- Ingestion:
  - $f_A$ value: defaults according to OIR Publication (in preparation), Appendix OIR-27 (see for example Table 27.2).

***Step 3.4*** : Using the assigned a priori parameter values, the intake is estimated by dividing the net value (N) by the appropriate retention or excretion function. Using the same assigned a priori parameter values the committed effective dose is calculated by multiplying the evaluated intake by the appropriate dose coefficient (dose per unit intake). Alternatively the dose per unit content approach can be applied to calculate the dose directly. When doing so, only the dose can be recorded.

***Step 3.5*** : The effective dose estimated in step 3.4 is added to the doses due to previous intakes in the accounting year. If the sum (annual dose) is less than 1 mSv, there is no need for further investigation (Step 3.5.1). Otherwise special procedures (Stage 4) are needed for more detailed evaluation of the case.

***Step 3.5.1*** : The results in terms of intake and committed effective dose from Step 3.5 are recorded together with the corresponding parameter values from Step 3.3.

### 8.2.5 *Stage 4. Identification of pathway of intake for special evaluation above Level 1 [Fig. 8.4]*

Special procedures are needed for the evaluation when there is evidence for an internal committed effective dose of more than 1 mSv or in all cases of special monitoring. In all these cases the evaluation procedures depend to some extent on the pathway of intake. Thus, in Stage 4 the pathway of intake has to be identified.

***Step 4.1*** : In many cases there is evidence for pure inhalation, as for example if room air contamination has been detected without detectable external contamination of the person under investigation. In those cases the special procedure for inhalation cases should be applied (Stage 5).

***Step 4.2*** : In a few cases there might be evidence for pure ingestion, as for example if contamination of the person or the working place has been detected, but not any contamination of the room air. In those cases the special procedure for ingestion cases should be applied (Stage 6). In cases where both contamination of the person or the working place and contamination of the room air is detected the pathway could be a combination of inhalation and ingestion. Some of those cases may be analysed assuming inhalation of aerosols with large particle size. If this assumption does not result in acceptable model consistency such cases may be analysed as a mixture of inhalation and ingestion. This procedure is described in detail in the IDEAS General Guidelines (Doerfel et.al. 2006).

***Step 4.3*** : In any case of a contaminated wound a special procedure would have to be developed to calculate the dose due to the intake via the wound (Stage 7). In cases where both contamination of the wound and contamination of the room air is detected the pathway could be a combination of inhalation and absorption from the wound. Such cases may be analysed as a mixture of inhalation and wound absorption

similar to the procedure for a mixture of inhalation and ingestion as described in the IDEAS General Guidelines (Doerfel et al, 2006).

***Step 4.4*** : In other cases there might be evidence for direct systemic intake by injection or skin absorption. These pathways are not considered by the guidelines at the present state (Step 4.4.1). If there is no evidence for one of the intake patterns above, then the evaluation should be started on the default assumption of pure inhalation, because this results in a conservative dose assessment.

### 8.2.6 *Stage 5. Special procedure for inhalation cases above Level 1 [Fig. 8.5]*

**Overview**

The special procedure is grouped in three subsequent stages (see overview flowchart Fig 8.5). In the first stage (5A), a simple evaluation is carried out using parameter values chosen a priori before the evaluation is carried out. The procedure is very similar to the "Standard procedure" (Stage 3). The main difference is that in a special procedure there should be more than one measurement.

In the second stage (5B), procedures are applied for varying the two main factors related to the inhaled material: the AMAD and absorption Type, and also the time of intake, if not known, using the measurement data (a posteriori).

In the third stage (5C), an advanced evaluation is carried out. It applies to cases where there are comprehensive data available. The fundamental approach of this stage is that the model parameter values are adjusted systematically, in a specific order, until the goodness of fit is acceptable (i.e. the fits obtained to all the data are not rejected by the specified criteria).

The details on stages 5A, 5B and 5C are given in Annex G

### 8.2.7 *Stage 6. Special procedure for ingestion cases above Level 1 [Figure 8.6]*

**Overview**

The special procedure is analogous to that for inhalation (Section 8.2.6) and there is, as a result a certain amount of repetition of that section here. It is grouped in three subsequent stages (see overview flowchart, Figure 8.9). In the first stage (6A), a simple evaluation is carried out using parameter values chosen a priori: before the evaluation is carried out. The procedure is very similar to the "Standard procedure" (Stage 3). The main difference is that in a special procedure there should be more than one measurement.

In the second stage (6B), procedures are applied for varying the main factor related to the ingested material: alimentary tract transfer factor $f_A$, and also the time of intake, if not known, using the measurement data (a posteriori).

In the third stage (6C), an advanced evaluation is carried out. It applies to cases where there are comprehensive data available. The fundamental approach of this stage is that the model parameter values are adjusted systematically, in a specific order, until the goodness of fit is acceptable (i.e. the fits obtained to all the data are not rejected by the specified criteria).

The details on stages 6A, 6B and 6C are given in Annex G.

### 8.2.8 *Stage 7. Special procedure for wound cases [Figure 8.7]*

**Overview**

The special procedure for wound cases takes into account that – contrary to those for inhalation and ingestion (Sections 8.2.6 and 8.2.7) – there is in principal the possibility of reducing the dose by excision of the contaminated tissue. Another principal difference is that in most wound cases the time of intake is known. Thus the special procedure for wound cases is substantially different from the others. It is grouped in three subsequent stages. In the first stage (7A), a simple evaluation is carried out based on the initial *in vivo* counting data using parameter values chosen *a priori* before the evaluation is carried out. The main goal of this stage of the procedure is to inform decisions as to whether there is a need for reducing the dose by excision of the contaminated tissue (Stage 7C).

In the second stage (7B), procedures are applied for selecting the wound absorption type by comparison of the NCRP model predictions with the data. At this stage more data relevant to the expected dose are needed, such as urine excretion rates or the activity in the regional lymph nodes.

The third stage (7C) deals with some special aspects of the excision of the contaminated tissue at the wound site.

The details on the more advanced stages 7A, 7B and 7C are given in Annex G.

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# 10 GLOSSARY

## Terms for General Dosimetry

Absorbed Dose

The physical dose quantity, which is given by

$$D = \frac{d\bar{\varepsilon}}{dm}$$

where $d\bar{\varepsilon}$ is the mean energy imparted by ionising radiation to the matter in a volume element and d*m* is the mass of the matter in this volume element. The SI unit for absorbed dose is joule per kilogram (J $kg^{-1}$) and its special name is gray (Gy).

Absorbed Fraction ($AF(T \leftarrow S)_R$)

The fraction of energy emitted as a specified radiation type *R* in a specified source region *S,* which is absorbed in a specified target tissue *T*.

Annual Limit on Intake (ALI)

The *ALI* was defined in Publication 60 (ICRP 1991, para S30) as an intake (in Bq) of a radionuclide in a year which would lead to a committed effective dose of 20 mSv (0.02 Sv). The average annual limit on effective dose for workers is thus:

$$ALI_j = \frac{0.02}{e_j(50)}$$

Biological half-time

The time required for a biological system to eliminate, by natural processes, half the amount of a substance, (eg. radioactive material) that has entered it.

Becquerel (Bq)

The special name for the SI unit of activity, 1 Bq = 1 $s^{-1}$.

Cells Near Bone Surfaces

those tissues which lie within 10 µm of endosteal surfaces and of bone surfaces lined with epithelium.

Committed Effective Dose ($E(\tau)$)

The sum of the products of the committed organ or tissue equivalent doses and the appropriate organ or tissue weighting factors ($w_T$), where τ is the integration time in years following the intake. The integration time is 50 y for adults, and from intake to age 70 y for children.

Committed Equivalent Dose ($H_T(\tau)$)

The time integral of the equivalent dose rate in a particular tissue or organ that will be received by an individual following intake of radioactive material into the body, where τ is the integration time in years following the intake. The integration time is 50 y for adults, and from intake to age 70 y for children.

Dose Coefficient

Committed tissue equivalent dose per unit intake at age $t_o$, $h_T(\tau)$, or committed effective dose per unit intake, $e(\tau)$, where τ is the time period in years over which the dose is calculated i.e. 50 y for adults and $(70–t_o)$ y for children. Note that elsewhere the term “dose per unit intake (DPUI)” is sometimes used for dose coefficient.

Dose Per Unit Exposure

Committed tissue equivalent dose per unit exposure at age $t_o$, or committed effective dos per unit exposure, calculated over 5 y for adults and (70-$t_o$) y for children.

Derived Air Concentration (DAC)

The *DAC* is the activity concentration in air in Bq/m$^3$ of the radionuclide considered which would lead to an intake of an *ALI* assuming a breathing rate of 1.2 m$^3$ h$^{-1}$ and an annual working time of 2000 h. Then the *DAC* is given by:

$$DAC_j = \frac{ALI_j}{2400}$$

Effective Dose (*E*)

The sum of the weighted equivalent doses in all tissues and organs of the body, given by the expression:

$$E = \sum_T w_T H_T$$

where $H_T$ is the equivalent dose in tissue or organ, *T*, and $w_T$ is the weighting factor for tissue *T*.

Endogenous Excretion

Term used to specify the excretion of materials from body fluids to the gastrointestinal (GI) tract, applying to biliary excretion and passage of materials through the GI tract wall.

Equivalent Dose ($H_T$)

The equivalent dose, $H_{T,R}$, in tissue or organ *T* due to radiation *R*, is given by:

$$H_{T,R} = w_R D_{T,R}$$

where $D_{T,R}$ is the average absorbed dose from radiation *R* in tissue *T* and $w_R$ is the radiation weighting factor. Since $w_R$ is dimensionless, the units for equivalent dose are the same as for absorbed dose, J kg$^{-1}$, and its special name is sievert (Sv). The total equivalent dose, $H_T$, is the sum of $H_{T,R}$ over all radiation types

$$H_T = \sum_R H_{T,R}$$

Exposure

In the context of inhalation, the product of the air concentration of a radionuclide to which a person is exposed (Bq m$^{-3}$) and the time of exposure. More generally, when the air concentration varies with time, the time integral of the air concentration of a radionuclide to which a person is exposed, integrated over the time of exposure.

Fractional Absorption in the Gastrointestinal Tract ($f_1$)

The $f_1$ value is the fraction of an element directly absorbed from the gut to body fluids.

Fractional absorption in the Alimentary Tract ($f_A$)

The $f_A$ value is the fraction of an element directly absorbed from the alimentary tract to body fluids as defined in the HATM.

Gray (Gy)

The special name for the SI unit of absorbed dose: 1 Gy = 1 J kg$^{-1}$.

Human Alimentary Tract Model (HATM)

Biokinetic model for describing the movement of ingested materials through the alimentary tract; published in ICRP Publication 100 (2006).

Human Respiratory Tract Model (HRTM)

Biokinetic model for describing the deposition, translocation and absorption of inhaled materials in the human lung; published in ICRP Publication 66.

Intake

Activity that enters the respiratory tract or gastrointestinal tract from the environment.

- Acute intake - a single intake by inhalation of ingestion, taken to occur instantaneously.
- Chronic intake - an intake over a specified period of time.

Organ Dose

The tissue- or organ-average absorbed dose $D_T$, is given by:

$$D_T = \frac{\varepsilon_T}{m_T}$$

where $\varepsilon_T$ is the total energy imparted in a tissue or organ $T$ and $m_T$ is the mass of that tissue or organ.

Radiation Weighting Factor ($w_R$)

The radiation weighting factor is a dimensionless factor to derive the equivalent dose from the absorbed dose averaged over a tissue or organ and is based on the quality of radiation (ICRP, 1991).

Red Bone Marrow (active)

The component of marrow which contains the bulk of the haematopoietic stem cells.

Reference Man

A person with the anatomical and physiological characteristics defined of in the report of the ICRP Task Group on Reference Man (Publication 89, ICRP, 2002).

Reference Value

The value of a parameter recommended by ICRP for use in a biokinetic model in the absence of more specific information, i.e. the exact value used to calculate the dose coefficients issued by ICRP. Reference values are given to sufficient precision for calculational purposes. This may be more precise than the biological data would support. In some cases corresponding values are given in *ICRP Publication 66* with less precision.

Relative Dose

The ratio of the dose coefficient calculated using specific information for one or more parameter values, to the corresponding dose coefficient given in an ICRP report (typically *ICRP Publication 68* or *72*) calculated using reference values for all parameters.

Sievert (Sv)

The special name for the SI unit of equivalent dose and effective dose: 1 Sv = 1 J $kg^{-1}$.

Source Region (S)

Region within the body containing the radionuclide. The region may be an organ, a tissue, the contents of the gastrointestinal tract or urinary bladder, or the surfaces of tissues as in the skeleton and the respiratory tract.

Specific Effective Energy ($SEE(T \leftarrow S)_R$)

The energy, suitably modified by the radiation weighting factor, imparted per unit mass of a target tissue, *T*, as a consequence of the emission of a specified radiation, *R*, from transformations occurring in the source region *S* expressed as Sv $(Bq\ s)^{-1}$.

Subcutaneous tissue

Loose fibrous tissue situated directly below the skin. It includes blood vessels, connective tissue, muscle, fat and glands. In the context of intake through wounds, it represents tissue at the wound site in which radionuclides could be retained prior to removal of soluble or dissolved material to blood or insoluble material via lymphatic vessels.

Target Tissue

Tissue or organ in which radiation is absorbed.

Tissue Weighting Factor ($w_T$)

The factor by which the equivalent dose in a tissue or organ is weighted to represent the relative contribution of that tissue or organ to the total detriment resulting from uniform irradiation of the body (ICRP, 1991).

Transfer Compartment

The compartment introduced for mathematical convenience into many of the biokinetic models used by ICRP to account for the translocation of the radioactive material through the body fluids from where they are deposited in tissues.

Uptake

Activity that enters the body fluids from the respiratory tract or gastrointestinal tract or through the skin.

**Terms for Human Respiratory Tract Model (HRTM) (ICRP, 1994a)**

Absorption
: Transfer of material to body fluids regardless of mechanism. Generally applies to dissociation of particles and the uptake into body fluids of soluble substances and material dissociated from particles.

Aerodynamic diameter ($d_{ae}$)
: Diameter (µm) of a unit density (1 g $cm^{-3}$) sphere that has same terminal settling velocity in air as the particle of interest.

Alveolar-Interstitial Region (AI)
: Consists of the respiratory bronchioles, alveolar ducts and sacs with their alveoli, and the interstitial connective tissue; airway generations 16 and beyond.

AMAD
: Activity Median Aerodynamic Diameter. Fifty percent of the activity in the aerosol is associated with particles of aerodynamic diameter ($d_{ae}$) greater than the AMAD. Used when deposition depends principally on inertial impaction and sedimentation, typically when the AMAD is greater than about 0.5 µm.

AMTD
: Activity Median Thermodynamic Diameter. Fifty percent of the activity in the aerosol is associated with particles of thermodynamic diameter ($d_{th}$) greater than the AMTD. Used when deposition depends principally on diffusion, typically when the AMAD is less than about 0.5 µm.

Basal cells
: Cuboidal epithelial cells attached to the basement membrane of extrathoracic and bronchial epithelium and not extending to the surface.

Bronchial Region (BB)
: Consists of the trachea (generation 0) and bronchi, airway generations 1 through 8.

Bronchiolar Region (bb)
: consists of the bronchioles and terminal bronchioles; airway generations 9 through 15.

Deposition Classes of Gases and Vapours According to Solubility and Reactivity:

- Class SR-0 Insoluble and nonreactive. Negligible deposition in the respiratory tract.
- Class SR-1 Soluble or reactive. Deposition throughout the respiratory tract, which may be complete or incomplete.
- Class SR-2 Highly soluble or reactive. Complete deposition in the respiratory tract with instantaneous uptake to body fluids.

Clearance
: The removal of material from the respiratory tract by particle transport and by absorption into body fluids.

Compartments in the Particle Transport Model Representing Retention of Material in each Region Defined in the Respiratory Tract Model:

- $AI_1$ relatively short-term retention (half-time, $t_{½}$ about 35 d) of a fraction, taken to be 0.3, of the deposit in the alveolar-interstitial region.
- $AI_2$ long-term retention ($t_{½}$ about 700 d) of a fraction, taken to be 0.6, of the deposit in the alveolar-interstitial region.
- $AI_3$ very long-term retention ($t_{½}$ about 6000 d) of a fraction, taken to be 0.1, of the deposit in the alveolar-interstitial region.
- $BB_1$ short-term retention ($t_{½}$ about 100 minutes) of particles in the bronchial region: the particles are removed by rapid mucociliary clearance.
- $bb_1$ short-term retention ($t_{½}$ about 8 hours) of particles in the bronchiolar region: the particles are removed by rapid mucociliary clearance.
- $BB_2$ intermediate retention ($t_{½}$ about 20 d) of particles in the bronchial region.
- $bb_2$ intermediate retention ($t_{½}$ about 20 d) of particles in the bronchiolar region.
- $BB_{seq}$ long-term retention ($t_{½}$ about 70 d) in airway walls of a small fraction of the particles deposited in the bronchial region.
- $bb_{seq}$ long-term retention ($t_{½}$ about 70 d) in airway walls of a small fraction of the particles deposited in the bronchiolar region.
- $ET_1$ retention of material deposited in the anterior nose (region $ET_1$, which is not subdivided).
- $ET_2'$ short-term retention ($t_{½}$ about 10 minutes) of the material deposited in the posterior nasal passage, larynx, pharynx and mouth (region $ET_2$), except for the small fraction, taken to be 0.0005, retained in $ET_{seq}$. (In *ICRP Publication 66* this *compartment* was labelled $ET_2$. It is here, as in *ICRP Publication 71*, labelled $ET_2'$ to distinguish it from the *region* $ET_2$ which also includes *compartment* $ET_{seq}$.)
- $ET_{seq}$ long-term retention ($t_{½}$ about 700 d) in airway tissue of a small fraction of particles deposited in the nasal passages.
- $LN_{ET}$ lymphatics and lymph nodes that drain the extrathoracic region.
- $LN_{TH}$ lymphatics and lymph nodes that drain the thoracic region.

Deposition

Refers to the initial processes determining how much of the material in the inspired air remains behind in the respiratory tract after exhalation. Deposition of material may occur during both inspiration and exhalation.

Extrathoracic (ET) Airways

Consists of anterior nose ($ET_1$) and the posterior nasal passages, mouth, pharynx and larynx ($ET_2$).

Functional Residual Capacity

The volume of gas remaining in the lungs at the resting expiratory level (ICRP, 1975, p. 349).

Habitual Mouth Breather

A person who breathes oro-nasally (partly through the nose and partly through the mouth) at all levels of exercise: “sleep”, “sitting” “light exercise” and “heavy exercise”. At “heavy exercise” such a person inhales a greater fraction of air through the mouth than a Nasal Augmeter.

Habitual Nose Breather
: A person who breathes entirely through the nose at the exercise level of "heavy exercise" as well as at "sleep", "sitting" and "light exercise". Such a person may switch to breathing oro-nasally (partly through the nose and partly through the mouth), but at a ventilation rate greater than the reference value for heavy exercise (3 $m^3$ $h^{-1}$).

Inhalability
: Fraction of particles that enters the nose and mouth, of those present in the volume of ambient air before inspiration.

Nasal Augmeter
: A person who breathes entirely through the nose at the exercise levels of "sleep", "sitting" and "light exercise", but oro-nasally (partly through the nose and partly through the mouth) during "heavy exercise". Also known as a "normal nose breather", because most people breathe according to this pattern. All reference subjects, including the Reference Worker are assumed to be Nasal Augmenters.

Normal Nose Breather
: See Nasal Augmeter.

Particle Transport
: Processes that clear material from the respiratory tract to the GI tract and to the lymph nodes, and move material from one part of the respiratory tract to another.

Secretory Cells
: Nonciliated epithelial cells that have mucous or serous secretions.

Target tissues in the Bronchial region of the Respiratory Tract Model:
: (See Tables 9 and 10. For each of the other regions only one target tissue is specified and hence no special symbol is required.)
$BB_{bas}$ tissue in bronchial region through which basal cell nuclei are distributed.
$BB_{sec}$ tissue in bronchial region through which secretory cell nuclei are distributed.

Thermodynamic Diameter ($d_{th}$)
: Diameter (μm) of a spherical particle that has the same diffusion coefficient in air as the particle of interest.

Thoracic (TH) Airways
: Combined bronchial, bronchiolar and alveolar-interstitial regions.

Types of Materials According to their Rates of Absorption from the Respiratory Tract to Body Fluids:

| | |
|---|---|
| Type F | deposited materials that are readily absorbed into body fluids from the respiratory tract. (Fast absorption.) |
| Type M | deposited materials that have intermediate rates of absorption into body fluids from the respiratory tract. (Moderate absorption.) |
| Type S | deposited materials that are relatively insoluble in the respiratory tract. (Slow absorption.) |
| Type V | deposited materials that, for dosimetric purposes, are assumed to be instantaneously absorbed into body fluids from the respiratory tract: applied in *ICRP Publications 71* and *72* (ICRP, 1995, 1996) only to certain gases and vapours. (Very fast absorption.) |

**Terms for Human Alimentary Tract Model (HATM)**

Alimentary tract
: The tube from mouth to anus in which food is digested.

Alimentary tract transfer factor ($f_A$)
: The alimentary tract transfer factor ($f_A$) is the fraction of activity entering the alimentary tract that is absorbed to blood, taking no action of losses due to radioactive decay or endogenous input of activity into the tract.

Caecum
: Large cul-de-sac at the beginning of the large intestine, continuous with the ascending colon.

Colon
: The colon is the largest part of the large intestine and can be regarded as comprising four regions: ascending, transverse, descending and sigmoid. It is the final site of fluid and electrolyte absorption in addition to absorption of short-chain fatty acids. The presence of bacteria in the colonic lumen is important for the digestion of dietary fibre.

Digestion
: Process of breakdown, absorption and utilisation of ingested material which is initiated in the mouth by mastication and salivary secretions, and continues along the length of the alimentary tract. Involves the breakdown of large insoluble food molecules into soluble constituents that can be absorbed through the intestinal epithelium.

Duodenum
: The duodenum is the initial part of the small intestine which begins at the pylorus and is around 25 cm long. It is devoid of mesentery glands and contains Brunner's glands as well as ducts that allow entry of pancreatic and hepatic secretions.

Faeces
: Eliminated waste products of digestion. Faeces contain material from sloughed intestinal cells, micro-organisms, and excretory materials, as well as undigested food materials.

Ileum
: The third and final region of the small intestine. It is continuous with, and difficult to delineate from, the jejunum, although villi are shorter and less numerous, and Peyer's patches are larger and more numerous.

Jejunum
: The second region of the small intestine, following from the duodenum and continuous with the ileum. The villi are larger and more numerous than in the ileum, and there are circular folds (valves of Kerring) which are large. These folds retard the digesta and provide an increased area for absorption. This section of the small intestine has the largest absorptive surface area in addition to a thick muscular wall.

Large intestine
: The large intestine is around 1 m long in the adult. It begins at the ileocaecal valve and consists of the caecum and appendix; the ascending, transverse, descending, and sigmoid colon; and the rectum.

Oesophagus
: Tabular part of the alimentary tract that connects the mouth and stomach.

Sphincters exist at both ends; the upper oesophageal sphincter which prevents entry of air into the oesophagus, and the lower sphincter which prevents reflux of gastric contents.

Pharynx

The pharynx, situated behind the nasal cavities, mouth, and larynx, is a connecting tube. The upper region above the soft palate, the nasopharynx, connects to the nasal passages. The mid region, the oropharynx, connects to the mouth. The laryngeal region connects to the larynx and also continues behind the larynx to the oesophagus.

Rectum

Continuous with the sigmoid colon. It stores faeces prior to defecation. The rectal mucosa has both longitudinal and horizontal folds.

Salivary glands

Exocrine glands that secrete saliva containing both organic (mucus, amylase) and inorganic ($Na^+$, $Cl^-$, $K^+$ and $HCO_3^-$) constituents. They are the parotid (near the ear), sublingual or submaxillary (under the tongue), and submandibular or submaxillary (under the mandible) glands.

Small intestine

Comprises the duodenum, jejunum, and ileum.

Stomach

Enlarged section of the alimentary tract that receives, stores, mixes and digests food. Acid and enzymes are secreted into the lumen for digestion of ingested material, which is then broken down to form chyme. The stomach is divided into here sections: fundus, corpus, and antrum.

**Terms for Bioassay Interpretation**

Activity

Physical quantity for the number of disintegrations per unit time (s) of a radioactive material. The SI-unit of the activity is Becquerel (Bq): 1 Bq = 1 $s^{-1}$

Bioassay

Any procedure used to determine the nature, activity, location or retention of radionuclides in the body by direct (in vivo) measurement or by indirect (in vitro) analysis of material excreted or otherwise removed from the body.

Biokinetic model

A mathematical model describing the intake, uptake and retention of a radionuclide in various organs or tissues of the body and the subsequent excretion from the body by various pathways.

Biokinetic function

A mathematical function describing the time course of the activity in the body (retention function) or the activity excreted via urine or faeces (excretion function) following a single intake at time t = 0. In general, the retention functions represent the body or organ activity at the time t after the intake, whereas the excretion functions represent the integral of the excretion rate from t – 1d until t.

Compartment

Pool of radioactive materials in the body which can be characterised by first order kinetics; a compartment can be an organ (as for example the liver), a part of an organ (as for example the RES of the liver), a tissue (as for example the bone), a part of a tissue (as for example the bone surface) or another substance of the body (as for example the body fluids)

Critical intake

Intake by inhalation of a radioactive material resulting in a committed effective dose of 20 mSv; the critical intake is defined as 20 mSv divided by the dose coefficient for inhalation of 5 µm AMAD aerosols with the respective absorption type.

Direct measurement

Generic term for any kind of in vivo measurement of incorporated radionuclides (i.e. whole body counting, lung counting, thyroid counting etc.)

Evaluation procedure

Procedure for the assessment of internal dose from incorporation monitoring data. The guideance refers to the following procedures:

Reference procedure
Individual procedure using standard biokinetic models
Individual procedure using modified biokinetic models
Integration procedure

Excretion analysis

Procedure for the assessment of the activity in the urine or faeces or in the exhaled air. The excretion analysis includes radiochemical separation, preparation of measuring samples and the evaluation of the measuring samples by spectrometric or other techniques (i.e. $\alpha$-spectrometry or ICP-MS)

Excretion rate

In general, the excretion rate is the amount of activity which is excreted via urine or faeces during 24 hours, with the decay of the radionuclide having been corrected for the end of the 24 hour sampling period. A special case is HTO where the excretion rate in general is given in terms of the activity concentration in the excreted material.

Investigation level

Investigation levels have been defined by ICRP (1997b) as levels above which the cause or the implications of them should be examined. They are therefore used retrospectively. Investigation levels can be set for any operational parameter related to the individual or to the working environment. For individual monitoring of exposure to intakes of radionuclides, they are most likely to relate to a measured body or organ/tissue content, an activity level in excreta, or an air concentration measured by a personal air sampler.

Measured quantity (*M*)

Primary result of incorporation monitoring; the measured quantity represents in the case of *in vivo* measurements the whole body, organ or tissue activity in terms of Bq and in the case of *in vitro* measurements the daily excretion rate in terms of Bq/d or the activity concentration in terms of Bq/l or the specific activity in terms of Bq/kg. The guidelines refer especially to the folowing kind of measured quantity:

*In vivo* measurement

Lung activity
Thyroid activity
Liver activity

Skeleton activity
Wound site activity
*In vitro* measurement
Urine activity excretion rate
Faeces activity excretion rate
Urine activity concentration
Breathed air activity concentration

Minimum detectable activity (MDA)

The minimum detectable activity (frequently also referred to as *detection limit* or *lower limit of detection*) is an a priori calculated value, which specifies the minimum body contribution that can be detected by a defined measurement procedure. The detection limit is complementary to the decision threshold, i.e. when considering the detection limit the wrong decision that there exists only a background effect when there is in fact a contribution from the body (Type II error), occurs with a well-defined probability $\beta$. Thus, the detection limit is closely related to the decision threshold defined by the Type I error probability $\alpha$. By definition the detection limit is given in terms of body or organ activity and it can be compared directly with guideline values.

Minimum significant activity (MSA)

The minimum significant activity (frequently also referred to as *decision threshold* or *critical level*) is an a posteriori calculated value at which the decision can be made, whether the registered pulses include contributions from the measured sample or are solely due to background. If this decision rule is observed, a wrong decision that there is a contribution from the measured sample when actually only a background effect exists (Type I error), occurs with a well-defined probability $\alpha$. By definition the decision threshold is given in terms of pulses but for practical application it is frequently transferred to the corresponding activity value.

Occupational exposure

Exposure to radiation incurred at work as the result of situations that can reasonably be regarded as the responsibility of the operating management.

Operational Quantities

These are used in monitoring and practical applications for investigating the situations involving external exposure and intakes of radionuclides.

Protection Quantities

values that ICRP has developed for radiological protection that allow quantification of the extent of exposure to ionising radiation from both whole and partial body external irradiation and from intakes of radionuclides.

Reference levels

Values of measured quantities above which some specified action or decision should be taken. They include:
Recording levels, above which a result should be recorded, lower values being ignored;
Investigation levels, above which the cause or the implication of the result should be examined;
Action levels, above which some remedial action should be considered

Reference procedure

General procedure for the evaluation of a single measured quantity from routine incorporation monitoring; the reference procedure The reference procedure assumes the following reference parameters for the intake:
Type: single intake

| | |
|---|---|
| Time: | constant chronic rate throughout the monitoring interval |
| Pathway: | inhalation |
| Absorption type: | conservative or operative absorption type, respectively |
| Particle size: | 5 μm AMAD or operative particle size |

Significance category

Classification of the measured quantities with respect to the evidence of a new intake; there are three significance categories:

SC(1) the net measured quantity is more than the total uncertainty: there is evidence for a new intake

SC(0) the absolute net measured quantity is less than the total uncertainty: there is no evidence for a new intake

SC(-1) the negative net measured quantity is more than the total uncertainty: there is a contradiction to the previous evaluations

Tracer procedure

Special procedure in internal dosimetry using a tracer radionuclide for the assessment of the intake of a mixture of radionuclides; in the first step the intake of the tracer radionuclide is derived from the respective monitoring data; in the second step the intake of the other radioclides is calculated taking into account the nuclide vector; in the case of inhalation the particle size of the leading radionuclide is applied also for the other radionuclides of the mixture; the absorption of the other radionuclides should be identical or close to that of the leading radionuclide. In some cases (as for example Am-241 as tracer for a plutonium mixture) it may be necessary to adopt the predominant absorption type of the nuclides in the mixture to the absorption type of the leading radionuclide; in the case of ingestion the fractional absorption in the alimentary tract should be handled in the same way as the absorption type in the case of inhalation.

Tracer radionuclide

Radionuclide, which can be easily detected and which can be considered to be representative for a mixture of radionuclides with respect to the deposition, retention and excretion in and from the body. Incorporation monitoring and internal dosimetry based on tracer radionuclide requires knowledge of the isotopic composition of the mixture (nuclide vector).

**Draft – 21 February 2006**

# INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION

## COMMITTEE 2

## Supporting Guidance Document
## Interpretation of Bioassay Data

## TABLES AND FIGURES

**Prepared by INDOS and DOCAL Task Groups**

**DRAFT DOCUMENT**

## Table 1.1 Table of recent ICRP publications on occupational exposure to radionuclides

| ICRP Publication No. (year) | Application | Intake | Contents |
|---|---|---|---|
| 30 (1979 etc) Parts 1-4 | Workers | Inhalation and ingestion | Limits for intakes of radionuclides by workers. Implementation of recommendations in Publication 26. |
| 54 (1988) | Workers | Inhalation and ingestion | Individual monitoring for intakes of radionuclides by workers: design and interpretation. Based on Publication 30 models. |
| 61 (1990) | Workers | Inhalation and ingestion | Annual limits on intake of radionuclides by workers based on the 1990 recommendations. Implementation of Publication 60 recommendations. Superceded by Publication 68 |
| 66 (1994) | Public and workers | Inhalation | Description of Human Respiratory Tract Model (HRTM) |
| 68 (1994) | Workers | Inhalation and ingestion | Effective dose coefficients for workers, for about 800 radionuclides: selected radioisotopes of the 91 elements covered in ICRP Publication 30, Parts 1–4. Applies the Human respiratory Tract Model, HRTM.The inhalation dose coefficients for workers exposed to $^{226}$Ra given in ICRP Publication 68 were revised in Annexe B of ICRP Publication 72. |
| 78 (1997) | Workers | Inhalation and ingestion | Individual monitoring for internal exposure of workers. Based upon Publication 68 models for selected radionuclides |
| CD-ROM (1998) | Public[a] and workers | Inhalation and ingestion | A database of equivalent doses to individual tissues corresponding to the effective dose coefficients in ICRP Publications 68 and 72. Inhalation dose coefficients for 10 particle sizes. |
| 88 (2001) | Embryo and fetus | Inhalation and ingestion by the mother | Dose coefficients for the offspring for intakes by the mother (worker or public) before or during pregnancy. Covers radionuclides of the 31 elements covered in the Publication 56 series. |
| 89 (2002) | Public and workers | Inhalation and ingestion | Basic anatomical and physiological data for use in radiological protection |
| CD-ROM2 (2002) | Embryo and fetus | Inhalation and ingestion by the mother | Database of dose coefficients extending information on radionuclides in Publication 88. |
| Supporting Guidance 3 (2002) | Workers | Inhalation | Guide for the practical application of the ICRP Human Respiratory Tract Model |
| 95 (2005) | Public and Workers | Inhalation and ingestion | Doses to the infant from radionuclides in mothers' milk. Covers radionuclides of the 31 elements covered in the Publication 56 series plus Na, Mg, P and K. |
| 99 (2006) | Public and workers | Ingestion | Description of Human Alimentary Tract Model (HATM). |

## Table 3.1. AMADs for different industries

| Type of industry | Range of AMAD[(a)] | Median[(a)] (μm) |
|---|---|---|
| All work places | 0.12 - 25 | 4.4 |
| Nuclear power industry | 0.28 – 7.2 | 3.9 |
| Uranium mills | 0.5 - 16 | 6.8 |
| Fuel handling facilities | 0.34 – 16.5 | 3.8 |

(a) Values taken from Dorrian M-D and Bailey M R (1995) Particle size distributions of radioactive aerosols measured in workplaces. Radiat. Prot. Dosim. 60: 119-133.

(b) It should be noted that inhaled larger (e.g., AMAD > 15-20 μm) insoluble particles will be preferentially cleared through the GI tract, and so may present the appearance of an intake by ingestion. In this case, analysis of swabs of the nostrils and mouth may help to characterize the intake route.

## Table 3.2 Regional deposition of inhaled aerosols in Reference Worker (% of inhaled activity)

| **Region** | **Deposition (%) (5 μm) AMAD[a]** |
|---|---|
| $ET_1$ | 34 |
| $ET_2$ | 40 |
| BB | 1.8 |
| Bb | 1.1 |
| Al | 5.3 |
| Total | 82 |

a values are rounded

## Table 3.3 Default absorption parameter values for Type F, M, and S materials

| **Type** | | **F(fast)** | **M (moderate)** | **S (slow)** |
|---|---|---|---|---|
| Fraction dissolved rapidly | $f_r$ | 1 | 0.1 | 0.001 |
| Dissolution rates: | | | | |
| Rapid ($d^{-1}$) | $s_r$ | 100 | 100 | 100 |
| Slow ($d^{-1}$) | $s_s$ | - | 0.005 | 0.0001 |

## Table 7.1 Examples of critical monitoring values $M_C$ for some selected radionuclides and the respective monitoring procedures

| Radionuclide | Absorption type (chemical form) | Type of monitoring | Monitoring interval (d) | Critical monitoring value $M_C$ |
|---|---|---|---|---|
| H-3 | HTO | Urine[a] | 7<br>14<br>30 | 2000 Bq/l<br>3200 Bq/l<br>4100 Bq/l |
| Co-60 | M | Whole body | 90<br>180<br>360 | 160 Bq<br>230 Bq<br>290 Bq |
| Sr-90 | F | Urine[b] | 90<br>180<br>360 | 0.4 Bq<br>0.2 Bq<br>0.2 Bq |
| I-131 | F | Thyroid | 7<br>14<br>30 | 18 Bq<br>26 Bq<br>26 Bq |
| Cs-137 | F | Whole body | 90<br>180<br>360 | 1200 Bq<br>1800 Bq<br>2000 Bq |
| U-235 | S | Lungs | 90<br>180<br>360 | 0.2 Bq < LLD<br>0.3 Bq < LLD<br>0.5 Bq < LLD |
| Pu-239 | M | Urine[b] | 90<br>180<br>360 | 0.007 mBq < LLD<br>0.011 mBq < LLD<br>0.017 mBq < LLD |

a Results for tritiated water in urine are given in terms of activity concentration (Bq l-1)
b Urinary results are given in terms of daily excretion, Bq or mBq.

Note that there is growing interest in the application of the "dose per unit content" function, $z(t) = e(50)/m(t)$, which represents the committed effective dose per unit organ (body) radionuclide content or per unit radionuclide content in the 24-hour excreta sample at time $t$ after an acute intake (Chapter 1). Thus $E = M\ z(t)$, where $E$ is the committed effective dose, and $M$ is the measured value. Its use simplifies the dose evaluation to a single step, instead of the traditional method of first applying the retention or excretion function $m(t)$ to calculate the intake, and then the dose coefficient $e(50)$ to calculate the resulting effective dose. Hence in the equation above, $m(T/2)/\ e(50)$ could be replaced by $1/z(T/2)$.

## Table 7.2 Isotopic composition of natural uranium

| Isotope | % Isotopic composition[a] | % Alpha activity | Alpha activity[b] Bq/g |
|---|---|---|---|
| U-238 | 99.2745 | 48.26 | 1.23E+04 |
| U-236 | 0.0000 | 0.00 | 0.00E+00 |
| U-235 | 0.7200 | 2.25 | 5.76E+02 |
| U-234 | 0.0055 | 49.49 | 1.27E+04 |
| Total alpha activity, Bq/g | | | 2.56E+04 |
| Alpha activity ratio U-234/U-238 | | | 1.03 |
| Alpha activity ratio U-235/U-238 | | | 0.047 |

[a] Composition is given as weight % of total U isotopes
[b] Alpha activity per gram uranium
Firestone and Shirley (1998) for isotopic composition

**Table 7.3 Isotopic composition of enriched (3.5 %) uranium**

| Isotope | % Isotopic composition[a] | % Alpha activity | Alpha activity[b] Bq/g |
|---|---|---|---|
| U-238 | 96.471 | 14.78 | 1.20E+04 |
| U-236 | 0.0000 | 0.00 | 0.00E+00 |
| U-235 | 3.5000 | 3.45 | 2.80E+03 |
| U-234 | 0.02884 | 81.78 | 6.64E+04 |
| Total alpha activity, Bq/g | | | 8.12E+04 |
| Alpha activity ratio U-234/U-238 | | | 5.53 |
| Alpha activity ratio U-235/U-238 | | | 0.233 |

[a] Composition is given as weight % of total U isotopes
[b] Alpha activity per gram uranium
Firestone and Shirley (1998) for isotopic composition

**Table 7.4 Isotopic composition of enriched (92.8 %) uranium**

| Isotope | % Isotopic composition[a] | % Alpha activity | Alpha activity[b] Bq/g |
|---|---|---|---|
| U-238 | 6.06 | 0.039 | 7.51E+02 |
| U-236 | 0.34 | 0.428 | 8.16E+03 |
| U-235 | 92.8 | 3.89 | 7.42E+04 |
| U-234 | 0.79 | 95.64 | 1.82E+06 |
| Total alpha activity, Bq/g | | | 1.91E+06 |
| Alpha activity ratio U-234/U-238 | | | 2452 |
| Alpha activity ratio U-235/U-238 | | | 99.8 |

[a] Composition is given as weight % of total U isotopes
[b] Alpha activity per gram uranium
Hoover, Newton, Guilmette et al (1998) for isotopic composition

### Table 7.5 Isotopic composition of depleted uranium

| Isotope | % Isotopic composition[a] | % Alpha activity | Alpha activity[b] Bq/g | Mass fraction |
|---|---|---|---|---|
| U-238 | 99.8000 | 83.39 | 1.24E+04 | 0.997987 |
| U-236 | 0.0000 | 0.00 | 0.00E+00 | 0.000003 |
| U-235 | 0.2000 | 1.07 | 1.60E+02 | 0.002 |
| U-234 | 0.0010 | 15.53 | 2.31E+03 | 0.0001 |
| Total alpha activity, Bq/g | | | 1.49E+04 | |
| Alpha activity ratio U-234/U-238 | | | 0.186 | |
| Alpha activity ratio U-235/U-238 | | | 0.013 | |

[a] Composition is given as weight % of total U isotopes
[b] Alpha activity per gram uranium

### Table 7.6 Specific activities of radionuclides of uranium

| Nuclide | Half-life[(b,c)] (y) | Atomic mass[(d)] (u) | Specific activity (Bq/g) |
|---|---|---|---|
| U-234 | $(2.457 \pm 0.003)\ 10^5$ | 234.0409456 | 2.3003E+08 |
| U-235 | $(7.037 \pm 0.007)\ 10^8$ | 235.0439231 | 7.9973E+04 |
| U-238 | $(4.468 \pm 0.005)\ 10^9$ | 238.0507826 | 1.2437E+04 |
| U-236 | $(2.342 \pm 0.003)\ 10^7$ | 236.0455619 | 2.3928E+06 |

(a) Avogadro's number 6.02214E+23
(b) Decay Data of the Transactinium Nuclides", IAEA Technical Report Series no. 261 (1986) (as recommended by NPL)
(c) 1 year = 365.2422 days as stated in Decay Data of the Transactinium Nuclides", IAEA Technical Report Series no. 261 (1986).
(d) NIST website (May, 2001)

### Table 7.7 Isotopic composition of Pu and Am in spent nuclear fuel from reprocessing plant

| Isotope | % Isotopic composition, Pu+Am a | % Pu-Alpha activity | % Total-Alpha activity |
|---|---|---|---|
| Pu-238 | 0.30 | 38.47 | 38.47 |
| Pu-239 | 78.65 | 36.56 | 36.56 |
| Pu-240 | 14.64 | 24.95 | 24.95 |
| Pu-241 | 5.55 | - | - |
| Pu-242 | 0.860 | 0.02 | 0.02 |
| Pu-244 | 0 | 0 | 0 |
| Am-241 | 0 | - | 0 |

(a) Composition is given as weight % of total Pu isotopes + Am-241

**Table 7.8 Composition of spent Light Water Reactor fuel**

| Isotope | % Isotopic composition, Pu+Am b | % Pu-Alpha activity | % Total-Alpha activity | % Total activity |
|---|---|---|---|---|
| Pu-238 | 2.23 (0.69) | 81.53 | 72.86 | 2.75 |
| Pu-239 | 54.05 (3.08) | 7.17 | 6.41 | 0.24 |
| Pu-240 | 23.18 (0.67) | 11.25 | 10.05 | 0.38 |
| Pu-241 | 12.98 (1.42) | - | - | 96.23 |
| Pu-242 | 5.94 (1.32) | 0.05 | 0.04 | 0.0 |
| Am-241 | 1.62 (1.22) | - | 10.64 | 0.40 |
| Pu-241 activity/Total Pu a activity | | | | 29 |
| Pu-241 /(Pu-239+Pu-240) a activity | | | | 155 |
| Am-241 activity/Pu-241 activity | | | | 0.004 |

(a) Light Water Reactors consitute >80% of all commercial reactors, the remaining 20% being evenly divided between Pressurised Heavy Water Reactors (PHWR) of the CANDU type and Gas Cooled (Magnox) Reactors. This data is compiled from six sets of data covering various LWR sub types and reactor power outputs. The composition is for spent Low Enriched Uranium fuel (LEU). Data compiled from Martell, 1975, Bende, 1999, AEN-NEA, 1999, Bunn et.al., 2003

(b) Composition is given as mean atom % of total Pu isotopes + Am-241. Sample Standard Deviations in parentheses.

**Table 7.9 Typical values for the components of log-normal uncertainty for *in vivo* measurements of radionuclides emitting low, intermediate and high photon energy radiation.**

| Source of uncertainty (Type) | Log-normal scattering factor SF | | |
|---|---|---|---|
| | Low photon energy E < 20 keV | Intermediate photon energy 20 keV < E < 100 keV | High photon energy E > 100 keV |
| Counting statistics (A) | 1.5 | 1.3 | 1.07 |
| Variation of detector positioning (B) | 1.2 | 1.05 | < 1.05 |
| Variation of background signal (B) | 1.5 | 1.1 | < 1.05 |
| Variation in body dimensions (B) | 1.5 | 1.12 | 1.07 |
| Variation of overlaying structures (B) | 1.3 | 1.15 | 1.12 |
| Variation of activity distribution (B) | 1.3 | 1.05 | < 1.05 |
| Calibration (B) | 1.05 | 1.05 | 1.05 |
| Spectrum evaluation[1)] (B) | 1.15 | 1.05 | 1.03 |

1) HPGe detector spectra

**Table 7.10 Typical values for the total type A and type B log-normal uncertainty for *in vivo* measurements of radionuclides emitting low, intermediate and high photon energy radiation**

| Uncertainty type | Log. normal scattering factor SF | | |
|---|---|---|---|
| | Low photon energy E < 20 keV | Intermediate photon energy 20 keV < E < 100 keV | High photon energy E > 100 keV |
| Total type A | 1.5 | 1.3 | 1.07 |
| Total type B | 2.06 | 1.25 | 1.15 |
| Total | 2.3 | 1.4 | 1.2 |

**Table 7.11 Default values for the log. normal scattering factor SF for various types of measurement from different studies**

| Quantity | Log. normal scattering factor SF | Reference |
|---|---|---|
| True 24-hr urine | 1.1 | Moss et al 1969 |
| Simulated 24-hr urine, specific gravity normalization | 1.3 | (a) |
| Simulated 24-hr urine, creatinine normalization | 1.8 | Riddell et al 1994 |
| Spot urine sample | 2.0 | Moss et al 1969 |
| Fecal 72-hr sample | 3 | (b) |
| Fecal 24-hr sample | 5 | (b) |
| Spot Fecal sample | 7 | (b) |
| Chest count (see Table 8.19) | 1.2 to 2.3 | (b) |

(a) At Los Alamos, Type B uncertainties, in terms of the coefficient of variation, for urine samples normalised using volume and specific gravity has been found to be 30% (ie. a SF of 1.3)
(b) Values based on judgement and experience.

## Table 7.12 Number and type of data required for assessment of dose for some categories of radionuclides and the respective monitoring procedures

| | | Required monitoring data | | | | | |
|---|---|---|---|---|---|---|---|
| | | D < 1 mSv (minimum requirement) | | 1 mSv < D < 6 mSv | | D > 6 mSv | |
| Radionuclide category | Type of monitoring | Number | Time range[a] (days) | Number | Time range[a] (days) | Number | Time range[a] (days) |
| All type of α-emitters with significant γ-component (U-235, Am-241 etc.) | Urine | - | - | 2 | 30 | 3 | 60 |
| | Faeces | 1 | - | 2 | 30 | 3 | 60 |
| | Whole body, critical organ or wound site, resp. | - | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| All type of α-emitters without significant γ-component (Po-210, Pu-239 etc.) | Urine | - | - | 3 | 30 | 5 | 60 |
| | Faeces | 1 | - | 3 | 30 | 5 | 60 |
| All type of β-emitters with significant γ-component (Co-60, I-131, Cs-137 etc.) | Whole body, critical organ or wound site, resp. | 1 | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| | Urine | - | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| F-type β-emitters without significant γ-component (H-3, C-14 etc.) | Urine | 1 | - | 4 | $<T_{1/2}$eff | 8 | $>T_{1/2}$eff |
| M/S-type β-emitters without significant γ-component (Sr-90 etc.) | Urine | 1 | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| | Faeces | - | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| Pure γ-emitters (I-123 etc.) | Whole body or critical organ | 1 | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |
| | Urine | - | - | 2 | $<T_{1/2}$eff | 4 | $>T_{1/2}$eff |

a Time range in days after intake (if known) or after first detection of incorporated radionuclide

## Table 7.12a Number and type of data required for assessment of dose for some selected radionuclides and the respective monitoring procedures (more details given in OIR)

| Radionuclide | Type of | Required monitoring data |
|---|---|---|

| | monitoring | D < 1 mSv | | 1 mSv < D < 6 mSv | | D > 6 mSv | |
|---|---|---|---|---|---|---|---|
| | | Number | Time range (days) | Number | Time range (days) | Number | Time range (days |
| H-3 | Urine | 1 | - | 4 | 10 | 8 | 30 |
| Co-60 | Whole body | 1 | - | 2 | 30 | 4 | 60 |
| | Urine | - | - | 2 | 30 | 4 | 60 |
| Sr-90 | Urine | 1 | - | 2 | 20 | 4 | 40 |
| | Faeces | - | - | 2 | 20 | 4 | 40 |
| I-131 | Thyroid | 1 | - | 2 | 7 | 4 | 14 |
| | Urine | - | - | 2 | 7 | 4 | 14 |
| Cs-137 | Whole body | 1 | - | 2 | 100 | 4 | 180 |
| | Urine | - | - | 2 | 100 | 4 | 180 |
| U-235 | Urine | - | - | 2 | 30 | 3 | 60 |
| | Faeces | 1 | - | 2 | 30 | 3 | 60 |
| | Lungs | - | - | 2 | 60 | 4 | 120 |
| Pu-239 | Urine | - | - | 3 | 30 | 5 | 60 |
| | Faeces | 1 | - | 3 | 30 | 5 | 60 |
| Am-241 | Urine | - | - | 2 | 30 | 3 | 60 |
| | Faeces | 1 | - | 2 | 30 | 3 | 60 |
| | Lungs | - | - | 2 | 60 | 4 | 120 |
| | Skeleton[a] | - | - | 1 | 90 | 2 | 180 |

a These measurements are desirable if facilities are available

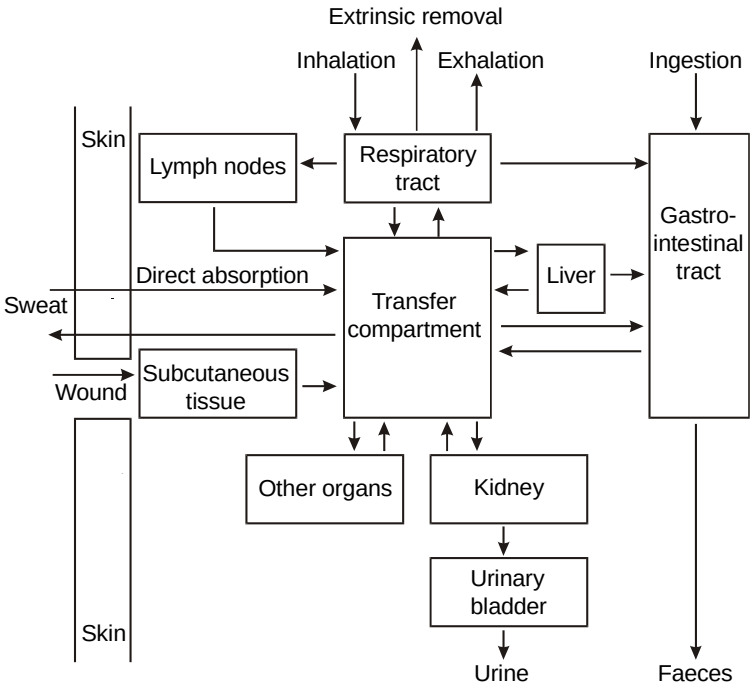



Fig. 3.1 Summary of the main routes in intake, transfers and excretion of radionuclides in the body (97I2)

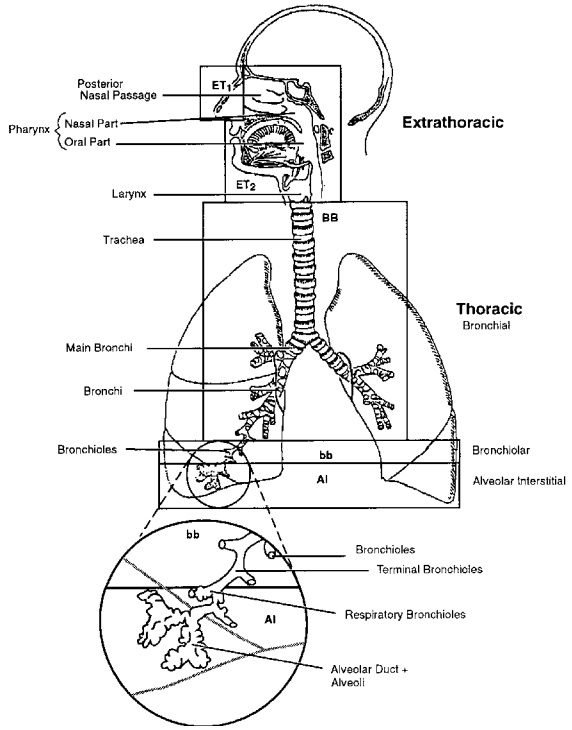



Fig. 3.2 Respiratory tract regions defined in the Human Respiratory Tract Model (94I2)

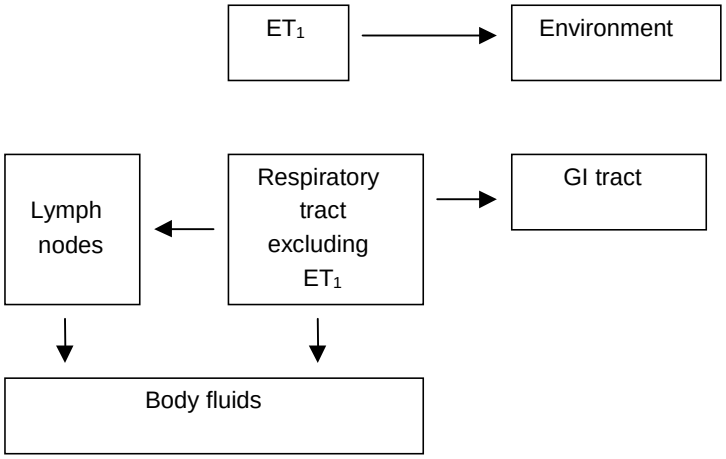



Fig. 3.3 Routes of clearance from the respiratory tract

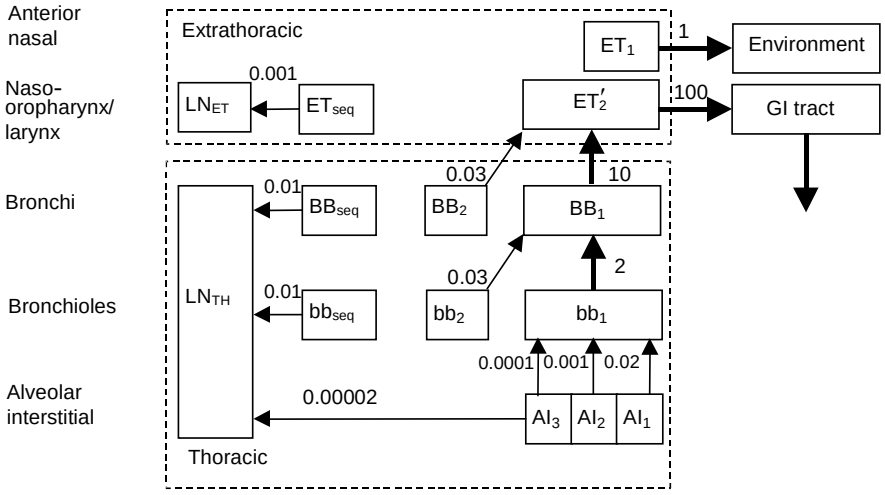



Fig. 3.4 Compartment model representing time-dependent particle transport from each respiratory tract region. Rates shown alongside arrows are reference values in units of $d^{-1}$. It is assumed that (i) the AI deposit is divided between $AI_1$, $AI_2$ and $AI_3$ in the ratio 0.3:0.6:0.1; (ii) the fraction of the deposit in BB and bb that is cleared slowly ($BB_2$ and $bb_2$) is 50% for particles of physical size <2.5 µm and decreases with diameter >2.5 µm, and the fraction retained in the airway wall ($BB_{seq}$ and $bb_{seq}$) is 0.7% at all sizes; (iii) 0.05% of material deposited in region $ET_2$ is retained in its wall ($ET_{seq}$) and the rest in compartment $ET_2'$ which clears rapidly to the GI tract. The model as shown above would describe the retention and clearance of a completely insoluble material. However, there is in general simultaneous absorption to body fluids of material from all the compartments except $ET_1$ (94I1)

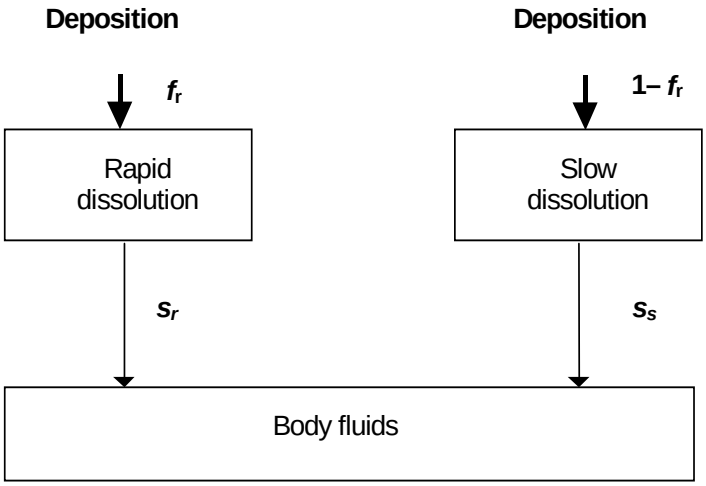



Fig. 3.5. Compartment model representing time-dependent dissolution, followed by instantaneous uptake to body fluids. A fraction $f_r$ of the deposit is initially assigned to the compartment labelled "Rapid dissolution", and the rest ($1 - f_r$) of the deposit is initially assigned to the compartment labelled "Slow dissolution".

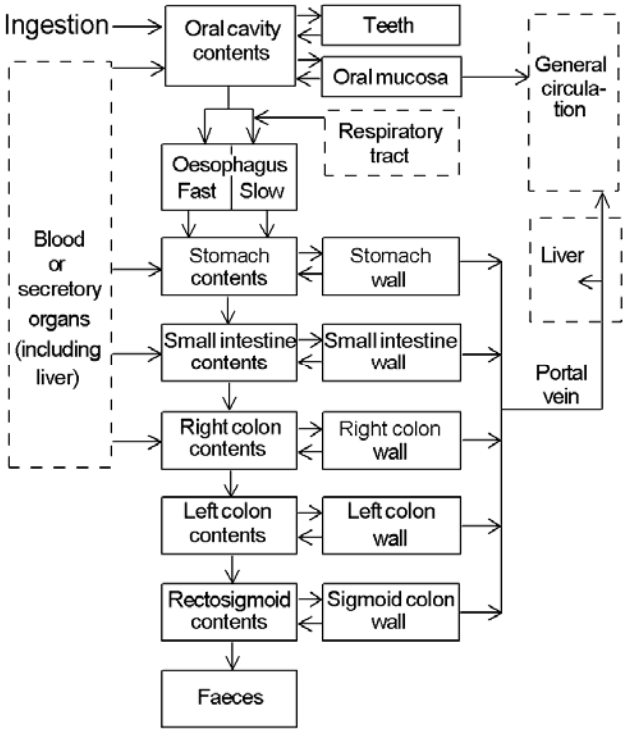



Fig 3.6. Structure of the HAT model. The dashed boxes are included to show connections between the HATM and the HRTM and systemic biokinetic models. $f_A$ gives net transfer to blood and replaces the $f_1$ value of the gastrointestinal tract model. In general, uptake of radionuclides is assumed to occur from the small intestine (Section 3.3).

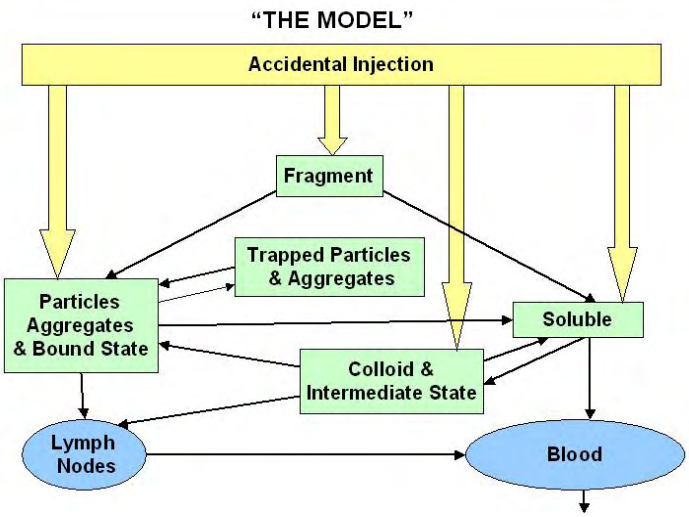



Figure 3.7. Diagram illustrating the NCRP Model for Wounds

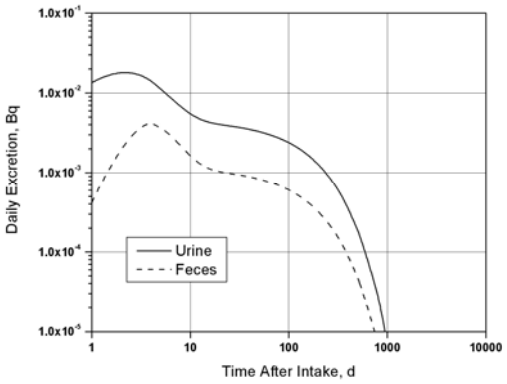



Figure 3.8 $^{137}$Cs Wound, Weak Category; predicted values (Bq per Bq intake) following acute intake.

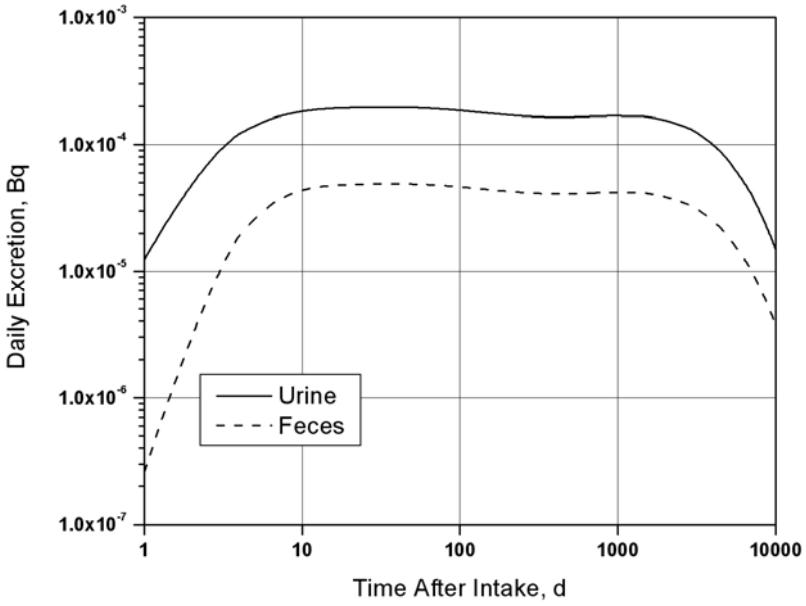



Figure 3.9 $^{137}$Cs Wound, Particle Category; predicted values (Bq per Bq intake) following acute intake.

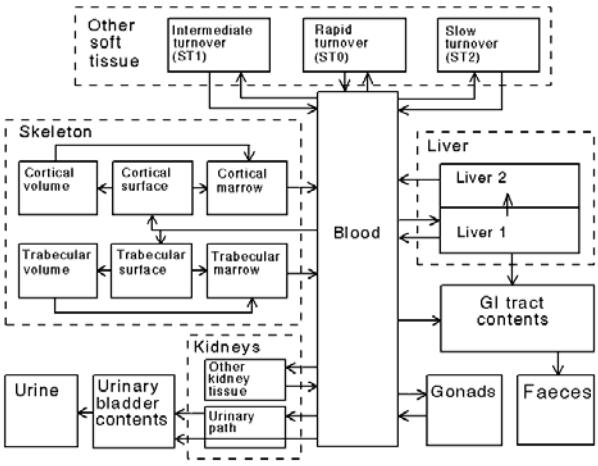
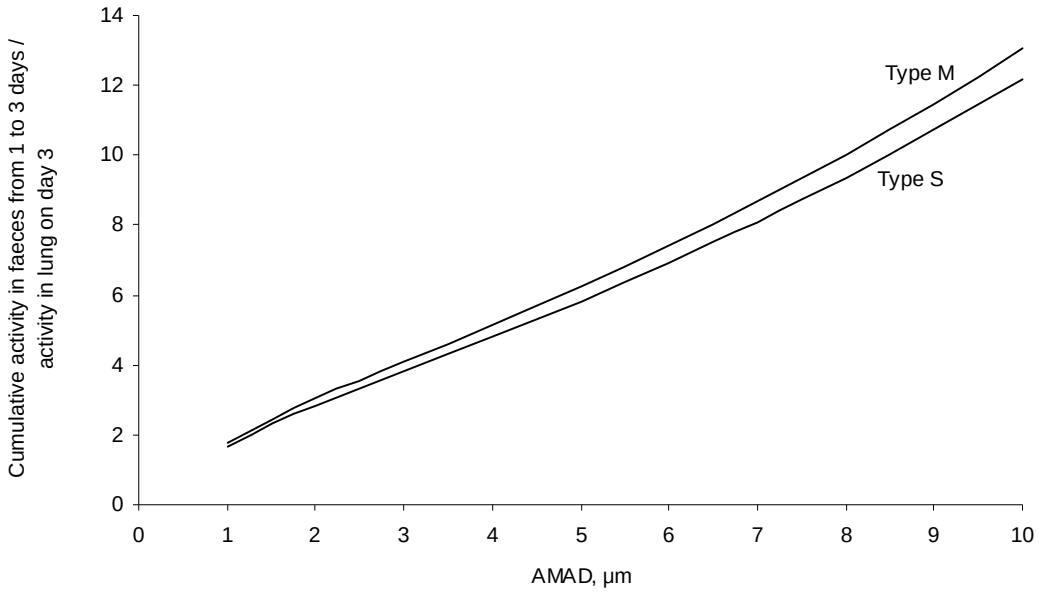



Fig. 3.10 Diagram of the biokinetic model for plutonium and americium.

Fig. 7.1 Variation with fraction inhaled of the ratio of $^{241}$Am lung activity at 3 days after




inhalation, to cumulative activity in faeces from 1 to 3 days predicted by the HRTM for a Reference Worker. (Will need to be revised using HATM etc.)

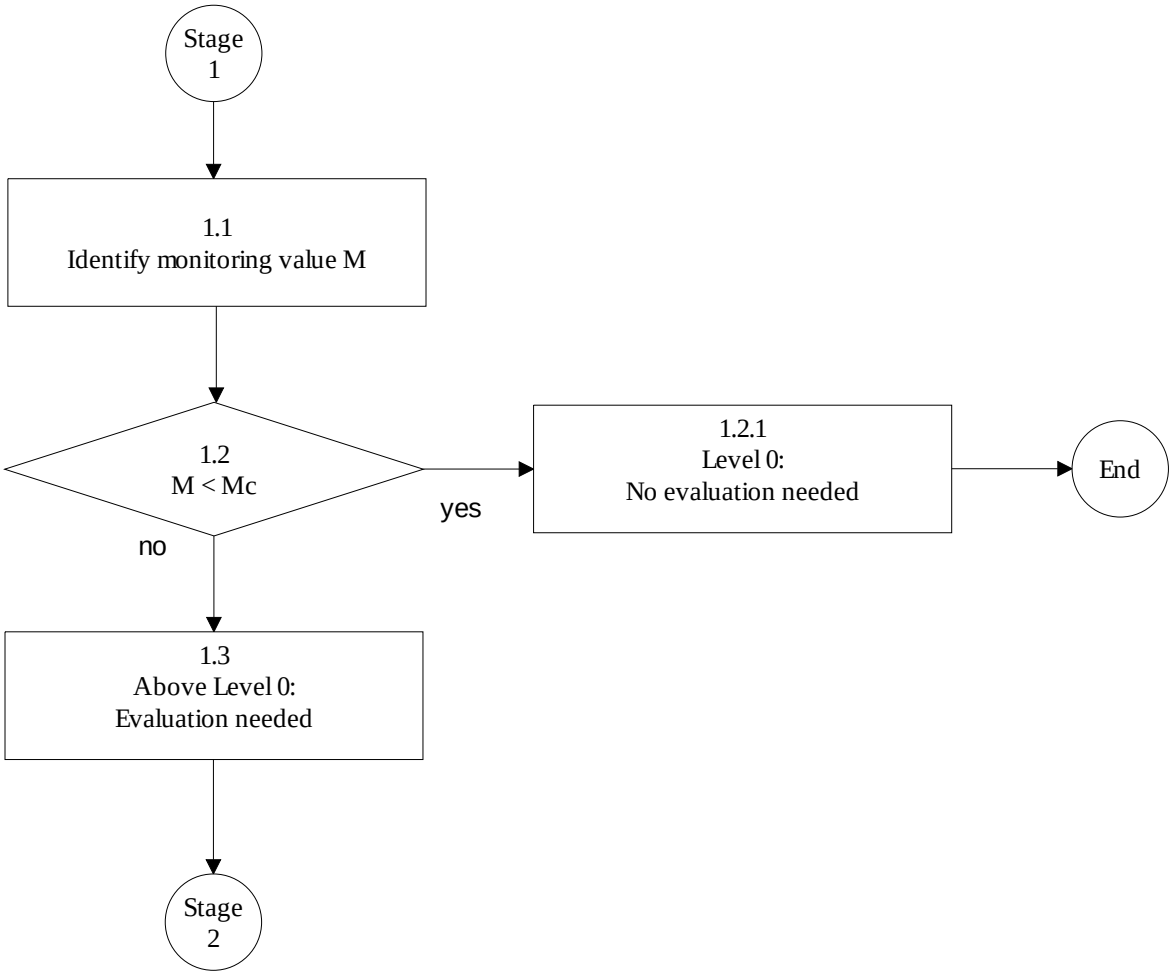



Fig. 8.1 Stage 1 – Structured approach to dose assessment. Level 0 refers to cases where it is expected that the committed effective dose from intakes of radionuclides that occur in the accounting year is likely to be below 0.1 mSv.

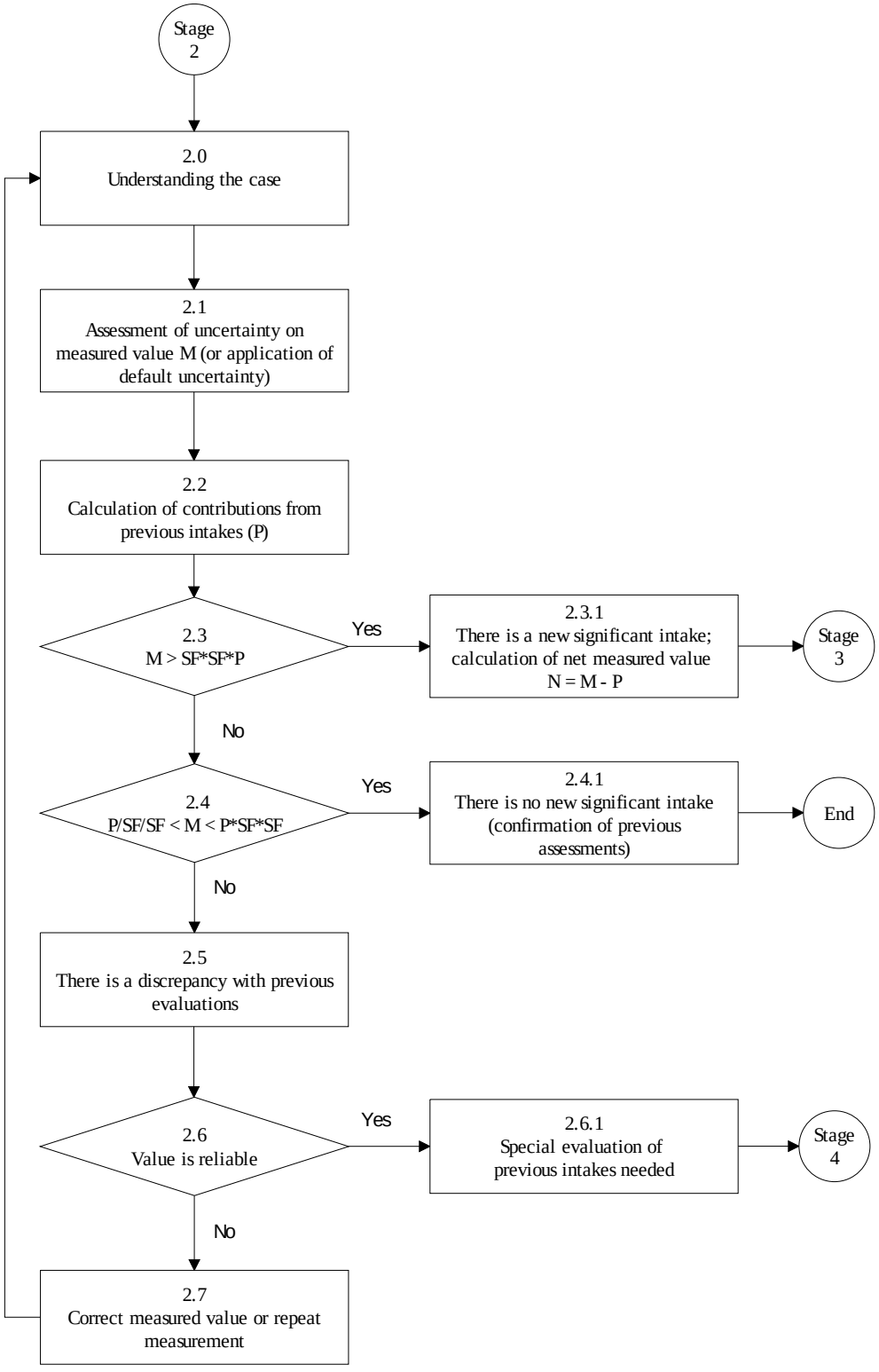



Fig. 8.2 Stage 2 – Structured approach to dose assessment. Check on significance of new measurement and consistency with previous evaluations. (M – measured value, SF – scatting factor, P- contributions from previous intakes – see text)

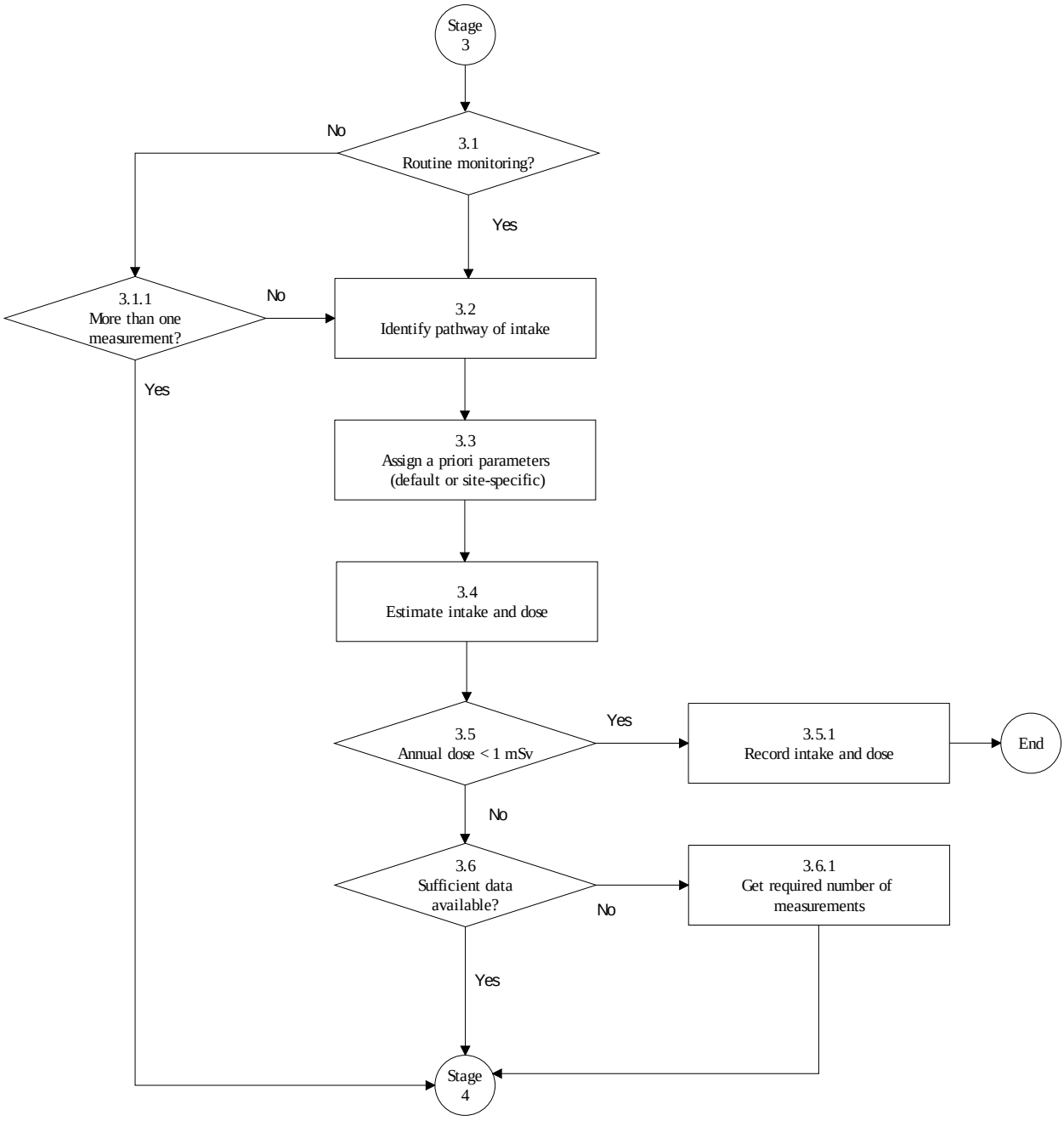



Fig. 8.3 Stage 3 – Structured approach to dose assessment. Having determined the measured value (M) to be due to a new intake, the intake and dose are evaluated from the net value (N) using a priori parameters. This standard evaluation procedure should be applied only for routine monitoring.

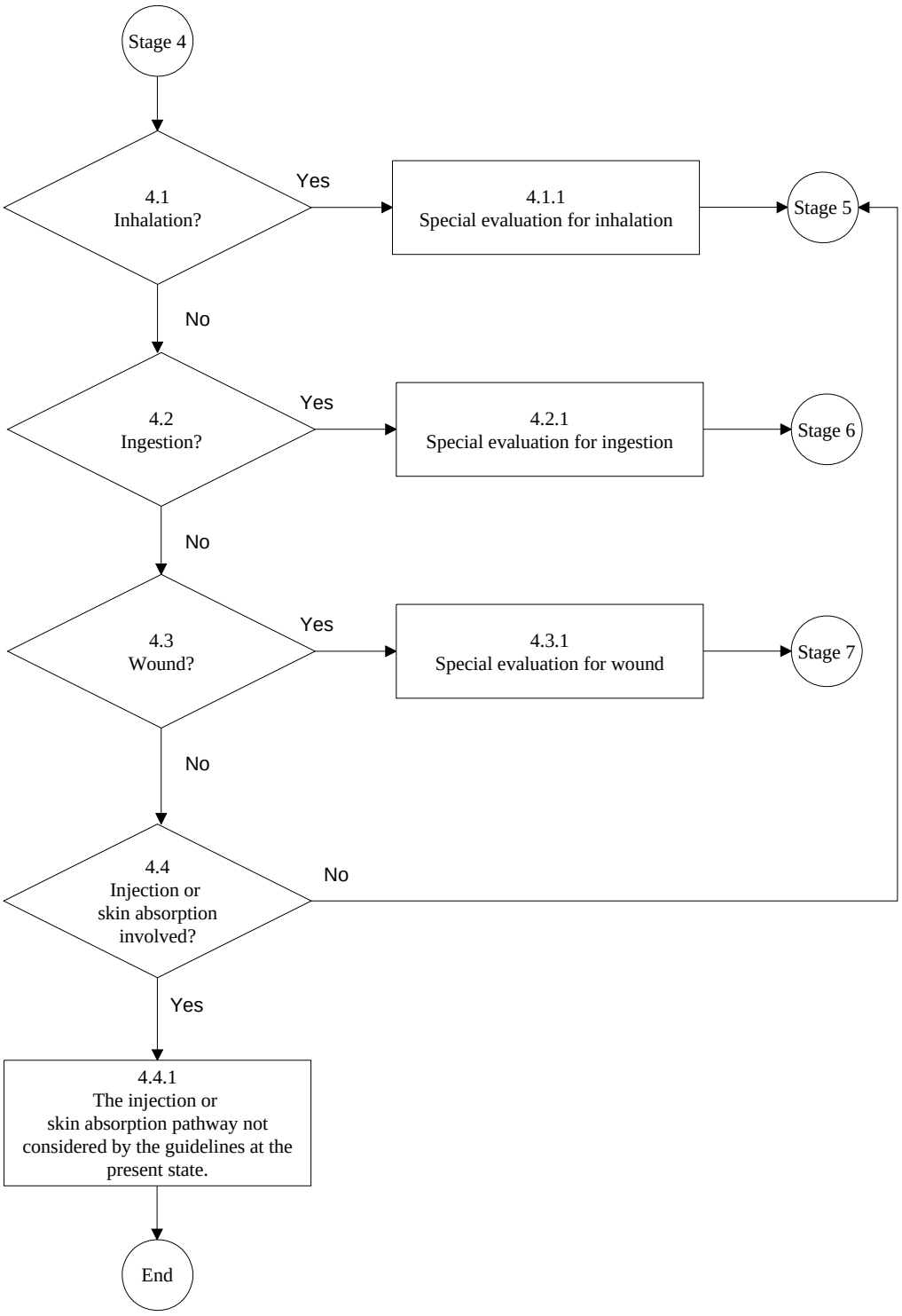



Fig. 8.4 Stage 4 – Structured approach to dose assessment. Special procedures are needed for the evaluation when there is evidence for an internal committed effective dose of more than 1 mSv or in all cases of special monitoring. In all these cases the evaluation procedures depend to some extent on the pathway of intake and thus, in Stage 4 the pathway of intake has to be identified.

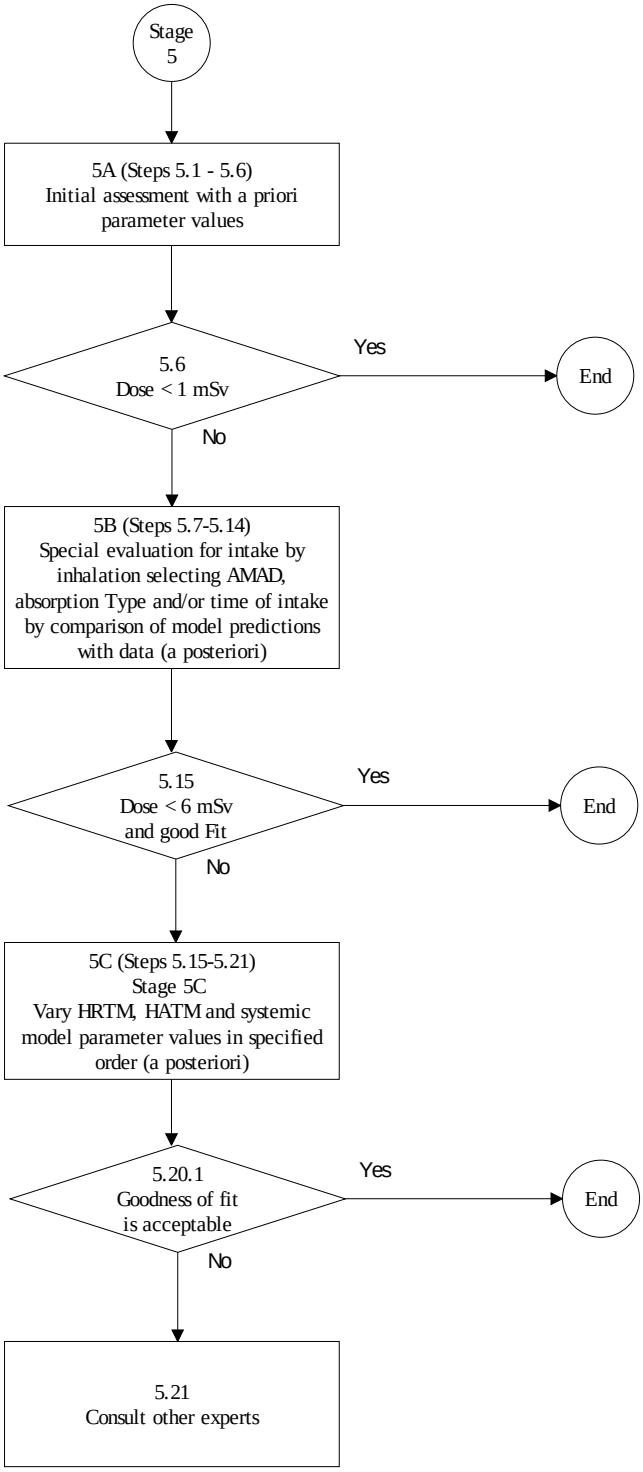



Fig. 8.5 Stage 5 – Structured approach to dose assessment. The special procedure for inhalation cases above Level 1 is grouped in three subsequent stages. In the first stage (5A), a simple evaluation is carried out using parameter values chosen a priori before the evaluation is carried out. The procedure is very similar to the “Standard procedure” (Stage 3). The main difference is that in a special procedure there should be more than one measurement. Details of Stages 5A to 5C are given in Annex G.

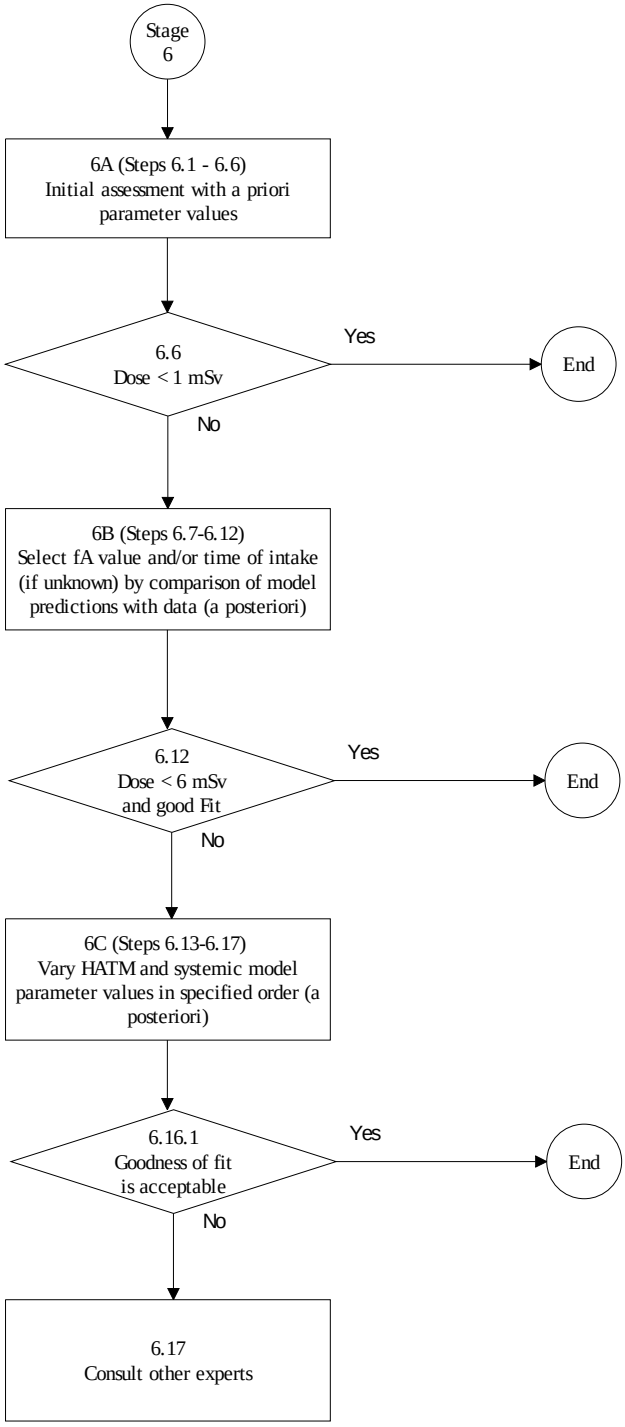



Fig. 8.6 Stage 6 – Structured approach to dose assessment. The special procedure is analogous to that for inhalation (Figure 8.5). It is grouped into three subsequent stages. In the first stage (6A), a simple evaluation is carried out using parameter values chosen a priori: before the evaluation is carried out. The procedure is very similar to the "Standard procedure" (Stage 3). The main difference is that in a special procedure there should be more than one measurement. Details of Stages 5A to 5C are given in Annex G.

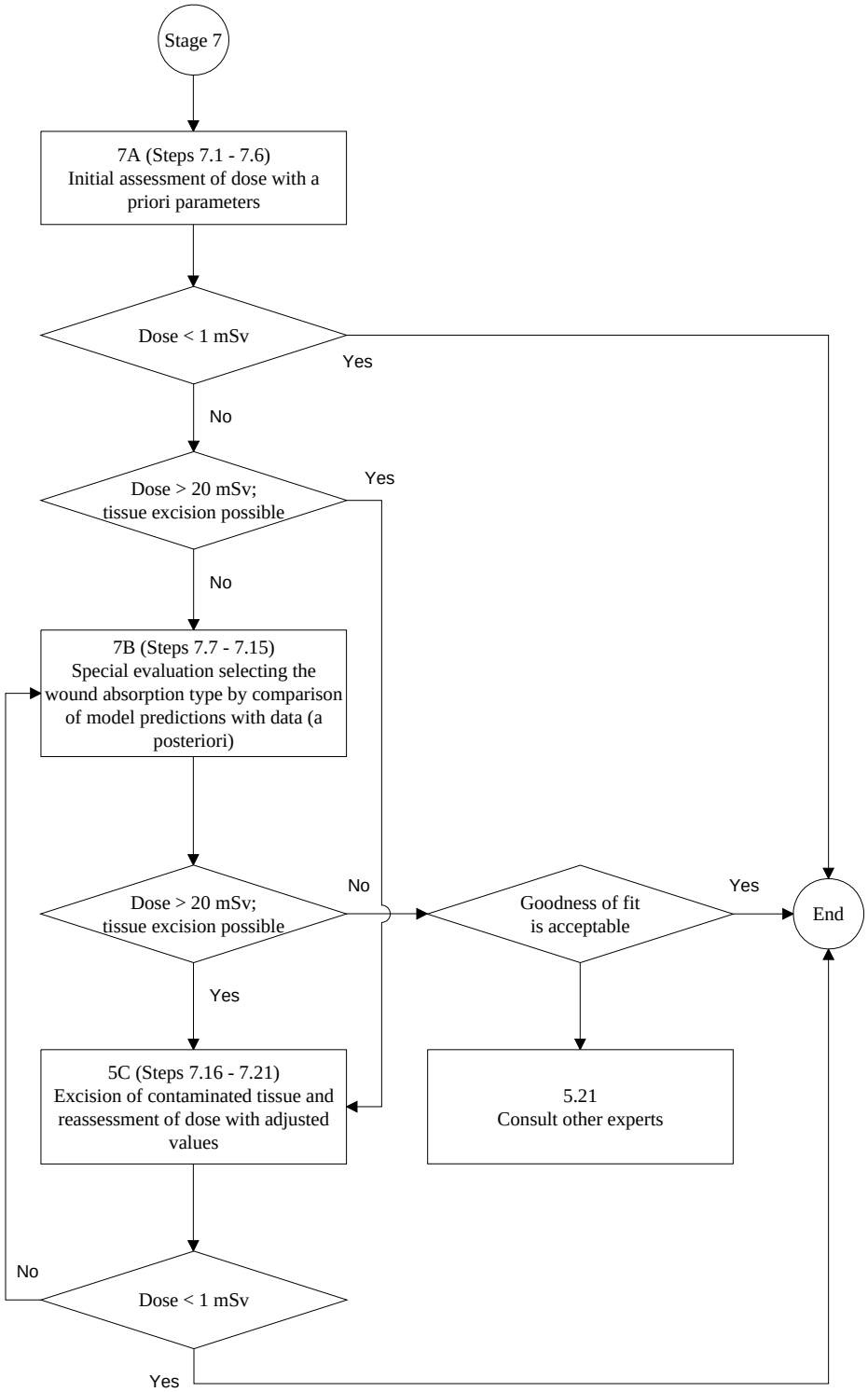



Fig. 8.7 Stage 7 – Structured approach to dose assessment for Steps 7A, 7B, 7C (see Annex G). The special procedure for wound deposition is grouped in three subsequent stages. In the first stage (7A), a simple evaluation is carried out using parameter values chosen a priori: before the evaluation is carried out.

# Annex A

## Proposed Content of Accompanying CD-ROM

**A Phipps, K F Eckerman**

The purpose of the CD is to provide the user with access to numerical data of aid in the interpretation of bioassay measurements and the subsequent calculation of dose.

A single CD would contain dose, excretion, and retention coefficients for the male and female workers for use in interpretation of bioassay measurements. The CD will address the information needs of two types of users:

- The casual user - reviewing the data supporting the published effective dose coefficients
- The expert user who wishes to access the coefficients from within his own software.

The casual user will be served by providing software which accesses the data files prepared to address the needs of the expert user.

The CD will contain only tabulated retention and excretion coefficients (q) (traditional reffered to as functions), ie retained (or excreted) activity per unit intake. The software will combine these data with dose coefficients to provide numerical values for the dose per unit content (z), and possibly the retained activity for an intake leading to a effective dose of 1 mSv ($Q_1$). This is a very simple operation for the computer code to perform.

The software will display q, z and $Q_1$ graphically, since often the user only wants to 'get a feel for' the data. This will reduce the number of figures required in the Publication itself.

The presentation of the various coefficients as numerical tables not only facilitates the development of the CD package but also protects the expert (and the package in his hands) from changes in computer operating systems. This reduces ICRP's burden in supporting the package. Note ICRP could release the source code with the CD or at a later time if changes in operating systems resulted in the software being inoperable.

The software could also include algorithms for interpolation and integration of the coefficients, eg. q. This will enable the convolution of the tabulated unit response functions on the CD (e.g., fractional excretion rate for a unit intake) with a user-postulated intake function. QA considerations suggest that well established algorithms (and their reduction in code) should be selected for these functions. Efforts will need to be devoted to the types of exposure incidents that users might encounter. As a minimum the software should be able to address acute intakes (single and repeating), chronic intake rates (continuous, periodic, and user specified).

It is intended to present the coefficients for inhalation intakes in a manner that enables the user to provide aerosol size or breathing patterns for the intake. It may also be possible to allow the user to change other aerosol parameters such as GSD, shape factor, density. This will be accomplished by presenting the coefficients as a linear combination of the coefficients for deposition in 9 regions of the respiratory tract. (This approach was taken in Publication 30, however that model only required a triplet.)

It had been suggested that the software provided on the CD should perform fits of intake to measurement data. This would be a departure from the current passive software, which simply displays numerical values, and would be substantially more complicated

than the software outlined above. To address real world needs in bioassay interpretation it would be necessary to perform numerical “fits” of expected to observed, based on coefficients on the CD and user supplied characterization of the exposure incident. The DOCAL Task Group now feels that the coding and support effort required to reliably offer this type of software is beyond the role of ICRP. Many laboratories have developed their own codes of this some of which are commercially available.

# Annex B

## Examples of the Interpretation of Monitoring Data

## 1 Introduction

It is intended that this Annex of the Guidance Document will include several examples, which will demonstrate how the guidance can be applied in practice to the assessment of committed doses in a variety of situations. They will illustrate various aspects of the structured approach described in the main text and for more complex assessments in Annex G. One example is included in this consultation document. Further examples will be included in due course.

The two examples given here were used in the recent joint IAEA-IDEAS Intercomparison Exercise on Internal Dose Assessment (Doerfel et al 2005b).

The first case concerns inhalation of cobalt-60 and is an artificial case, and so the true intakes are known. The second case concerns exposure to tritium.

The description of the information available to the assessor follows the structure adopted in recent European Intercomparison Exercises:

- Case description
  - The event
  - Description of the working area
  - Characteristics of work
  - Reasons for monitoring; initiating event
  - Actions taken
- Additional information, e.g:
  - Air monitoring
  - Chemical form
  - Physical characteristics, particle size
  - Any intervention used (blocking, chelating, etc.)
- Body monitoring data
- Excretion monitoring data
- Personal data

The assessment of the case then follows the Guidelines:

- Plot of the data and simple calculation to obtain an overview of the case and a rough estimate of intake and dose.
- Analysis as for a special procedure assuming inhalation (Stage 5).

# 1. ACUTE INHALATION OF COBALT 60 IN A SOURCE PREPARATION FACILITY

## 1.1 Case description

### 1.1.1 The event

#### 1.1.1.1 Description of the working area

Preparation facility for $^{60}$Co sources.

#### 1.1.1.2 Characteristics of work

Cobalt wires irradiated by neutrons in a nuclear reactor facility were used for the preparation of sealed $^{60}$Co sources.

#### 1.1.1.3 Reasons for monitoring; initiating event

An irradiated capsule containing 900 TBq of $^{60}$Co wire was opened in a hot cell and after 10 minutes dose rate alarms sounded.

#### 1.1.1.4 Actions taken

Operators closed the source, put on protective clothing and respirators, stopped the leakage and decontaminated the workplace. A program of *in vivo* monitoring was carried out ten days after the event and continued up to 3 years. Urine samples were also taken.

### 1.1.2 Additional information

#### 1.1.2.1 Air monitoring

Not available

#### 1.1.2.2 Chemical form

Cobalt metal and/or oxide (temperature during irradiation was around 300°- 400°C).

#### 1.1.2.3 Physical characteristics, particle size

Aerosol

#### 1.1.2.4 Nose swab, bronchial slime or similar

None

#### 1.1.2.5 Non removable skin contamination

None

#### 1.1.2.6 Wound site activity

N.A.

#### 1.1.2.7 Any intervention used (blocking, chelating, etc.)

None

### 1.1.3 Body monitoring data

#### 1.1.3.1 Organ activity measurement

None

#### 1.1.3.2 Whole body activity measurement

| Time of measurement after intake (d) | Whole body activity of $^{60}$Co (Bq) |
|---|---|
| 10 | 2.39E+04 |
| 14 | 2.92E+04 |
| 17 | 2.01E+04 |
| 20 | 1.82E+04 |
| 27 | 2.16E+04 |
| 40 | 1.98E+04 |
| 60 | 2.16E+04 |
| 80 | 1.75E+04 |
| 190 | 1.16E+04 |
| 370 | 8.1E+03 |
| 747 | 4.8E+03 |
| 1010 | 2.7E+03 |

### 1.1.4 Excretion monitoring data

#### 1.1.4.1 Urine activity measurement

| Time of measurement after intake (d) | Daily urinary excretion rate of $^{60}$Co (Bq/d) |
|---|---|
| 14 | 7.09E+02 |
| 27 | 6.4E+01 |
| 40 | 7.1E+01 |
| 60 | 3.7E+01 |
| 80 | 2.9E+01 |
| 190 | 1.1E+01 |
| 370 | 1.7E+00 |

#### 1.1.4.2 Faeces activity measurement

None

### 1.1.5 Personal Data

#### 1.1.5.1 Sex

Male

#### 1.1.5.2 Age

35 years

#### 1.1.5.3 Weight

70 kg

### 1.1.6 Other comments relevant for intake and dose estimation

It is recommended in the Guidelines to assume that the *in vivo* measurements can be approximated by a log-normal distribution. The scattering factor (SF) due to counting errors (i.e. Type A errors) was assumed to be 1.07 whereas the SF due to other errors (i.e. Type B errors) was assumed to be 1.18. The SF is the geometric standard deviation of the log-normal distribution.

It is recommended to assume that the urine measurements can be approximated by a log-normal distribution with a total SF, due to Type A and Type B errors, of 1.8.

**Estimate the intake and the committed effective dose E(50)**

## 1.2 Assessment of case

Before following the guidelines to assess the case it is useful to plot the available data (Figure 1.2.1) and perform a simple calculation to assess the intake and dose.

From the case description (Section 1.1), the time of intake is known and the intake pathway can be considered as inhalation. Furthermore, the data appears to be consistent with an acute inhalation of $^{60}$Co (Figure 1.2.1).

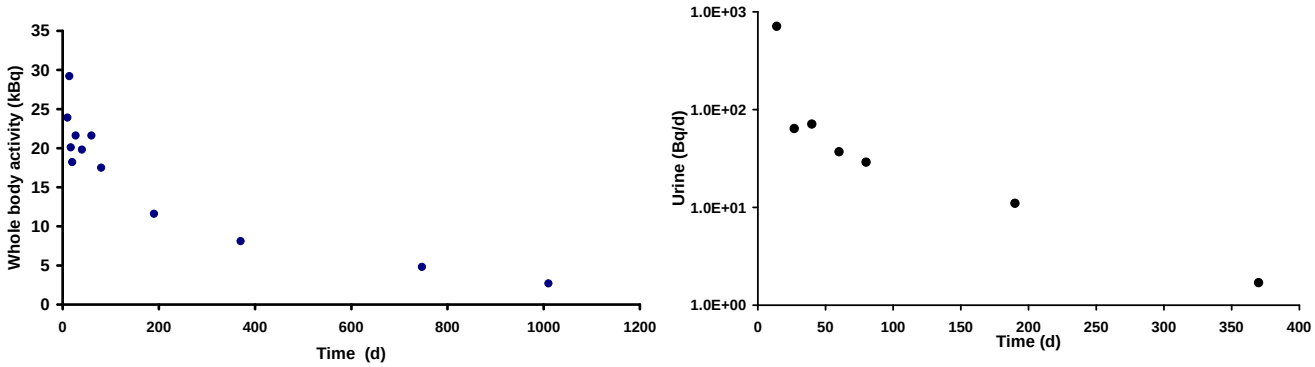



Figure 1.2.1 A plot of the measurement data. Twelve whole body data and 7 urine data are given.

The first whole body measurement at 10 days is 2.39 $10^4$ Bq. As the chemical form is given as metallic cobalt or cobalt oxide it is reasonable to assume absorption Type S for a simple calculation of intake and dose. ICRP Publication 78[(i)] gives a whole body activity content of 0.065 Bq for a worker after 10 days following an acute inhalation of 1 Bq of $^{60}$Co assuming Type S and a 5 μm AMAD aerosol. Therefore, on this basis alone, the intake is 2.39 $10^4$/ 6.5 $10^{-2}$ = 3.7 $10^5$ Bq (i.e. about 370 kBq). The corresponding dose coefficient given in ICRP Publication 68[(ii)] (for inhalation of $^{60}$Co by a worker assuming Type S and a 5-μm AMAD aerosol) is 1.7 $10^{-8}$ Sv/Bq. This gives E(50) = 3.7 $10^5$ Bq × 1.7 $10^{-8}$ Sv $Bq^{-1}$ = 6.3 $10^{-3}$ Sv (i.e. about 6 mSv). So by carrying out a simple calculation the estimated intake is about 370 kBq and the resulting E(50) is about 6 mSv. This gives us a rough estimate of the intake and dose.

The following sections describe the assessment of the case by following the Guidance. As this is a special monitoring case for inhalation the steps in Stage 5 are followed.

The models that will be used to assess the intake and dose include the ICRP Publication 66 Human Respiratory Tract model[iii], the ICRP Publication 30 Gastrointestinal Tract model[iv] and the ICRP Publication 67 systemic biokinetic model for cobalt[v].

### 1.2.1 Step 5.1: Identification of data and assignment of realistic uncertainties

Twelve whole body measurement data and 7 urine data points are available (Figure 1.2.1).

The case description recommended the assessor to assume the whole body measurements are log-normally distributed. Scattering factors (SF) values for Type A errors (i.e. counting errors) of 1.07 and for Type B errors (i.e. other errors such as calibration errors) of 1.18 were given for the whole body measurements. By using the following formula, which is given in the guidelines, an overall SF of 1.2 is calculated for the whole body measurements.

$$\text{Overall SF} = \exp\left[\sqrt{\sum_i \ln^2(\text{SF}_i)}\right]$$

where $SF_i$ is the scattering factor for each component i (i.e. Type A and Type B errors).

Thus, for the whole body measurements a log-normal distribution is assumed with a SF of 1.2. The SF is the geometric standard deviation of the log-normal distribution.

The case description recommended the assessor to assume the urine data are log-normally distributed with an overall SF value of 1.8. Thus, for the urine data a log-normal distribution is assumed with a SF of 1.8.

At this stage there is no reason to reject any of the data so all of the data will be used to assess the intake.

### 1.2.2 Step 5.2: Assessment of contributions from previous intakes.

In this case, no information is given about previous intakes so assume that the measured activities all arise from this incident.

### 1.2.3 Step 5.3: Assign a priori parameters (default or site-specific)

In the case description the chemical form of the material was given as metallic cobalt or cobalt oxide. The ICRP default absorption Type for cobalt oxide is Type S[ii]. The default parameter values assumed are:

5 μm AMAD aerosol
Absorption Type S
$f_1$ value 0.05
Reference worker

### 1.2.4 Step 5.4: Is the time of intake known?

The time of intake is known so proceed to step 5.5.

### 1.2.5 Step 5.5: Calculate dose with a priori parameters

To estimate the intake it is necessary to calculate the predicted values, $f(t_i)$ of each of the measured quantities assuming unit intake. The best estimate of intake (I) is determined so that the product $I\ f(t_i)$ best fits the measurement data ($M_i$, $t_i$) . The fitting method recommended by the guidelines is the maximum likelihood method.

The equations given in the section entitled 'Best estimate of intake' of the guidelines can be applied to cases where multiple types of measurement data are available. The equations given in the guidelines are analytical solutions to the maximum likelihood method where the measurement data are log-normally distributed with a given SF. It should be noted that the equations do not apply to data that are reported as being below the limit of detection.

The equation giving the best estimate of intake, I is given by:

$$\ln(I) = \frac{\sum_{i=1}^{N}\left(\frac{\ln(I_i)}{(\ln SF_i)^2}\right)}{\sum_{i=1}^{N}\frac{1}{(\ln SF_i)^2}} \qquad (1.2.1)$$

where

$SF_i$ is the scattering factor for $M_i$
$I_i$ is the estimated intake from each measurement value $M_i$ and is given by

$$I_i = \frac{M_i}{f(t_i)}$$

For this case, 19 intake estimates are determined (12 from the 12 whole body measurements and 7 from the 7 urine measurements).

Substituting the SF of 1.2 for the whole body data and 1.8 for the urine data into equation 1.2.1 gives:

$$\ln(I) = \frac{\sum_{i=1}^{12}\left(\frac{\ln(I_i)}{(\ln 1.2)^2}\right) + \sum_{j=1}^{7}\frac{\ln(I_j)}{(\ln 1.8)^2}}{\sum_{i=1}^{12}\frac{1}{(\ln 1.2)^2} + \sum_{j=1}^{7}\frac{1}{(\ln 1.8)^2}}$$

where $I_i$ refers to the intake estimates from the whole body data and $I_j$ refers to the intake estimates from the urine data.

Alternatively, the best estimate of intake can be determined using appropriate internal dosimetry software. The IMBA Professional software was used here to assess this case. It implements the current ICRP dosimetric and biokinetic models but enables the user to alter parameter values from the ICRP defaults. It uses the maximum likelihood method to fit multiple data and can assess the intake by fitting predicted values to different types of data simultaneously. The intake was estimated by fitting the predicted values to both the whole body data and the urine data simultaneously. This is identical to calculating the intake using the above equations.

With the default parameter values given in step 5.3 the estimated intake is 389 kBq and E(50) is 6.4 mSv. The fits to the data are shown in Figure 1.2.2. However, the fit to the urine data is poor (Figure 1.2.2), and this indicates that some important model parameter values are incorrect.

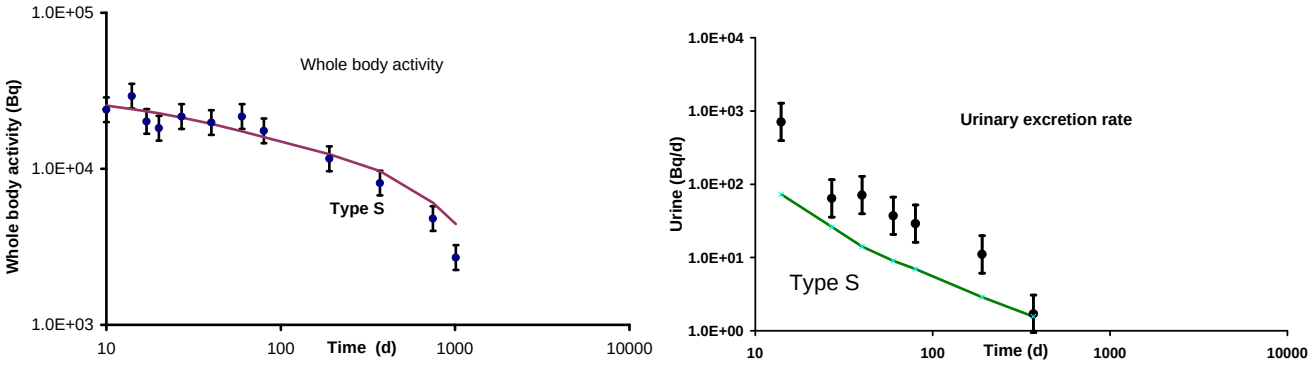



Figure 1.2.2. Model fits to whole body and urine data assuming Type S (step 5.5)

### 1.2.6 Step 5.6: Is E(50) < 1 mSv?

With the default parameter values E(50) was calculated to be 6.4 mSv. As this is greater than 1 mSv proceed to the next step

### 1.2.7 Step 5.7: Are there sufficient relevant data?

The Guidance suggests a minimum number of data that are required for a dose assessment for certain radionuclides. The minimum number suggested depends on the dose level. For $^{60}$Co the minimum number is 5 whole body measurements over a time period of 30 days if the dose level is greater than 6 mSv. In this case, there are 12 whole body measurements and 7 urine measurements. Therefore there are enough data for this dose assessment, so proceed to the next step. However, it should be pointed out that suitable early data that can be used to estimate an effective AMAD are not available.

### 1.2.8 Step 5.8: Is the time of intake known?

The time of intake is known so proceed to step 5.9.

### 1.2.9 Step 5.9: Are early and lung faeces available?

There are no early lung and faecal data available so proceed to step 5.11

### 1.2.10 Step 5.11: Assessment of dose by fitting absorption Type

In this step intakes and doses are assessed using the default absorption Types for cobalt given in ICRP Publication 68[(ii)]. ICRP Publication 68 suggests Type S for cobalt oxide and Type M for unspecified compounds of cobalt.

#### 1.2.10.1 Type S

Assuming Type S the fit to the urine data is poor (step 5.5, Figure 1.2.2). The estimated intake is 389 kBq and E(50) is 6.4 mSv.

#### 1.2.10.2 Type M

Assuming Type M with $f_1$= 0.1 and 5 μm AMAD, the estimated intake is 481 kBq and E(50) is 3.4 mSv. The fit to the data is poor (Figure 1.2.3).

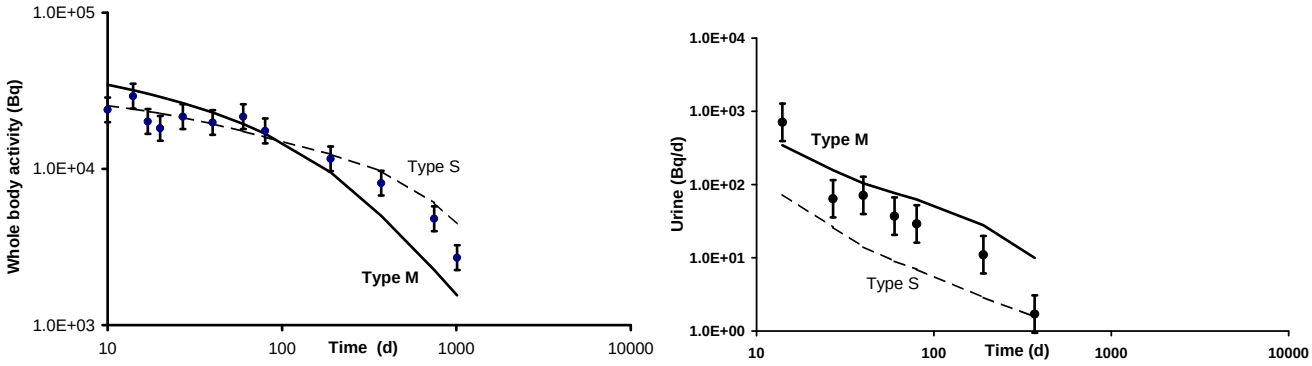



Figure 1.2.3 Model fits to whole body and urine data assuming default absorption Types (step 5.11). **[** — Type M; – – – Type S**]**

### 1.2.11 Step 5.8: Is the goodness of fit acceptable?

The guidelines suggest rejecting the fits if

- the chi squared test ($\chi^2$ ) fails (i.e. if p-value < 0.05). In other words if the fit is inadequate at the 5% level of significance, or if
- the fit displayed graphically looks unreasonable by eye.

It is acknowledged that whether or not the fit displayed graphically looks unreasonable by eye is a subjective judgement. However, generally, a fit would be considered unreasonable if all, or a long series, of data were systematically underestimated or overestimated.

As the measurements are log-normally distributed, the $\chi_O^2$ is calculated using the following formula for N measurements

$$\chi^2 = \sum_{i=1}^{N} \left( \frac{\ln(M_i) - \ln[I\, f(t_i)]}{\ln(SF_i)} \right)^2$$

When fitting predicted values to different types of data simultaneously, the overall $\chi_O^2$ is the sum of the calculated $\chi_O^2$ values for each data set. The number of degrees of freedom is the total number of measurements minus one. In this case it is 18 (i.e. 12 whole body data + 7 urine data – 1 = 18). The expected value of $\chi^2$ is equal to the number of degrees of freedom.

For the calculated $\chi_O^2$ value with N-1 degrees of freedom, the corresponding p-value can be obtained from Statistical Tables. Alternatively, the p-value can be obtained from Microsoft Excel using the function CHIDIST($\chi_O^2$, N-1). The p-value is the fraction of the actual $\chi^2$ distribution that lies above the calculated $\chi_O^2$ value. So if p is very small, the

calculated $\chi_{O,}^2$ value is very much larger than expected and therefore it can be concluded that the fit is inconsistent with the data.

Assuming Type S the overall $\chi_{O,}^2$ is 57 with 18 degrees of freedom and the corresponding p value is $6.2\ 10^{-6}$. As the p-value is < 0.05, the fit is rejected.

Assuming Type M the overall $\chi_{O,}^2$ is 72 with 18 degrees of freedom and the corresponding p value is $2.3\ 10^{-8}$. As the p-value is < 0.05, the fit is rejected.

To summaries, for both Type M and Type S assumptions the p-value is < 0.05. On this basis the fits are rejected and so it is necessary to proceed to step 5.13. It is also worth pointing out that the fits also look unreasonable by eye (Figure 1.2.3).

### 1.2.12 Step 5.13: Assessment of dose by fitting a mixture of default absorption Types

In this step, the intake is estimated by fitting a mixture of absorption Types (M and S) to the whole body and urine data simultaneously. The best fit to the data was obtained for a mixture consisting of 44% Type M and 56% Type S (Figure 1.2.4). The estimated intake is 404 kBq and E(50) is 5.0 mSv.

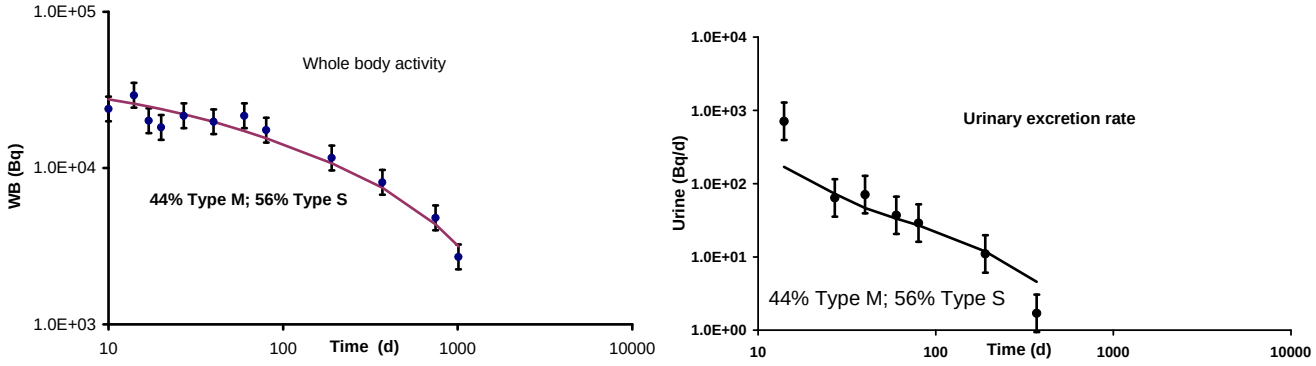



Figure 1.2.4 Model fits to whole body and urine data assuming 44% Type M and 56% Type S (step 5.13).

### 1.2.13: Step 5.15: Is the goodness of fit acceptable?

For a mixture consisting of 44% Type M and 56% Type S, the fits to the data are good (Figure 1.2.4). The overall $\chi_{O,}^2$ is 17 with 18 degrees of freedom and the corresponding p-value is 0.5. As the p-value is > 0.05, the fits are not rejected. This is, therefore, the best estimate of intake and dose. So the intake and dose with the corresponding parameter values are recorded in the next step (i.e. step 5.15.1).

### 1.2.14 Step 5.15.1 : Record dose with all parameter values.

The intake and the dose are recorded with the corresponding parameter values.

- Intake: Acute inhalation of 404 kBq of $^{60}$Co
- Committed effective dose, E(50): 5.0 mSv
- Mixture of Absorption Types M and S

- 44% Type M; 56% Type S
- $f_1$ = 0.10 (Type M); $f_1$ = 0.05 (Type S)
- 5 μm AMAD aerosol
- Reference worker
- ICRP Publication 66 Human Respiratory Tract model
- ICRP Publication 30 Gastrointestinal Tract model
- ICRP Publication 67 systemic biokinetic model for cobalt

### 1.2.15 Summary of assessments

A summary of the assessments of intake and dose is given in Table 1.21, including each calculated $\chi_{O,}^2$ value and the corresponding p-value.

It was not possible to obtain good fits to the both the whole body and urine data with the ICRP default absorption Types. However, good fits were obtained to both data sets by fitting a mixture of absorption Types (44% Type M and 56% Type S). This was carried out in step 5.13.

**Table 1.2.1 Summary of estimated intakes of $^{60}$Co and resulting doses[a, b]**

| Assessment procedure step | Absorption Type | Goodness of fit | | Comment | Intake (kBq) | E(50) (mSv) |
|---|---|---|---|---|---|---|
| | | $\chi_o^{2}$ [c] | p- value [d] | | | |
| Steps 5.5 and 5.11 | Default: Type S | 57 | $6.2\ 10^{-6}$ | Very poor fit to urine data | 389 | 6.4 |
| Step 5.11 | Type M | 72 | $2.3\ 10^{-6}$ | Very poor fit to whole body and urine data | 481 | 3.4 |
| Step 5.11.3 | Mixture of absorption Types (44% Type M and 56% Type S) | 17 | 0.50 | Good fit to both whole body and urine data sets | **404** | **5.0** |

(a) Intake estimates were obtained by fitting the predicted bioassay values to the whole body and urine data simultaneously with IMBA Professional.
(b) The default AMAD of 5 μm was assumed in all assessments.
(c) The expected value of $\chi^2$ is equal to the number of degrees of freedom; (i.e. number of data points – 1 = 18).
(d) The p- value is the probability that $\chi^2$ is greater than $\chi_o^{2}$ for 18 degrees of freedom.

It is interesting to note that if Type S is assumed, then the intake estimated using the whole body data alone is about a factor of 4 lower that that obtained using the urine data alone. Furthermore, if Type M is assumed, then the intake estimated using the whole body data alone is about a factor of 2 greater that that obtained using the urine data alone. This indicates that the material is not Type S or Type M. However, if a mixture of absorption Types (i.e. 44% Type M and 56% Type S) is assumed then the estimates of the intakes using either the whole body data or the urine data are very similar; only 12% different.

It is worth noting that the simple calculation carried out resulted in an intake of 370 kBq and E(50) of about 6 mSv. The final estimate of intake of 404 kBq and the resulting E(50) of 5.0 mSv are similar to those obtained with the simple calculation. This gives the assessor some confidence that an error has not been made while using software to assess the intake and dose.

# 2. ACUTE INTAKE OF HTO

## 2.1 Case description

### 2.1.1 The event

### 2.1.1 Description of the working area

Unknown

### 2.1.2 Characteristics of work

Plant decontamination. The man was refilling a vacuum pump used for cleaning rooms contaminated with tritium.

### 1.2.16 Reasons for monitoring; initiating event

The man removed a filler cap resulting in the expulsion of contaminated air.

### 1.2.17 Actions taken

The man took a shower and changed his clothes. A urine sample was taken. The man was put on a plethoric hydrous diet (8 litres a day) to enhance the excretion of tritium.

### 2.1.2 Additional information

#### 2.1.2.1 Air monitoring

None

#### 2.1.2.2 Chemical form

Tritiated water (HTO).

#### 2.1.2.3 Physical characteristics, particle size

Vapour

#### 2.1.2.4 Nose swab, bronchial slime or similar

None

#### 2.1.2.5 Non removable skin contamination

None

#### 2.1.2.6 Wound site activity

Not applicable

#### 2.1.2.7 Any intervention used (blocking, chelating, etc.)

Plethoric hydrous diet

### 2.1.3 Body monitoring data

#### 2.1.3.1 Organ activity measurement

N.A.

#### 2.1.3.2 Whole body activity measurement

N.A.

### 2.1.4 Excretion monitoring data

#### 2.1.4.1 Urine activity measurement

| Time after intake (d) | Urine activity concentration of $^{3}H$. (MBq/l) |
|---|---|
| 0 | 80.1 |
| 1 | 67.7 |
| 2 | 57.5 |
| 3 | 47.5 |
| 4 | 39.2 |
| 5 | 32 |
| 6 | 27.6 |
| 7 | 24.2 |
| 8 | 22.9 |
| 9 | 19.5 |
| 10 | 16.5 |
| 11 | 14.3 |
| 12 | 12.4 |
| 13 | 11 |
| 14 | 9.62 |
| 15 | 8.23 |
| 16 | 7.81 |
| 18 | 6.36 |
| 20 | 5.25 |
| 22 | 4.26 |
| 24 | 3.52 |
| 26 | 2.86 |
| 28 | 2.80 |
| 30 | 2.08 |
| 33 | 1.54 |
| 35 | 1.25 |
| 36 | 1.02 |
| 38 | 0.97 |
| 39 | 0.78 |
| 41 | 0.64 |
| 44 | 0.56 |
| 47 | 0.42 |
| 49 | 0.36 |
| 50 | 0.31 |

| | |
|---|---|
| 54 | 0.23 |
| 56 | 0.17 |
| 58 | 0.15 |
| 61 | 0.12 |
| 63 | 0.11 |
| 66 | 0.099 |
| 68 | 0.078 |
| 70 | 0.064 |
| 72 | 0.057 |
| 75 | 0.05 |
| 77 | 0.044 |
| 79 | 0.044 |
| 82 | 0.036 |
| 84 | 0.034 |
| 87 | 0.029 |
| 89 | 0.025 |
| 91 | 0.023 |
| 94 | 0.021 |
| 96 | 0.019 |
| 98 | 0.018 |
| 100 | 0.018 |
| 103 | 0.014 |
| 142 | 0.0087 |
| 149 | 0.0081 |
| 156 | 0.0074 |
| 163 | 0.0066 |
| 169 | 0.0064 |
| 177 | 0.0057 |
| 184 | 0.0063 |
| 191 | 0.0043 |
| 196 | 0.0048 |
| 216 | 0.004 |
| 219 | 0.0038 |
| 226 | 0.0041 |
| 233 | 0.0037 |
| 239 | 0.0033 |
| 246 | 0.0028 |
| 254 | 0.0025 |
| 268 | 0.002 |

| | |
|---|---|
| 270 | 0.0022 |
| 274 | 0.0021 |

#### 2.1.4.2 Faeces activity measurement

None

### 2.1.5 Personal Data

#### 2.1.5.1 Sex

Male

#### 2.1.5.2 Age

32 years

#### 2.1.5.3 Weight

71 kg

## 2.2 ASSESSMENT OF CASE

### 2.2.1 Simple hand calculation

Before following the guidelines to assess the case it is useful to plot the available data (Figure 0-1) and perform a simple calculation to assess the intake and dose.

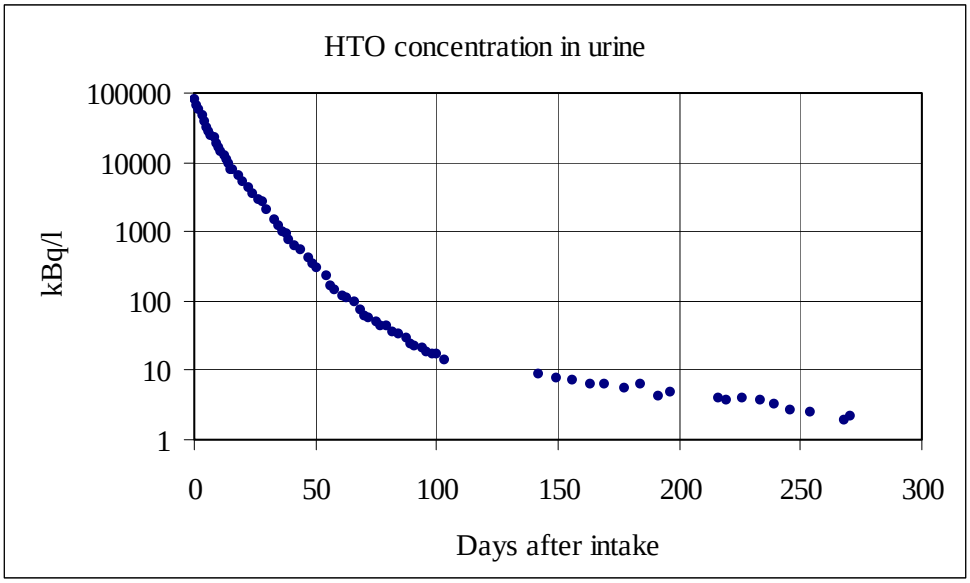



Figure 0-1. Plot of the excretion data for Case 1

The effective dose from intakes of HTO can be assessed using the direct dose assessment method (Figure 0-2). This method involves calculating the area under the urine activity concentration data to determine the number of nuclear transformations.

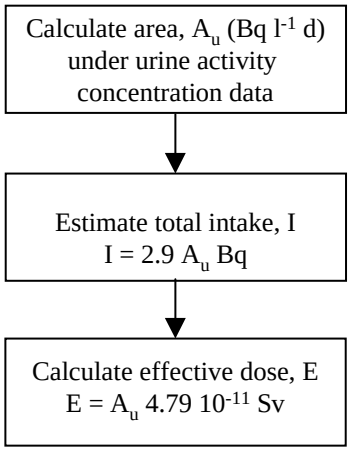



Figure 0-2. Direct dose method for intakes of tritiated water

Assuming that the urine activity concentration data can be approximated by a series of exponentials, the area $A_u$ under the data can be roughly estimated by the following equation:

$$A_U = \frac{e(t=0) \cdot T_{0.135}}{2} \qquad (0\text{-}1)$$

with

| | |
|---|---|
| $A_u$ | approximated area under the urine activity concentration data |
| $e(t=0)$ | activity concentration in the urine at time t=0 |
| $T_{1/ee}$ | time after which the activity concentration is 0.135 e(t=0) |

The initial activity concentration is

$e(t=0) = 8.0\ E07 Bq\ l^{-1}$

and thus

$0.135\ e(t=0) = 1.08\ E07\ Bq\ l^{-1}$

which is reached on day 13. Thus

$T_{0.135} = 13\ d$

and with Equ. (0-2)

$A_u = 5.2\ E08\ Bq\ l^{-1}\ d$

Hence the effective dose is

$E = A_u\ 4.79\ E\text{-}11\ Sv = 0.025\ Sv = 25\ mSv$

### 2.2.2 Assessment according to the guidelines

Following intakes of tritiated water (HTO), most monitoring programmes consist of measuring the activity concentration of $^3H$ in urine samples. The resulting effective dose from intakes of HTO can be assessed using the direct dose assessment method (Fig. 0-3). This method involves calculating the area under the urine activity concentration data to determine the number of nuclear transformations. Thus, the method is not covered by the Structured Approach provided by the IDEAS General Guidelines. There is, however, a special section in the Guidelines describing the direct dose assessment method in detail especially for HTO.

ICRP assumes that HTO is instantaneously translocated to blood following inhalation or ingestion. HTO is assumed to mix rapidly and completely with total body water after its entry to blood. It can be assumed that the activity concentration in urine *(Bq $l^{-1}$)* equals that of total body water. Thus, the activity in total body equals the activity concentration in urine multiplied by the total volume of body water, which is 42 l for reference man (ICRP Publication 23). Finding the area under the activity total body curve then gives the number of nuclear transformations in the total body.

If $A_u$ is the area under the urine activity concentration data *(Bq $l^{-1}$ d)* from the time of the first intake (t=0) to infinity then the total number of nuclear transformations, $U_s$ is given by:

$$U_s = A_u\ 42\ b \qquad (0\text{-}2)$$

where b is a numerical constant converting days to seconds: 86400 s $d^{-1}$.

For intakes of HTO the equivalent dose to each of the target organs is identical and equal to the effective dose, E. Thus, E is obtained by multiplying $U_s$ by the specific effective energy, for the source organ whole body, *SEE(T ← WB).* *SEE(T ← WB)* represents the equivalent dose in a target organ per disintegration in the whole body source organ. For $^3$H $SEE(T \leftarrow WB) = 1.32\ 10^{-17}$ *Sv* per disintegration. This is the value used by ICRP in calculating the dose coefficients for HTO given in ICRP Publication 68[2]. The mass of the source organ whole body is 68.831 kg (Cristy and Eckerman 1993)[3].

The committed (50y) effective dose, *E(50)* is thus approximated by:

$$E(50) = U_s\ 1.32\ 10^{-17}\ Sv \qquad (0\text{-}3)$$

Substituting equation (0-2) into (0-3)gives:

$$E(50) = A_u\ 4.79\ 10^{-11}\ Sv \qquad (0\text{-}4)$$

The total intake, I can be determined by calculating the total amount of activity lost from the body. The ICRP Publication on the revised reference man (ICRP Publication 89, Table 2.30)[4], gives the total water loss per day as 2.9 l $d^{-1}$ for an adult male. Thus, the total activity lost from the body, which gives the total intake is given by:

$$I = 2.9\ A_u\ Bq \qquad (0\text{-}5)$$

The direct dose method does not depend upon a systemic biokinetic model, as $U_s$ is obtained directly by calculating $A_u$ from urine activity concentration data. The accuracy of the method depends on the accuracy of $A_u$. Errors occur in interpolating the data and in extrapolating the data to earlier or later times. If the measurements are frequent then linear interpolation, i.e. using the trapezoidal method, is recommended. Thus, in the present case, the area under the measurement data is approximated by:

$$A_U = \sum_{i=1}^{n-1} \frac{(C_{i+1} + C_i)(t_{i+1} + t_i)}{2}\ Bq\ l^{-1}\ d \qquad (0\text{-}6)$$

where:

- $C_i$ is the activity concentration of HTO (Bq $l^{-1}$) in urine sample i
- $t_i$ is the corresponding measurement time (d) for urine sample i
- n is the total number of urine samples.

In the present case the data cover a time period much greater than the effective half time of HTO in the body (4-18 d)[5] and thus the last data value is relatively low and the error caused by not extrapolating the data to later times is insignificant. This has been demonstrated already by the simple hand calculation (Section 1.2.1).

In the present case Equ. (0-6) results in

$A_u = 5.31\ E08\ Bq\ l^{-1}\ d$

Which is very close to the value derived from the simple hand calculation. Hence the effective dose is

$E = A_u\ 4.79\ E\text{-}11\ Sv = 0.0254\ Sv = 25.4\ mSv$

For an acute intake of HTO as in the present case, improved estimates of $A_u$ can be obtained by fitting a sum of exponential terms, *f(t)*, to the urine activity concentration data *(Bq $l^{-1}$)*. The fitted function *f(t)* is defined as follows:

$$f(t) = \sum_{i=1}^{n} a_i e^{-\lambda_i t} \quad Bq\,l^{-1} \qquad (0\text{-}7)$$

where t is time after the acute intake in days.

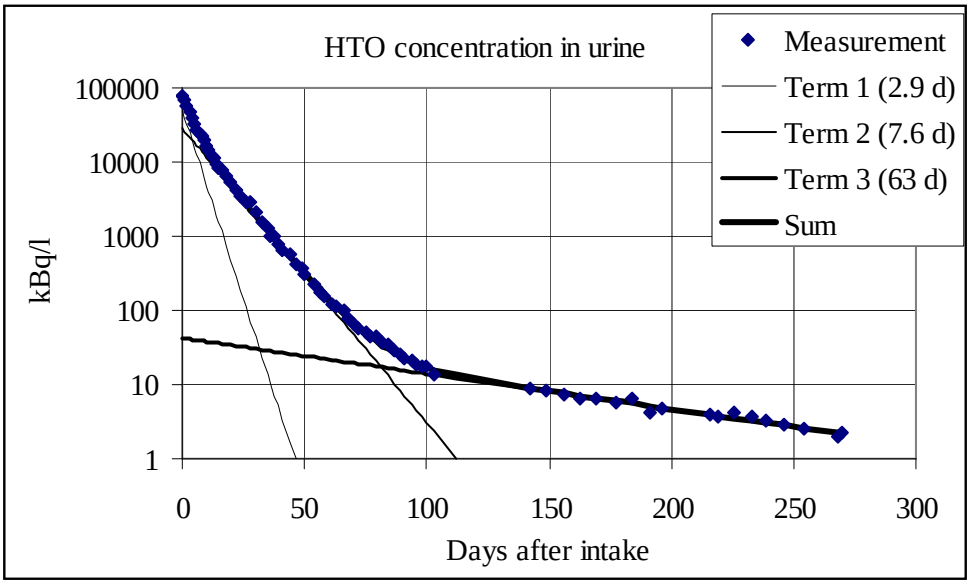



Figure 0-3. Approximation of the measured data by a sum of three exponential terms

Table 0-3. Parameters of the exponential terms for the estimation of the area under the urine activity concentration data

| Term *i* | $a_i$ *in Bq* $l^{-1}$ | $\lambda_i$ *in* $d^{-1}$ | $A_{ui}$ *in Bq* $l^{-1}$ *d* |
|---|---|---|---|
| 1 | 5.10 E07 | 0.235 | 2.17E08 |
| 2 | 2.85E07 | 0.091 | 3.13E08 |
| 3 | 4,2E04 | 0.011 | 4E06 |
| Sum | | | 5.34E08 |

In the present case the measured data can be approximated by a sum of three exponential terms as shown in Fig. 0-3. The respective parameters of theexponential terms are listed in the second and third column of Table 0-4.

The intake is given by:

$$I = 42 \sum_{i=1}^{n} a_i \quad Bq \tag{0-8}$$

$A_U$ can be calculated by integrating f(t) between zero and infinity:

$$A_U = \sum_{i=1}^{n} A_{Ui} = \sum_{i=1}^{n} \frac{a_i}{\lambda_i} \quad Bq\,l^{-1}\,d \tag{0-9}$$

The $A_{Ui}$ values derived in the present case are listed in the last column of Table 0-4. So in the present case Equ. (0-9) results in

$A_u$ = *5.34 E08 Bq* $l^{-1}$ *d*

and thus in

$E = A_u$ *4.79 E-11 Sv = 0.0254 Sv = 25.4 mSv*

Again, these values are very close to the values derived from the simple hand calculation. Note that the first term contributes about 40 % and the second term about 60 % whereas the contribution of the third term is less than 1 %. This demonstates once more that in this case it is not necessary to extrapolate the measured data to infinity because more than 99 % of the dose is covered by the measured data.

**References**

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# Annex C

## Direct Dose Assessment

D Noßke

In special cases a direct dose assessment without use of a biokinetic model can be performed. This method is applicable only if:

- the distribution of activity is nearly homogeneously distributed in the body (as, for example, for caesium isotopes), or if the activity can be measured in an organ which is the main contributor to effective dose (for example, thyroid for iodine isotopes), and
- if the dose contribution of daughter nuclides is negligible or the daughter nuclides can be considered to be in equilibrium with the parent nuclide.

If these conditions are fulfilled, the effective dose is independent on the path of intake and largely independent of the model.

If the retention function in the source region S (which may be the total body) can be approximated from measurement results, the equivalent dose to the target tissue T can be calculated by

$$H_T = SEE(T \leftarrow S) \cdot \int_{t_B}^{t_E} A_S(t)dt$$

with:

$t_B$, $t_E$ begin and end of the period for which the committed dose is to be calculated

$A_S(t)$ retention function in source region S in Bq

$H_T$ dose equivalent in target tissue T in Sv

SEE(T←S) specific effective energy in Sv in tissue T per nuclear transformation in source region S (possibly with consideration of daughter nuclides in equilibrium with the parent nuclide)

The effective dose E can also be calculated by

$$E = \sum_T w_T \; SEE(T \leftarrow S) \cdot \int_{t_B}^{t_E} A_S(t)dt$$

with the tissue weighting factors $w_T$. For the assessment of total body measurements of $^{137}$Cs, for example a value of 1.1E-15 Sv/nuclear transformation can be derived from data from Cristy and Eckerman for $\sum_T w_T \cdot SEE(T \leftarrow TotalBody)$ which also considers the contribution of the daughter nuclide $^{137m}$Ba.

To apply this method, it is necessary to have a sufficient number of measurement results to be able to approximate the retention function $A_S$. If the dose contribution before the first and after the last measurement is negligible the integral can be derived by a simple numerical method:

$$\int_{t_B}^{t_E} A_S(t)dt = \frac{b}{2} \cdot \sum_{i=0}^{N-1} (M_{S,i} + M_{S,i+1}) \cdot (t_{i+1} - t_i)$$

with:

| | |
|---|---|
| $t_B$, $t_E$ | begin and end of the period for which the committed dose is to be calculated |
| $A_S(t)$ | retention function in source region S in Bq |
| $M_{S,i}$ | measured activity in source region S at time $t_i$ in Bq |
| $t_i$ | time of measurement in Tagen, $t_0=t_B$, $t_N=t_E$ |
| b | numerical constant: 86400 s/d |

If the dose before the first and after the last measurement is not negligible, these parts of the retention function must be estimated by some assumptions. A conservative assumption might be to assume only physical decay (and no biokinetic removal from the body) after the last measurement.

In general, this method can only be applied for radionuclides which can be measured by external measurements, for example by a total body counter. However, for some radionuclides the total activity in the body can be assessed by excretion measurement results. Such an example is tritiated water for which it is assumed that the activity concentration in total body water is the same as that in urine.

Unfortunately, with the new respiratory tract model of ICRP Publication 66, it is not possible to assess lung (and effective) doses from lung measurements if the effective dose is dominated by the lung dose as it was possible with the lung model of ICRP Publication 30. This is because due to the dosimetric model of ICRP Publication 66 the lung dose is a weighted mean of the doses to several regions of the respiratory tract and the activity content of these regions cannot be assessed separately by external measurements.

## Annex D

## DOSE ASSESSMENTS WITH THE DOSE PER UNIT CONTENT

Vladimir Berkovski

As described in the main text it has been demonstrated that in some circumstances it can be of value to assess doses directly from the quantity measured. This may be the whole body, organ/tissue content, an excreta sample of even an environmental measurement. This can be done by using the quantity "dose per unit content". This value will take onto account all the most up to date information on biokinetic and dosimetric models and can

serve as the final reporting value for official dose records and demonstration of compliance with dose limits.

Mathematically the quantity can be determined by the simple aggregation of the pair {*m(t)* , *e*} into the function *'Dose of record per unit content' z(t)*. The aggregation leads to the simplification of notations with the substitution of the ratio $\frac{e}{m(t)}$ by the new symbol $z(t)$:

$$z(t) = \frac{e}{m(t)},$$

(D.1)

The tabulated values of $z(t)$ for a range of radionuclides and types of materials are to be given in the OIR publications.

If only a single measurement is made, and the contribution of previous intakes to the measured quantity *M* is negligible, the effective dose *E*, associated with the intake, *I*, can be determined by:

$$E = Ie = M\frac{e}{m(t)} = M\,z(t)$$

(D.2)

The extended discussion of application of *'Dose per unit content' z(t)* for routine, special and task-related monitoring is given in the next section.

The dose per unit content concept is applicable also for the prospective/retrospective problems, when the *integrated air concentration*[1] *C* is the input information for dose assessments:

$$E = CAe = CZ$$

(D.3)

where C is *integrated air concentration* (Bq c $m^{-3}$),

*A is the breathing rate* ($m^3$ $c^{-1}$) and

*Z=Ae.*

Advantages of the *Dose per unit content* are:

[1] Integral of the air concentration over the time. In retrospective problems the measurement equipment gives this quantity as a primary data, the average air concentration is derived by normalisation by the time of sample accumulation.

- a simple one-step dose assessment for a majority of practical cases and support for data interpretation in a case of multiple measurements and/or multiple intakes;

- in many inhalation cases the correct selection of the monitoring program (e.g. urine vs. faecal data) may ensure the low sensitivity of the assessed dose to some important - but usually unknown - parameters of exposure (such as AMAD or even the chemical form), while the sensitivity of the reconstructed intake to these parameters will remain high. Such self-compensatory effects for the biases introduced consequently on the first and second steps of the assessment can be illustrated in: cases of inhaled aerosols of radioiodine with measurements of the thyroid gland; in cases of inhaled insoluble compounds with lung measurements; in cases of inhaled plutonium with urine measurements; and in many others cases (Berskovski et al, 2003].

- the use of the aggregated conversion function *z(t) prevents the* use of values of *m(t)* which do not match *e.*and helps to avoid the discussion of the uncertainty of the reconstructed intake *I,* which is essentially an internal parameter of dose assessments. *Training in the use of these tabulations is still needed however*

- the graphical presentation *of z(t)* for different types of bioassay measurements provides a convenient and simple tool for planning of the monitoring programs: e.g. the detection limits for activity could be easily converted to detection limits for the effective dose and visa verse. Measurement per unit dose may be found to be more useful for this.

**Routine monitoring**

Routine monitoring is a tool to identify and quantify events of a non-trivial internal exposure (doses above the recording level) in conditions of a chronic risk of internal contamination. Due to its “screening” nature, the routine monitoring program consists of series of equidistant bioassay measurements. Most nuclear facilities operate routine monitoring as an extensive basic-level program for a large number of workers. For the simplicity of the program the schedule of measurements usually is not associated with the pattern of the intake probability function attributed to the particular worker.

An unknown or uncertain time and duration of intake, as well as a nonstationarity of the intake probability function substantially complicate the interpretation of the routine monitoring data. The temporal-spatial variability of *parameters of exposure* (see Glossary), such as chemical forms of incorporated materials, dispersion of aerosols, effectiveness of individual protection equipment (e.g. obturation of masks), the rout of intake (inhalation/ingestion/intact skin) and others, makes matters worse.

One of the main objectives of radiation protection programs operated on facilities with a chronic risk of the internal contamination is to minimise the magnitude and probability of the internal exposure. The intake is a low-probability event on facilities which comply with this standard requirement.

The routine monitoring is supported by the special and task-related programmes. If routine monitoring identifies (suspects) the internal exposure above the investigation level, a special monitoring program is initiated. The design and data interpretation of the special monitoring program are normally more complex than in routine monitoring. Sophisticated or ad-hoc approaches for the ‘low-probability events’ are justified by higher demands to the reliability of assessments and by advantages of the small number of occurrence.

In a framework of the graded approach to the design of monitoring programs and interpretation of monitoring data the uncertainties of the routine monitoring data interpretation may be accepted if the resulting *dose of record* (see Glossary) is not underestimated by more than a factor of 3. For the basic monitoring operated in the range of exposure below the investigation level such a compromise between the uncertainty in the quantification of exposure and the complexity of data interpretation is considered as justified. The design of the routine monitoring program in accordance with the indicated criterion is discussed in [Ref].

One of important sources of uncertainty in the interpretation of routine monitoring data is the time of intake or the pattern of intake(s) occurred between two consecutive measurements. The Guideline recommends the assessment of a conventional quantity '*mid-point dose*' (see Glossary) for reporting and for initiating of a special monitoring program. For this purposes it is assumed that an acute intake took place in the middle of the monitoring interval of *T* days. If the contribution of previous internal exposures (intakes) to the measured quantity *M* is negligible (a typical situation for the normally operated facility), for a given measured quantity *M*, obtained at the end of the monitoring interval, the *mid-point dose E* associated with intake *I* is:

$$E = Ie = M\frac{e}{m(T/2)} = M\,z(T/2) \qquad \text{(D.4)}$$

where *m(T/2)* is the *bioassay response functions* for the mid-point of the monitoring interval.

In a general case the internal exposures (intakes) identified in preceding monitoring intervals may contribute into the bioassay quantity *M*. For a series of measurements in a routine monitoring programme, the following simple procedure may be followed:

- Determine the effective dose $E_1$ associated with the first monitoring interval (with the intake event on this interval).
- Predict the contribution to each of the subsequent measurements from exposure on the first monitoring interval.
- Subtract the corresponding contributions from all subsequent data.
- Repeat above for the next monitoring interval *i*.

It is convenient to assume that for each monitoring interval *i* the associated effective dose $E_i$ could be equal to 0 or positive. For a given measured quantity, $M(t_k)$, obtained at the end of the last monitoring interval *k*, the associated effective dose $E_k$ is:

$$E_k = \left( M(t_k) - \sum_{i=1}^{k-1} \frac{E_i}{z(t_k - \tau_i)} \right) z(t_k - \tau_k)\,, \qquad \text{(D.5)}$$

where $t_k$ is the time of measurement *k* (end of the last monitoring interval *k*); $\tau_i$ and $\tau_k$ are the time at mid-points of monitoring intervals *i* and *k*, respectively. If $M(t_k)$ is below the decision threshold (see [Ref]) or the result of background subtraction is negative, then $E_k \equiv 0$. If $E_k$ is higher that the derived investigation level, then special monitoring could be initiated so that the dose can be assessed more accurately.

For the simplicity the equation (D.5) does not consider the error of measurements and the propagation of errors in the consecutive subtraction, but for purposes of further

detailed assessments it is recommended to keep records of all bioassay data and estimated uncertainties, irrespectively to level of uncertainties (incl. the negative net values and values below the decision threshold)[2].

The equation (D.5) is used consecutively for each new result of routine monitoring data obtained during the reporting year and gives *N mid-point committed effective dose*s at the end of the reporting year. The *dose of record* is:

$$E^{\mathrm{Re}cord} = \sum_{i=1}^{N} E_i \ ,$$

**Special and task-related monitoring**

Usually, in special and task-related monitoring the bioassay data for a dose estimate consist of results for different measurements performed at different times, and even from different monitoring techniques, eg., direct and indirect measurements.

To determine the best estimate of a dose associated with the single intake, when the time of intake is known, it is first necessary to use given in OIR the tabulated values of effective dose $z(t_i)=e/m(t_i)$ for unit content. It is then required to determine the best estimate of the effective dose $E$, such that the product $E\ z^{-1}(t_i)$ (the predicted bioassay value) "best fits" the measurement data $(t_i, M_i)$. In cases where multiple types of bioassay data sets are available, it is recommended to assess the dose by fitting predicted values to the different types of measurement data simultaneously. For example, if urine and faecal data sets are available then, the intake is assessed by fitting predicted values to both data sets simultaneously.

The overview of statistical methods for data fitting is given in Annex E. Table D.1 summarizes key formulas of Annex E and illustrates the simplicity of the transition from the 'intake reconstruction approach' to the 'dose reconstruction approach' based upon the dose per unit content approach.

[2] The recommendations of IUPAC and IAEA [IAEA] in such cases are unambiguous: experimental results should not be censored, and they should *always* include quantitative estimates of uncertainty, following the guidelines of *GUM*

**Table D.1 Summary of notations and basic equations**

| | ***Reconstruction of dose*** | ***Reconstruction of intake*** |
|---|---|---|
| **Single measurement** (no contributions from previous intakes): Committed effective dose *E* | $E = M\, z(t)$ | $I = M\, m^{-1}(t)$; $E = I\, e$ |
| **Multiple measurements,** | | |
| Target value in the task of "best fits" of the measurement data | $E$ | $I$ |
| Predicted content in the task of "best fits" of the measurement data $(t_i, M_i)$ | $E\, z^{-1}(t_i)$ | $I\, m(t_i)$ |
| **Likelihood function** | $L_i(E) = P(M_i \mid E)$ | $L_i(I) = P(M_i \mid I)$ (E.1) |
| **Combined Likelihood Function** When there are *n* independent measurements, the combined likelihood function is the product of the likelihood functions for the individual measurements: | $L(E) = \prod_{i=1}^{n} L_i(E)$ | $L(I) = \prod_{i=1}^{n} L_i(I)$ (E.3) |

**Table D.1** (continue) **Summary of notations and basic equations**

| | **Reconstruction of dose:** |
|---|---|
| Likelihood function in assumption of lognormal distribution with geometric standard deviation $SF_i$ | $L_i(E) = \frac{1}{M_i \ln(SF_i)\sqrt{2\pi}} \exp\left[ -\frac{\left[\ln(M_i) - \ln(E\, z^{-1}(t_i))\right]^2}{2\left[\ln(SF_i)\right]^2} \right]$ |
| **Effective dose** | $E = \sqrt[n]{\prod_{i=1}^{n} E_i}$ |
| | **Reconstruction of intake:** |
| Likelihood function in assumption of lognormal distribution with geometric standard deviation $SF_i$ | $L_i(I) = \frac{1}{M_i \ln(SF_i)\sqrt{2\pi}} \exp\left[ -\frac{\left[\ln(M_i) - \ln(I\, m(t_i))\right]^2}{2\left[\ln(SF_i)\right]^2} \right]$ *(E.7)* |
| ***Intake*** | $I = \sqrt[n]{\prod_{i=1}^{n} I_i}$ (E.17) |

## GLOSSARY

**Bioassay response functions m(t):**
*Predicted bioassay value calculated for the time t after a unit acute intake.*

**Parameters of exposure:**
*Characteristics of radioactive materials (Type of Materials, GI absorption fractions, AMAD, etc.) and conditions of exposure (breathing rate, route of intake, nasal fraction of inhaled material, etc.), which are not associated with parameters of the systemic kinetics of radioactive materials in the body.*

**Committed dose to the reference worker:**
*Committed effective dose calculated with parameters of the systemic kinetic recommended in the OIR publications and by gender-averaging over male and female reference persons (see the definition of the effective dose). Parameters of exposure used for assessments of the committed dose to the reference worker could be generic (recommended by ICRP as default values) or site-specific.*

**Mid-point dose:**
*Committed effective dose to the reference worker associated with a single interval of routine monitoring and assessed in the assumption of an acute intake at the mid-point of the monitoring interval. The mid-point dose should be assessed with appropriated generic (recommended by ICRP as default values) or site-specific parameters of exposure. Site-specific reference parameters of exposure should be defined during planning of exposure and should not be the subject of the retrospective evaluation.*

**Dose of record:**
*Dose assigned to worker for purposes of for recording, reporting and demonstration of compliance:*

1. *In the routine monitoring program: sum of mid-point doses assessed for the year of reporting.*
2. *In the special or task-related monitoring programs: committed effective dose assessed to the reference worker assessed with the use of the known or reconstructed time of intake, appropriated generic (recommended by ICRP as default values) or site-specific parameters of exposure. In the routine monitoring program the site-specific parameters of exposure could retrospectively be re-assessed for the investigated case.*

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# Annex E

## Data Fitting

James Marsh and Guthrie Miller

Usually, the bioassay data for an intake estimate will consist of results for different measurements performed at different times, and even from different monitoring techniques, eg., direct and indirect measurements.

Data fitting techniques need to be applied to determine the best estimate of intake from multiple measurement data. Numerous statistical methods for data fitting are available [IAEA, 2004]. The two methods that are most widely applicable are the maximum likelihood method and the Bayesian approach. These two methods can be applied to the cases where it is assumed that the measurements are lognormally distributed as recommended in these guidelines. Other methods, such as the least squares method are special cases of the maximum likelihood method under certain assumptions. The standard equations given for the least square method apply to cases where the measurements are normally distributed and therefore do not strictly apply to these guidelines.

In this annex the maximum likelihood method and the Bayesian approach are reviewed. However, the Bayesian approach is not applied in these guidelines but its main advantage is that the method yields the probability distribution of the intake. It is therefore possible to determine a confidence interval on the best-estimated intake by applying the Bayesian approach.

As the likelihood function is the central statistical quantity for both approaches it is discussed first. The following topics are discussed.

- Likelihood function
- Likelihood function for “less than” measurements
- Maximum likelihood method
- Bayesian method
- Biokinetic model assumptions

To apply these methods it is necessary to calculate the predicted value, $m(t_i)$, for unit intake of the $i^{th}$ measured bioassay quantity. In each of these methods it is required to determine the best estimate of the intake, $I$, such that the product $I\ m(t_i)$ “best fits” the measurement data $(t_i, M_i)$.

The following sections assume that an acute intake, $I$ has occurred at a known time. It is also assumed that no previous intakes have occurred. However, Section 3.1 discusses how the maximum likelihood method can be extended to deal with previous intakes.

## 1 Likelihood function

A fundamental statistical quantity is the likelihood function $L_i(I)$, defined by

$$L_i(I) = P(M_i \mid I) \qquad \text{(E.1)}$$

where $P(M_i|I)\ dM_i$ is the probability of observing measurement value $M_i$ in the interval between $M_i$ and $M_i + dM_i$ given that the true value of the intake is $I$.
As an example, $P(M_i|I)$ might be given by a lognormal distribution:

$$L_i(I) = \frac{1}{M_i \ln(SF_i)\sqrt{2\pi}} \exp\left[ -\frac{[\ln(M_i) - \ln(I\ m(t_i))]^2}{2[\ln(SF_i)]^2} \right] \quad \text{(E.2)}$$

where $SF_i$ is the geometric standard deviation.

The meaning of $L_i(I)$ is that if the intake was indeed, $I$ and many measurements could, hypothetically, be repeated at the same time, $t_i$, then the distribution of the measurement results would be described by $L_i(I)$. The probability of a measurement result being in the interval between $M_i$ and $M_i + dM_i$ would then be $P(M_i|I)\ dM_i$. The likelihood function can therefore be determined by measurement if the true measurement value remains relatively constant with time [Moss et al. 1969].

When there are $n$ independent measurements, the combined likelihood function is the product of the likelihood functions for the individual measurements:

$$L(I) = \prod_{i=1}^{n} L_i(I) \quad \text{(E.3)}$$

Therefore, $L(I)$ is associated with the probability of observing all the data given the intake.

In practice a counting measurement is converted to activity by a normalisation (or calibration) factor, $C_n$. An important situation is where $C_n$, has a large uncertainty, which is assumed to be lognormal with a geometric standard deviation of $SF_B$. The overall uncertainty on the activity consists of two parts:

- the uncertainty due to counting statistics described by Poisson statistics, referred to as a Type A uncertainty, and
- the more subjective normalisation uncertainty, referred to as a Type B uncertainty.

Miller *et al.* (2002) gives the exact likelihood function for measurements involving counting. The function describes uncertainties due to counting statistics (Type A uncertainties) with a Poisson distribution whereas all other uncertainties (Type B uncertainties) are described with a single lognormal distribution. When $SF_B$ is small (less than about 1.6), the likelihood function is fairly well represented (even for small or zero counts) by a normal distribution:

$$L_i(I) = \frac{1}{\sqrt{2\pi}\sigma_i(I)} exp\left[ -\frac{(M_i - I\,m(t_i))^2}{2\sigma_i(I)^2} \right] \quad \text{(E.4)}$$

where the overall standard deviation is a function of $I$ given by

$$\sigma_i(I) = \sqrt{\sigma_{A_i}^2 + (I\,m(t_i)\ln(SF_B))^2} \quad \text{(E.5)}$$

The measurement central value and its Type A uncertainty are given in terms of measured quantities by

$$M = C_n\left(N_G - \frac{N_B}{R_B}\right)$$

(E.6)

$$\sigma_A = C_n\sqrt{N_G + \frac{N_B}{R_B^2}}$$

Where $N_G$ is the number of measured counts, $N_B$ is the number of measured background counts, $R_B$ is the ratio of background count time to sample count time, and $C_n$ is the normalization factor. In the expression for $_A$, the quantity $N_G + N /R_B^2$ under the square root sign is replaced by 1 if it is less than 1, which allows application of the formula even when the measured counts are small or zero.

When the number of sample counts $N_G$ is large enough (greater than about 10 counts with small background), the normal approximation to the Poisson likelihood function is approximately lognormal, because the normal and log normal distributions approach each other as the uncertainty goes to zero,

$$\frac{1}{\sqrt{2\pi}\,\sigma_{A_i}}\exp\left[-\frac{(M_i - I\,m(t_i))^2}{2\sigma_{A_i}^2}\right] \rightarrow \frac{1}{\sqrt{2\pi}\;M_i\ln(SF_{A_i})}\exp\left[-\frac{[\ln(M_i) - \ln(I\,m(t_i))]^2}{2[\ln(SF_{A_i})]^2}\right]$$

where $\ln(SF_{A_i}) = \frac{\sigma_{A_i}}{M_i}$.

The convolution of this log normal distribution with the log normal distribution of $C_n$ (having a geometric standard deviation of $SF_B$) then leads to another log normal distribution:

$$L_i(I) = \frac{1}{M_i\ln(SF_i)\sqrt{2\pi}}\exp\left[-\frac{[\ln(M_i) - \ln(I\;m(t_i))]^2}{2[\ln(SF_i)]^2}\right] \qquad \text{(E.7)}$$

where

$$SF_i = \exp\sqrt{[\ln(SF_{A_i})]^2 + [\ln(SF_{B_i})]^2}$$

This is the likelihood function recommend by the guidelines and is applicable to cases where the counts are relatively large (i.e. when $SF_A < 1.4$).

The log normal distribution has an important qualitative property that $I=0$ always has zero likelihood. The exact likelihood function, on the other hand, often has a significant nonzero value at $I=0$, and sometimes the maximum occurs at this point. Equation (E.7) should not be applied when the number of measured counts is small, since it rules out a small or zero intake in the interpretation of the data.
Sometimes measurements are given without uncertainties. In such cases it is necessary to make reasonable assumptions about both Type A and Type B uncertainties in order to calculate the likelihood function. In some cases only $SF_B$ needs be estimated because the Type A uncertainty is given or is known to be negligible. Some default values of $SF_B$ for various types of measurements are given in Tables E.1 and E.2.

**Table E.1 Typical values for the components of Type B log-normal uncertainty for *in vivo* measurements of radionuclides emitting low, intermediate and high photon energy radiation.**

| | Log-normal scattering factor $SF_B$ | | |
|---|---|---|---|
| Source of uncertainty | Low photon energy E < 20 keV | Intermediate photon energy 20 keV < E < 100 keV | High photon energy E > 100 keV |
| Variation of detector positioning | 1.2 | 1.05 | < 1.05 |
| Variation of background signal | 1.5 | 1.1 | < 1.05 |
| Variation in body dimensions | 1.5 | 1.12 | 1.07 |
| Variation of overlaying structures | 1.3 | 1.15 | 1.12 |
| Variation of activity distribution | 1.3 | 1.05 | < 1.05 |
| Calibration | 1.05 | 1.05 | 1.05 |
| Spectrum evaluation[(a)] | 1.15 | 1.05 | 1.03 |
| Total Type B[(b)] | 2.06 | 1.25 | 1.15 |

(a) HPGe detector spectra

(b) Total scattering factor for Type B uncertainties is given by $SF_B = \exp\left[\sqrt{\sum_j \ln^2(SF_j)}\right]$ where $SF_j$ is the scattering factor due to component j.

**Table E.2 Default values for the log. normal scattering factor SF for various types of measurement from different studies (Type B errors). Ranges are given in parentheses**

| Quantity | Log. normal scattering factor $SF_B$ |
|---|---|
| True 24-hr urine | 1.1[(a)] |
| Simulated 24-hr urine, creatinine or specific gravity normalised. | 1.6[(b)] (1.3[(c)] - 1.8[(d)]) |
| Spot urine sample | 2.0[(a)] |
| Faecal 24-hr sample | 3 (2 - 5)[(b)] |
| Faecal 72-hr sample | 2 (1.5 – 2.5)[(e)] |
| Chest count (see Table E.1) | 1.2 to 2.1 |

(a) Value given by Moss et al, 1969 based on plutonium in urine measurements of workers at Los Alamos.
(b) Value based on judgement and experience.
(c) At Los Alamos, Type B uncertainties, in terms of the coefficient of variation, for urine samples normalised using volume and specific gravity has been found to be 30% (i.e. a SF of 1.3).
(d) Value given by Riddell et al, 1994 based on plutonium in urine measurements of Sellafield workers. Because sampling procedures and measurements techniques have improve over the years recent measurements are likely to have a SF less than 1.8.
(e) SF values for 72-hr faecal samples are consistent with 24-hr faecal samples.

If Type A uncertainty is not negligible then it can be determined from equation E.6 by making reasonable assumptions about $C_n$, $N_B$, and $R_B$ based upon some knowledge of the measurement technique.

For an *in-vivo* measurement,

$$C_n(\mathrm{Bq\,count^{-1}}) = \frac{1}{\Delta T_{count}\, Y_{nuc} \varepsilon_{count} \varepsilon_{absorb}} \qquad \text{(E.8)}$$

where $T_{count}$ is the counting time (s), $Y_{nuc}$ is the yield per decay of the detected radiation, $_{count}$ is the counting efficiency ($Bq^{-1}$), and $_{absorb}$ is the efficiency of transmission of the detected radiation through overlaying tissues. The uncertainty $SF_B$ results mostly from individual geometric variations causing counting efficiency variations and variable absorption by overlaying tissue. For an *in-vivo* measurement, the quantity $R_B$ has the meaning of a general factor that relates measured background counts (for example counts measured in spectral channels on either side of a peak region) to a background interfering with the measurement.

For a urine or faecal radiochemical measurement normalised to 24 h excretion,

$$C_n(\mathrm{Bq\,d^{-1}count^{-1}}) = \frac{1}{\Delta T_{count}\, \Delta T_{excretion}\, \varepsilon_{count} \varepsilon_{chem}} \qquad \text{(E.9)}$$

where $T_{count}$ is the counting time (s), $T_{excretion}$ is the sample collection period (d), $_{count}$ is the counting efficiency ($Bq^{-1}$), and $_{chem}$ is the chemical recovery efficiency. The uncertainty $SF_B$ results mostly from uncertainty in the collection period, which depends on the sampling procedures and the techniques used to calculate collection period. One method of determining the collection period is from the ratio of volume or mass of a sample to the reference 24 hour excretion volume or mass. The reference values, for

males and females respectively, are: 1.6 litres and 1.2 litres for urine; and 150 g and 120 g for faeces (ICRP, 2002). For urine, the collection period can also be determined by normalising to the amount of creatinine excreted per day; 1.7 g and 1.0 g for males and females respectively [ICRP, 2002].

Given $C_n$, $N_B$, and $R_B$, the measured counts can be determined with the following equation obtained by rearranging equation E.6:

$$N_G = \frac{M}{C_n} + \frac{N_B}{R_B} \qquad \text{(E.10)}$$

Therefore, the Type A uncertainty can be determined from equation E.6 knowing $C_n$, $N_G$, $N_B$ and $R_B$:

$$SF_A = \exp\left(\frac{\sigma_A}{M}\right) \qquad \text{(E.11)}$$

where, $\sigma_A$ is given by equation E.6.

## 2 Likelihood function for 'less than' measurements

Both the maximum likelihood method and the Bayesian method can be applied to assess intakes from data sets consisting of positive values (i.e. values significantly greater than background) and values reported as below the lower limit of detection (<LLD). The likelihood function for a 'less than' measurement gives the probability that a measured value is reported as <LLD given the true intake is *I*.

The LLD, which is also referred to as minimum detectable activity, is an a priori calculated value, which specifies the minimum activity that can be detected by a defined measurement procedure [Health Physics Society (1996)]. Associated with the LLD is the decision threshold, $L_c$, which is also referred to as the critical level or decision level. If the measured value is below $L_c$ then a decision is made that the measured value is solely due to background and as a result the value is usually reported as being <LLD. In such a case the likelihood function is given by an integral quantity:

$$L_j(I) = \int_{M=-\infty}^{L_{c_j}} P(M \mid I)\, dM \qquad \text{(E.12)}$$

where:

$P(M/I)\ dM$ is the probability of observing a measurement value between $M$ and $M + dM$ given that the true intake is $I$.

$L_{c_j}$ is the decision threshold for a measurement carried out at $t_j$.

Given that the true intake is $I$, $L_j(I)$ is the probability of the measured value being below $L_{c_j}$ and therefore being reported as <LLD.

If a data set, of independent measurements, consist of $n$ data points that are not reported as <LLD (i.e. above $L_c$) and $p$ points reported as <LLD then the combined likelihood function for the data set is given by:

$$L(I)=\left(\prod_{i=1}^{n} P(M_i|I)\right)\left(\prod_{j=1}^{p}\int_{M=-\infty}^{L_{cj}} P(M|I)\,dM\right) \qquad \text{(E.13)}$$

Therefore, L(I) is associated with the probability of observing the data set given the intake.

For example, if it is assumed that the measurements are lognormally distributed (i.e. given by equation E.7) then the last parenthesis of equation E.13, which gives the probability of observing p measurements reported as <LLD, is given by:

$$\prod_{j=1}^{p}\int_{M=-\infty}^{L_{cj}} \frac{1}{M\ \ln(SF_j)\sqrt{2\pi}}\exp\left[-\frac{\left[\ln(M)-\ln(I\,m(t_j))\right]^2}{2\left[ln(SF_j)\right]^2}\right]dM$$

where $SF_j$ is the geometric standard deviation.

## 3 Maximum likelihood method

Using the maximum likelihood method, the "best fit" value of the intake, *I*, is that which maximises the likelihood function given by equation E.3 or equation E.13. In general, the maximum must be determined numerically. This can be accomplished by stepping *I* from 0 to some maximum value and searching for the maximum, or a more sophisticated numerical method may be employed.

It has been shown that applying the maximum likelihood method to data sets consisting of positive values and values reported as <LLD leads to unbiased estimates of the intake [Marsh et al. (2000)].

If the likelihood functions for all individual measurements are given by lognormal distributions (i.e. given by equation E.7) and none of the measurements are reported as <LLD, then the combined likelihood function is obtained by substituting equation E.7 into equation E.3:

$$L(I)=Const\times\exp\left[-\frac{\chi^2(I)}{2}\right] \qquad \text{(E.14)}$$

where

$$\chi^2(I)=\sum_{i=1}^{n}\frac{\left[\ln(M_i)-\ln(I\,m(t_i))\right]^2}{\left[\ln(SF_i)\right]^2}$$

The maximum of the likelihood function occurs where $\chi^2(I)$ is a minimum. In order to minimise $\chi^2$ this expression is differentiated with respect to ln(*I*) and set equal to zero. Re-arranging for *I* gives:

$$\ln(I)=\frac{\sum_{i=1}^{n}\frac{\ln(M_i/m(t_i))}{\left[\ln(SF_i)\right]^2}}{\sum_{i=1}^{n}\frac{1}{\left[\ln(SF_i)\right]^2}} \qquad \text{(E.15)}$$

Substituting $I_i = \frac{M_i}{m(t_i)}$ where $I_i$ is the intake calculated from the i[th] measurement gives:

$$\ln(I) = \frac{\sum_{i=1}^{n} \frac{\ln(I_i)}{[\ln(SF_i)]^2}}{\sum_{i=1}^{n} \frac{1}{[\ln(SF_i)]^2}} \qquad \text{(E.16)}$$

(8.14)

So ln*(I)* is a weighted average of ln($I_i$), the log of the individual intake estimates calculated from a single bioassay measurement. Various methods of weighting the individual determinations of intake $I_i$ to obtain an average "best fit" value of *I* look to the maximum likelihood method for their justification.

As an example, consider urine data where the scattering factor is dominated by Type B uncertainties (i.e. uncertainties other than counting errors such as calibration errors, and errors related to biological variability and sampling procedures). In this case, the SF can be assumed to be constant for each of the urine measurements, i.e. $SF_i = SF_u$ = constant. Therefore, the equation for the best estimate of intake (E.16) reduces to

$$\ln(I) = \frac{1}{n}\sum_{i=1}^{n} \ln(I_i) = \ln\left[\left(\prod_{i=1}^{n} I_i\right)^{\frac{1}{n}}\right]$$

That is

$$I = \sqrt[n]{\prod_{i=1}^{n} I_i} \qquad \text{(E.17)}$$

Therefore, when the values of the SF of the individual measurements can be considered equal to one another, the best estimate of intake is the geometric mean of the individual intake estimates.

Equation E.16 can also be applied to cases where data sets from different monitoring techniques are available. For example, if $n_u$ urine and $n_f$ faecal data are available and the scattering factors for the urine and faecal data are $SF_u$ and $SF_f$ respectively, then equation E.16 becomes:

$$\ln(I) = \frac{\sum_{i=1}^{n_u} \frac{\ln(I_i)}{(\ln(SF_u))^2} + \sum_{j=1}^{n_f} \frac{\ln(I_j)}{(\ln(SF_f))^2}}{\sum_{i=1}^{n_u} \frac{1}{(\ln(SF_u))^2} + \sum_{j=1}^{n_f} \frac{1}{(\ln(SF_f))^2}} \qquad \text{(E.18)}$$

where $I_i$ refers to the individual intake estimates from the urine data and $I_j$ refers to the individual intake estimates from the faecal data.

### 3.1 Extension to multiple intakes

Any previous intakes that influence the actual measurement result need to be taken into account. The guidelines propose to calculate the net value of the activity of the

radionuclide, $N_i$ by subtracting the contributions from previous intakes, $P_i$ from the measurement value (i.e. $N_i = M_i - P_i$). For simplicity, ignoring the uncertainty in $P_i$, equation E.16 can be applied to determine the best estimate of intake but with:

$$I_i = \frac{N_i}{m(t_i)} \qquad \text{(E.19)}$$

In applying equation E.16 to such cases, it is assumed that the net values of the activity are lognormally distributed with a given *SF*. It is acknowledged that the actual distribution of the net values is not lognormal because subtracting a value ($P_i$) from lognormally distributed values ($M_i$) does not result in another lognormal distribution.

An alternative approach is to fit the previous intakes as well as the intake of interest to all the data simultaneously using the maximum likelihood method. The maximum likelihood methodology can easily be extended to deal with several intakes. For *k* intakes, the likelihood function becomes *k*-dimensional, and the problem becomes of one of finding the set of *k* values of *I* (intake amounts) that maximises it. For example, equation E.7 becomes:

$$L_i(I) = \frac{1}{M_i \ln(SF_i)\sqrt{2\pi}} \exp\left[ - \frac{\left[\ln(M_i) - \{\ln(I_1\ m_1(t_i - \tau_1)) + \ln(I_2\ m_2(t_i - \tau_2)) + ... + \ln(I_k\ m_k(t_i - \tau_k))\}\right]^2}{2\left[\ln(SF_i)\right]^2} \right] \qquad \text{(E.20)}$$

where $\tau_1$, $\tau_2$, … $\tau_k$ are the times of each intake.

## 4 Bayesian method

The maximum likelihood method does not have a clear probabilistic basis and does not address the issue of uncertainty of *I* in a clear way. In the Bayesian method, the probability distribution of *I*, given the data, is directly calculated using Bayes theorem. Another probability distribution, *P(I)*, the prior probability distribution of *I* before the measurements is required. This distribution is logically necessary, since the nature of the measurement process is to update a pre-existing probability distribution. The prior probability distribution encapsulates information that may be available about the magnitude of *I* before the measurements are carried out. This information might come from air monitoring data, nasal swipes, or other workplace indicators, or it might just be the result of accumulated experience about the magnitude of intakes that have occurred at a particular facility. The prior distribution can be chosen to be completely flat, a broad log normal, or some other function of *I* depending on the circumstances [Miller et al. (2001)]. Generally speaking, as more and more data are accumulated, the prior becomes less and less important. Ideally, enough data is collected such that the influence of the prior on the final result is small.

By Bayes theorem, the probability distribution of *I* given the data (the posterior probability distribution) is given by:

$$P(I \mid \{M_i\}) = Const \times L(I) P(I) \qquad \text{(E.21)}$$

The constant multiplying factor is a normalizing factor obtained from the condition:

$$\int dI\, P(I \mid \{M_i\}) = 1 \qquad \text{(E.22)}$$

The central estimate of I is usually chosen to be the average or expectation value $<I>$, since this has the additivity property that the expectation value of the sum of two intakes is the sum of the expectation values of the individual intakes. The expectation value of intake is given by:

$$< I >= \int dI\, I P(I \mid \{M_i\}) = \frac{\int dI\, I L(I) P(I)}{\int dI\, L(I) P(I)} \quad , \tag{E.23}$$

which is evaluated by numerical integration over *I* from 0 to some maximum value.

The Bayesian method also allows the calculation of credible limits, for example, the 5% and 95% limits. These are the values of *I* that satisfy the following equations:

$$\int_0^{I_5} dI\, P(I \mid \{M_i\}) = 0.05$$
$$\int_0^{I_{95}} dI\, P(I \mid \{M_i\}) = 0.95 \tag{E.24}$$

## 5 Biokinetic model assumptions

The choice of biokinetic model and its parameter values that determines the predicted value function, $m(t_i)$, is very important. Sometimes the uncertainty surrounding biokinetic model assumptions dominates all other sources of uncertainty. The biokinetic model used to calculate $m(t_i)$ usually has a large number of parameters. The step-by-step approach described in these guidelines deals with this issue concerning the biokinetic model assumptions. A fundamental feature of this approach is that the model parameter values are adjusted systematically until an adequate fit is obtained to all the data. The initial assessment is carried out assuming default ICRP parameter values for the biokinetic models. The estimated intake is taken to be the "best estimate" unless the fits are inconsistent with the data (i.e. the fits are "inadequate"). If the fits are judged to be inadequate then the subsequent steps are carried out in turn, testing the fit after each step to see if further steps are required. These steps involve changing (or fitting) biokinetic model parameter values to improve the fits to the data. When the fits are consistent with the data the best estimate of the resulting effective dose is calculated with the same specific model parameter values that were assumed in the assessment of the intake.

An alternative approach is to deal with the uncertainty surrounding biokinetic model assumptions within the Bayesian method. One such approach is to enumerate various discrete choices of biokinetic model parameters, such as long term lung dissolution and particle size for an inhalation intake, by an integer index *l*. To simplify, only a fairly small number of possibilities *l* are usually considered, although the number need not be limited. Then, in addition to the intake *I*, the biokinetic model index *l* is another parameter that needs to be determined from the data. The Bayesian method is the clearest framework for incorporating this additional variable, since the choice of a discrete set of models to consider is equivalent to choosing a prior probability distribution for *l*. The posterior distribution of $(I,l)$ is then determined from the data just like the posterior distribution of *I*, using Bayes theorem:

$$P(I,l \mid \{M_i\}) = Const \times L(I,l) P(I) P(l) \tag{E.25}$$

where $L(I,l)$ is the likelihood function given by equation E.3 with the assumption of biokinetic type *l*. The prior probability distribution of *l* is usually chosen to be constant, or perhaps a default biokinetic type is assigned a larger probability.

The posterior probability of dose *D* is given by:

$$P(D \mid \{M_i\})\, dD = \sum_l P(I = \frac{D}{D_l}, l \mid \{M_i\}) \frac{dD}{D_l} \qquad \text{(E.26)}$$

where $D_l$ is the dose coefficient for biokinetic type *l*.

The expectation value of dose is given by:

$$< D >= \frac{\sum_l D_l \int dI\, I\, L(I,l) P(I) P(l)}{\sum_l \int dI\, L(I,l) P(I) P(l)} \qquad \text{(E.27)}$$

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# ANNEX F

## Obtaining an Unbiased Estimate of Intake in Routine Monitoring when the Time of Intake is Unknown

A Birchall and M Puncher

## 1. INTRODUCTION

A common problem in internal dosimetry occurs in routine monitoring, when it is required to estimate an intake from a measurement made at the end of a monitoring interval. In this situation, it is known that the intake occurred at some time in the monitoring interval (between *0* and T days), but the precise time of the intake, *t*, is unknown. Since the bioassay function (the function, *B(t),* that gives the predicted measurement value with time, per unit intake) depends on the time since the intake *t*, it follows that the estimate of intake will vary, depending on when it is assumed the intake took place (see Figure F1). Generally, if the time of intake, *t*, is known, then the estimate of intake, $\hat{I}$, is simply:

$$\hat{I} = \frac{M}{B(t)} \qquad (1)$$

Where *M* is the measurement value. However, if the time of intake, *t*, is unknown, how can the estimate be made? In Publication 54, ICRP (1997) argues, that in the absence of any information, the time of intake is equally likely to have occurred before the mid-point of the monitoring interval, than after it, and therefore suggests that in these situations, a value of *t*=T/*2* should be used. In this case, the estimate of intake is given by:

$$\hat{I} = \frac{M}{B(T/2)} \qquad (2)$$

While this is a pragmatic and simple approach, and is the one that is recommended for use used in the main text of this document because of its simplicity, it has been shown that it suffers from the limitation that if it is applied regularly, to an individual worker, then it will tend to overestimate the worker's real intake (Puncher et al, 2006). Since bioassay functions usually decrease with time, intakes that occur before T/2, will be underestimated, while those occurring after T/2 will be overestimated. Although the number of intakes occurring at times less than T/*2* will tend to be the same as those occurring at times greater than T/*2*, the cumulative effects of the underestimates will not exactly cancel those of the overestimates. This can be demonstrated by a simple example. Consider a case where a radioactive material remains in the body with a half life of 10 d, and let the monitoring interval (T) be 40 d. Assume that, in two consecutive 40 d intervals, one intake (1 Bq) occurs at 10 d, and the other (1 Bq) occurs at 30 d. In the first case, 3 half-lives will have elapsed by the end of the interval, and in the second case one half-life. Hence, use of the mid-point assumption will lead to an estimate of 0.5 Bq for the first intake and 2 Bq for the second, i.e. a total intake of 2.5 Bq. However, the actual intake was 2 Bq. The tendency to over or underestimate a quantity is known as biasing, and is widely acknowledged in the scientific literature (Strom, 2003).

In order to overcome the biasing which occurs when the mid-point assumption is made, the United States Department of Energy (USDOE), in their DOE standards (Strom, 2003), adopts a different approach, referred to here as the Mean-Inverse Method. This can be illustrated by dividing the monitoring interval into *n* equal time intervals. Thus, if it is assumed that the intake occurs in the $^{i}$th interval, then the estimate of intake will be:

$$\hat{I}_i = \frac{M}{B(t_i)} \qquad (3)$$

Where $t_i$ is a time taken to be representative of the $i^{th}$ interval. It is then argued that since the probability of an intake occurring in any interval ($t_i$) is the same, an unbiased estimate of intake is the arithmetic mean of all of these intakes:

$$\hat{I} = \frac{1}{n}\sum_{i=1}^{n}\frac{M}{B(t_i)} \qquad (4)$$

On the face of it, this approach looks reasonable. However, it has been recently shown (Puncher et al, 2006), that rather than lead to an unbiased estimate of a worker's intake, this method actually leads to intakes which tend to be more biased than those obtained with the mid-point assumption. Furthermore, it is also shown that the only unbiased estimate of the intake, $\hat{I}$ , is numerically equal to that which would be obtained by assuming that the intake was delivered continuously, at a constant rate, throughout the monitoring interval. Thus, if $B^c$ is the bioassay quantity calculated at the end of the monitoring interval, due to a unit intake delivered at a constant rate throughout that interval, then the unbiased estimate of intake is given by:

$$\hat{I} = \frac{M}{B^c} \qquad (6)$$

This method is referred to here as the Constant-Chronic Method.

## 2. MONTE CARLO CALCULATIONS

Given that the 'Mid-point Method' and the 'Mean-Inverse Method' both lead to biased estimates of intake, it is useful to investigate their accuracy in different situations. In order to do this, Monte Carlo simulations have been performed (Puncher et al, 2006). in order to quantify the degree of bias.

The simulations were carried out as follows. For each run, i, a time of intake $t_i$ was chosen randomly within the monitoring period, and current ICRP models were used to predict the bioassay quantity (*i.e.* the simulated measurement) at the end of the monitoring period. The reverse process was then carried out ie. the same models were then used to estimate the intake $\hat{I}_i$, using (a) the Mid-point Method, (b) the Mean-Inverse Method, and (c) the Constant-Chronic Method. This was repeated 10,000 times. For a method to be unbiased, the expectation (or arithmetic mean) value of all of the estimates, $I_i$, must tend to the true intake. A positive bias indicates that on average, the method will tend to overestimate by a factor >1.

The magnitudes of the bias, for a selection of radionuclides together with typical monitoring programmes, have been calculated for both the Mean-Inverse and Mid-point methods. These are summarised in Table F1. It can be seen that the Mid-Point Method and the Mean-Inverse Method both result in a positive bias. The results for the Constant-Chronic Method are not shown since these were all unbiased (*i.e.* =1.00).

Another advantage of Monte Carlo simulations is that they not only quantify the degree of bias, but they can be used to show how close the individual estimates $\hat{I}_i$ are to the true value (1 Bq). This property is known as the *efficiency* of the method. More precisely, the efficiency is the standard deviation of the $\hat{I}_i$'s around the true value. The

more efficient a method is, the closer the estimates are to the central value. It has been shown (Puncher et al, 2006). that the 'Constant-Chronic Method is the most efficient of the three methods discussed here. A detailed mathematical analysis of the Constant-Chronic method has been carried out, and it was been shown that the same results apply:

(a) when intakes are not unit intakes, but themselves random intakes chosen from a well-defined distribution;
(b) when the number of intakes in any given interval is greater than 1;
(c) for any bioassay function

Since these three situations are likely to include the real situation, it can be concluded that the best method of estimating an intake when the time of intake is unknown, is the Constant Chronic Method.

Although it is clear from the analysis carried out, that the Constant-Chronic Method is the only unbiased method considered here, it is not immediately obvious why the Mean-Inverse method should lead to biased estimates. This can be explained by applying a Bayesian Approach, as described below.

## 3. THE PROBLEM FROM A BAYESIAN PERSPECTIVE

At first sight, the Mean-Inverse Method seems to be a reasonable approach to obtaining an unbiased estimate of the intake. First, the monitoring interval is divided into equal time segments, $t_i$. Then, once the measurement is made at the end of the monitoring interval, one has to decide in which of the time segments $t_I$ the intake occurred. Since this is not known, it is assumed that it could have occurred in any of the time segments, with equal probability. On the face of it, this looks like a reasonable assumption, since there is no reason to prefer one time segment over another. The average of all of the estimated intakes, $\hat{I}_i$, for each time $t_i$ is therefore taken. Given the reasonableness of the method, it is surprising that this method of estimation does give a biased estimate of intake.

Bayes' theorem draws a distinction between two types of probability distribution used to describe an event: the '*prior*' distribution and the '*posterior' distribution'*. The prior distribution, $\Phi(t).dt$, represents the state of knowledge about a variable (in our case the time of intake, $t$) before any measurement is made. If nothing at all is known about the value of $M$, then it is reasonable to assume that all times of intake are equally probable, i.e. $\Phi(t).dt$ is constant between 0 and T. This is often termed a *uniform prior*, and is used to describe the lack of knowledge or ignorance about $t$. On the other hand, the posterior distribution represents the state of knowledge about $t$ after a measurement is known, $p(t|M).dt$. Bayes' theorem mathematically relates the posterior distribution to the prior distribution in the following way:

$$p(t \mid M)\, dt = \frac{p(M \mid t)\Phi(t)}{P(M)}\, dt \qquad (7)$$

Where:

| | |
|---|---|
| $P(M)$ | The total probability of observing measurement $M$. |
| $p(M\|t)$ | The likelihood function: the probability density function associated with the probability of observing $M$ given $t$. |
| $\Phi(t).dt$ | The prior distribution: the probability that the intake occurred between t and t+dt (before the measurement M is made). |

Bayes' theorem thus shows how the initial knowledge of $t$ (i.e. the prior distribution ) is mathematically modified to express the new knowledge of $t$ (i.e. the posterior

distribution) once the measurement has been made. The key point here is that if $\Phi(t).dt$ is constant, then $p(t|M).dt$ cannot be constant, and *vice versa*. In other words, it is contradictory to hold the beliefs simultaneously that both $\Phi(t).dt$ and $p(t|M).dt$ are uniform. It can now be seen that the difference between the Mean-Inverse Method and the Constant-Chronic Method is that the former makes the assumption that AFTER a measurement is made, all times $t$ are equally probable (i.e $p(t|M).dt$ is constant) while the latter makes the assumption that BEFORE a measurement is made, all intake times are equally probable (i.e. $\Phi(t).dt$ is constant).

The reason why the Monte Carlo simulations exhibited a zero bias for the Constant-Chronic Method is because, by generating random times of intake before each measurement was calculated, the program implicitly assumed a uniform prior distribution of time, i.e. $\Phi(t).dt$, is constant. Presumably, one could devise an alternative Monte Carlo simulation where $p(t|M).dt$ is constant, which would demonstrate that the Mean-inverse method was unbiased.

Thus, in order to decide which of these assumptions is the more rational, it is necessary to decide which assumption most closely corresponds to reality. If we are to choose the assumption that $p(t|M).dt$ is constant (implicit in the Mean-Inverse method), then we are in effect saying that all nuclear facilities are designed in such a way that the probability of a release event happening at time $t$, follows a precise mathematical probability distribution ($\Phi(t).dt$), with some times more probable than others. Moreover, that after a release happens at unknown time $t$, and is subsequently measured to be $M$, then the resulting posterior probability distribution, $p(t|M)$, as calculated by Bayes' theorem, just happens to cancel exactly, such that $p(t|M).dt$ becomes constant over the monitoring interval. It is argued (Puncher et al, 2006). that a more realistic prior assumption to adopt, if the time of intake cannot be inferred, is one in which a release is equally probable at any time. It follows that the method that should be used to give an unbiased estimate of intake is the Constant Chronic method. In fact, it has been shown[(4)] that the Mean-Inverse Method can be derived directly from Bayes' theorem using the 'unrealistic' assumption that $p(t|M).dt$ is constant.

## 4. MEASUREMENT UNCERTAINTY

So far, all of the preceding arguments have been based on the assumption that there is no error on the actual measurement m. Of specific concern here, is the observation by Blanchardon and Molokanov (2006), that if the measurement error is assumed to be lognormally distributed, then this could lead to a bias in the estimate of intake.

For any given bioassay function and monitoring interval, the estimated intake is directly proportional to the recorded measurement (e.g. if the measurement is doubled, then the estimated intake will also be doubled). Clearly, if there is a bias in the measurement procedure itself, then this bias will be transferred directly to a bias in the estimated intake. A bias in the measurement procedure is defined as occurring when in a sequence of repeated measurements, the mean value of the measurements does not tend to the true value.

If the measurement error is assumed to be normally distributed around the true value, then it follows that for that no further bias will occur, and the CC method will lead to an unbiased estimate of the intake. This has been confirmed independently by Blanchardon and Molokanov (2006). However, if one assumes that the measurement errors are lognormally distributed around a median value (as is currently recommended by the IDEAS guidelines (Doerfel, 2006), then it can be shown that this will indeed lead to a bias in the estimate of intake. It is worth noting that this bias is not caused by the constant chronic method itself, but rather from a bias in the reporting of the

measurement result itself. This bias would be introduced in all three of the methods described here.

Fortunately, for the case of a lognormally distributed measurement error with a geometric standard deviation of $\sigma_g$ then it has been shown[6] that it is possible to eliminate this bias by multiplying the observed measurement M by a correction factor Fc given by

$$F_c = \exp(-g \frac{M}{e^{\frac{\sigma_g^2}{2}}}$$

where *M* is the measurement.

It can be shown that, after applying this technique, an unbiased estimate of intake can be obtained from $M_{bc}$ using the Constant Chronic Method[6].

## 5. CONCLUSIONS

Theoretical considerations aside, the choice of an appropriate method of estimating intakes is also dependent on broader considerations such as ease of implementation, and in many situations, the Mid-point method is adequate, considering other uncertainties. Indeed, this is the default method suggested in the main text of this document. However, it is always preferable to avoid bias, and if possible, the Constant Chronic method should be used. If a software program is being used to estimate the intake, then this method can be simply applied by selecting the intake regime to be constant and chronic throughout the monitoring interval.

It should be noted that the Constant Chronic method should only be applied when the time of intake is unknown or cannot be inferred from the measurement data. Where multiple measurements are available, such as often occurs if special monitoring has been initiated after a high measurement has been obtained during routine monitoring, an estimate of the actual time of intake might be possible if the bioassay function is described by more than one exponential term.

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**Table A1. Magnitude of the bias of intake estimation for the Mid-point and Mean-inverse methods.**

| | | | | Mid-point Method | | | Mean-inverse Method | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Monitoring Interval (T days) | | | | | |
| | | | | 14 | 30 | 60 | 14 | 30 | 60 |
| Nuclide | Absorption | $f_1$ | Bioassay quantity | | | | | | |
| $^{3}$H (HTO) | N/A[(a)] | N/A[(a)] | Whole Body | 1.04 | 1.18 | 1.79 | 1.07 | 1.37 | 2.85 |
| $^{239}$Pu | M | 0.0005 | Urine | 2.01 | 2.52 | 1.95 | 2.13 | 2.05 | 1.73 |
| $^{239}$Pu | S | 0.00001 | Urine | 1.77 | 1.89 | 1.51 | 1.69 | 1.56 | 1.35 |
| $^{238}$U[(b)] | M | 0.02 | Urine | 1.05 | 1.14 | 1.28 | 1.05 | 1.14 | 1.24 |
| $^{137}$Cs | F | 1 | Urine | 1.07 | 1.04 | 1.03 | 3.9 | 1.01 | 1.25 |
| $^{131}$I | F | 1 | Thyroid | 1.01 | 1.31 | 2.82 | 1.15 | 1.80 | 8.28 |
| $^{125}$I | F | 1 | Thyroid | 1.00 | 1.00 | 1.04 | 1.06 | 1.03 | 1.10 |

[(a)] Route of intake specified as injection for tritiated water, inhalation for all other nuclides.
[(b)] Based on a regime for $^{238}$U described in "Monte Carlo Calculations". The time of intake was assumed to occur between *0* and T-*2* days, where T is the measurement time and the end of the monitoring interval.

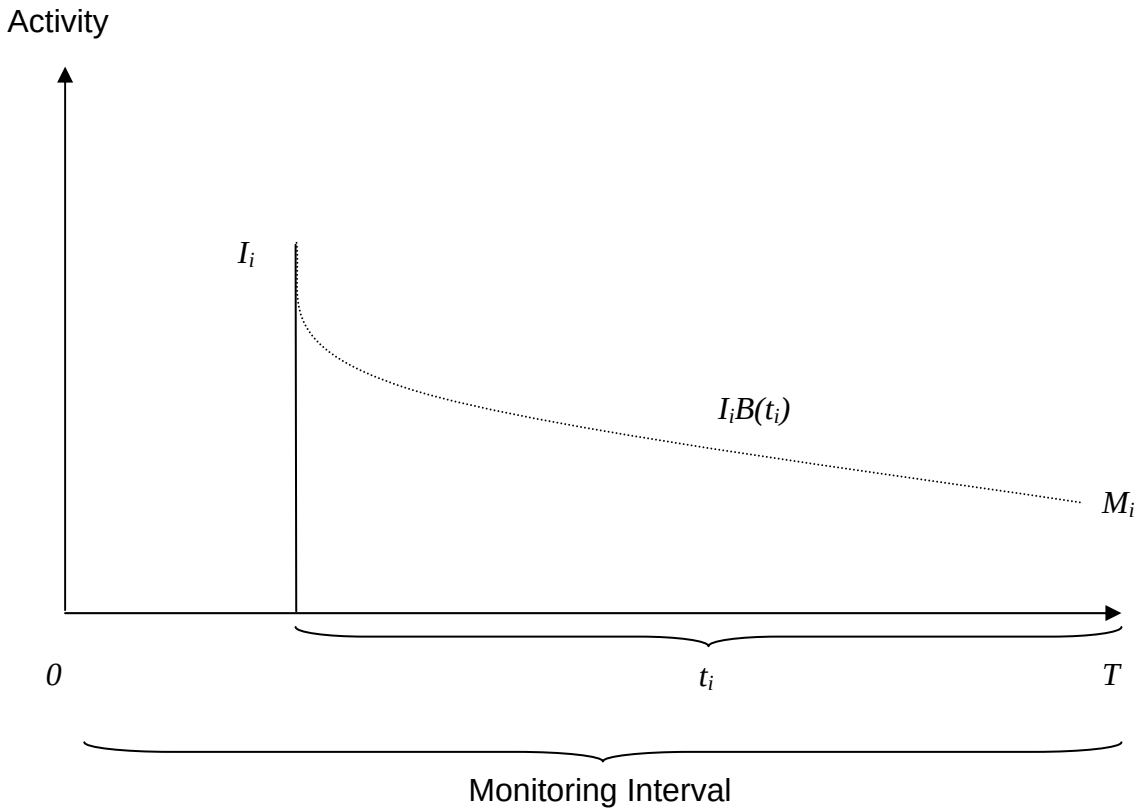


**Figure 1.** Measurement $M_i$ made at the end an arbitrary monitoring interval (*0-T* days) from an intake, $I_i$, at time (T-$t_i$.) The function, $B(t_i)$, gives the measurement per unit intake at time $D$.

# ANNEX G

## Structured Approach to Advanced Dose Assessment

(H Doerfel + IDEAS Working Group)

The information given below develops the structured approach to dose assessment given in Chapter 8 for situations in which more detailed analysis is required, typically above Level 1 and Stage 4. Dose criteria are given as an example of the need to make the details of the assessment proportionate to the expected dose. Levels are <1 mS, ≥1 mSv to 6 mSv, and ≥6 mSv. The levels chosen in a particular facility while be determined by local circumstances.

## G5 Stage 5

### G5.1 *Stage 5A [Fig. G1]*

Simple evaluation carried out using parameter values chosen *a priori* before the evaluation is carried out. The procedure is very similar to the "Standard procedure" (Stage 3). The main difference is that in a special procedure there should be more than one measurement.

***Step 5.1*** : Identification and preparation of measurement data. It is expected that there will be more than one measurement available for a special assessment ($M_i$ for i = 1 to n). It is therefore important that realistic uncertainties are assigned to the data ("scattering factor", SF, Step 2.1) There may be more than one type of measurement (urine, faeces, etc), and there may be measurements of more than one radionuclide involved in the exposure.

***Step 5.2*** : (As Step 2.3 for a single measurement.) The contributions ($P_i$) from all previous intakes of the radionuclide considered are calculated, taking into account all pathways of intake, and all intakes of mixtures where the radionuclide was involved. The net values ($N_i$) of the radionuclide are calculated by subtracting $P_i$ from the measured value $M_i$.

***Step 5.3*** : (As Step 3.2 in the Standard Procedure, Stage 3, except for time of intake). Case or site specific parameter values should be assigned as far as they are available. Such a priori information needs to be well established and documented. Examples might include the Activity Median Aerodynamic Diameter (AMAD) – if it has been determined by appropriate air sampling (eg. cascade impactor), or the time of intake, if potential exposure was limited, or an incident was known to occur. Otherwise the following default parameter values should be used:

- Mode of intake: Single intake
- Absorption Type and $f_A$ value: defaults according to OIR document (in preparation)
- Particle size: 5 µm AMAD

***Step 5.4*** : Time of intake known/unknown. If the special procedure was initiated as a result of a known incident (and hence the time of intake is known) then a simple assessment (Step 5.5) should be carried out which is consistent with the Standard evaluation (Stage 3). If the special procedure was initiated as a result of a routine measurement being inconsistent with previous assessment (Step 2.6) or for say a dose ≥1 mSv resulting from the Standard evaluation (Step 3.4) where the time of intake is probably not known, then further special procedures (Stage 5B) are needed for more detailed evaluation of the case.

***Step 5.5*** : (As step 3.3 in the Standard Procedure, Stage 3, but for more than one measurement). Using the assigned *a priori* parameter values, an estimate of intake $I_i$ is obtained by dividing the net value $N_i$ by the appropriate retention or excretion function. The best estimate of intake is given by Equ. 8.11.

***Step 5.6*** : If the effective dose estimated in step 5.5 is less than 1 mSv, there is no need for further investigation (Step 5.6.1). (The dose from the intake under consideration, rather than the "annual dose" as in Step 3.4, is the criterion, because intakes requiring special assessment procedures should be unusual for any individual worker.) Otherwise further special procedures (Stage 5B) are needed for more detailed evaluation of the case.

***Step 5.6.1*** : The results in terms of intake and committed effective dose from Step 5.6 are recorded together with the corresponding parameter values from Step 5.3.

## G5.2 *Stage 5B [Fig. G2]*

In this stage, procedures are described for varying the two main factors related to the inhaled material: the AMAD and absorption Type, and also the time of intake, if not known, using the measurement data (*a posteriori*).

In this Stage, and in Stage 5C that follows, parameter values are selected on the basis of the "fit" of the model predictions to the observations (data). A check on whether the fit is adequate is used to decide whether to stop the evaluation, or to go on to further steps. A measure of the "Goodness of fit" (GOF) and the criteria for deciding that the fit is good enough are therefore critical issues. There may be conflict between "harmonisation" and "optimisation". Generally the better the data (quality and quantity) the more likely it is that a statistical test will show that the data are inconsistent with the model. If the data are poor it is more likely that the model will fit – in the extreme case of a single measurement any model will fit. It is therefore important that there should be sufficient data available for assessment of a significant dose, and the higher the dose, the better the data should be. Proposals are therefore made for the minimum amounts of data that would be acceptable ("sufficient").

***Step 5.7*** : Are there are sufficient data? As noted in the introduction, criteria for the "sufficient" number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose. In this Step, the numbers for the range 1 mSv <Dose < 6 mSv are appropriate, because a Special procedure is generally initiated on the assumption that the dose could exceed 1 mSv, and doses greater than 6 mSv are considered in steps 5.11.2 and 5.12.2 below.

***Step 5.7.1*** : Get additional dose relevant data. This assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those available are insufficient. When the additional data have been obtained, a simple re-evaluation Stage 5A is made.

***Step 5.8*** : Is the time of intake known? As noted in the introduction, there are two main alternative routes through this stage of the process, according to whether or not the time of intake is known. Generally, Special Procedures follow from an identified incident for which the time is known: Steps 5.9 to 5.11, and if necessary 5.13 are followed. However, previously unidentified intakes are sometimes found through eg. routine monitoring, and so the time of intake is unknown, or known only to be within a certain interval. Step 5.12 and if necessary 5.14 are followed, but provide less opportunity for a posteriori characterisation of the material.

***Step 5.9*** : Are early lung and faeces data available? During the first few days after an accidental inhalation intake of a relatively insoluble material (Type M or Type S) most of the activity will be in the respiratory tract, or cleared through the GI tract to the faeces. In the event of such an incident with potential for a significant intake it would therefore be expected that if feasible, measurements of lung and faeces would be made. If the cumulative fecal excretion over the first few days, and a measurement on which the initial lung deposit can be estimated are available, then an estimate can be made of the effective AMAD (Step 5.10).

***Step 5.10*** : Derive effective AMAD from early lung and faeces data. The main effect of the aerosol AMAD is to determine the relative amounts deposited in (i) the upper respiratory tract (eg nose) which is rapidly cleared to faeces, and (ii) the lungs. Hence the ratio of early fecal clearance to lung activity can be used to estimate the AMAD (see Section 7.3.4).

***Step 5.11*** : Assessment of dose by fitting the absorption Type. At this step the AMAD has been determined according to the information available: default 5 µm AMAD, a priori characterisation, or a posteriori derivation. The other main characteristic of the inhaled material is the absorption Type. An a priori assignment of the absorption Type has been made in Step 5.3 above according to the OIR document (in preparation) based on what is known of the chemical form of the inhaled material. A check is made on the Goodness of fit (Step 5.11.1) using this default absorption Type. If it is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 5.11.2 etc. If it is not, then other absorption Types are tried, as follows.

The ICRP default absorption Types for particulate materials: F (fast), M (moderate) and S (slow) each represent very wide ranges of absorption rates. There can be large differences between the actual absorption behaviour of a material and that assumed for the default to which it is assigned, which can greatly affect lung retention and urinary excretion. Evaluations are therefore made assuming each of the other default Types available for that element. In each case a check is made on the Goodness of fit (Step 5.11.1). If the fit is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 5.11.2 etc. (If more than one absorption Type fits, the one giving the best fit is chosen).

***Step 5.11.1*** : Is the goodness of fit acceptable? If it is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. Otherwise further special procedures (Step 5.13 onwards) are needed for more detailed evaluation of the case.

***Step 5.11.2*** : Is the dose is less than 6 mSv? If the effective dose estimated in Step 5.11 is less than 6 mSv, there is no need for further investigation (Step 5.11.3). Otherwise further special procedures (Step 5.11.4 onwards) are needed for more detailed evaluation of the case.

***Step 5.11.3*** : The results in terms of intake and committed effective dose from Step 5.11 are recorded together with the corresponding parameter values from Step 5.11.

***Step 5.11.4*** : Check that there are sufficient data, and get more if necessary. This is similar to steps 5.7 and 5.7.1. Criteria for the “sufficient” number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose level. In this Step, the numbers for Dose > 6 mSv are appropriate.

To get additional dose relevant data assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those available are insufficient. When the additional data have been obtained, further special procedures (Step 5.13 onwards) are needed for more detailed evaluation of the case.

***Step 5.12*** : Assessment of dose by simultaneous fitting of the time of intake and the absorption Type. As can be seen this Step is reached through 5.8 when the time of intake is unknown. At this step the AMAD has been determined according to the information available: default 5 µm AMAD or a priori characterisation.

The other main characteristic of the inhaled material is the absorption Type. An a priori assignment of the absorption Type has been made in Step 5.3 above according to the OIR document based on what is known of the chemical form of the inhaled material. A check is made on the Goodness of fit (Step 5.11.1) using this default absorption Type and the default time of intake. (As in Step 3.2: Mid-point of the monitoring interval, i.e. the mid-point of the time range between the date of the measurement being considered and the date of either the previous measurement or the beginning of monitoring). If the fit is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 5.12.2 etc. If it is not, then other absorption Types and times of intake are tried, as follows.

The ICRP default absorption Types for particulate materials: F (fast), M (moderate) and S (slow) each represent very wide ranges of absorption rates. There can be large differences between the actual absorption behaviour of a material and that assumed for the default to which it is assigned, which can greatly affect lung retention and urinary excretion. Evaluations are therefore made assuming each of the default Types available for that element, for several times of intake spanning the period of possible intake. In each case a check is made on the Goodness of fit (Step 5.12.1).

If an acceptable fit is found, it is likely that acceptable fits will be found for a range of times of intake, and therefore the combination of absorption Type and time of intake giving the best fit is chosen. The dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 5.12.2 etc.

***Step 5.12.1*** : Is the Goodness of fit acceptable? If the goodness of fit is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. Otherwise further special procedures (Step 5.14 onwards) are needed for more detailed evaluation of the case.

***Step 5.12.2*** : Is the dose less than 6 mSv? If the effective dose estimated in Step 5.12 is less than 6 mSv, there is no need for further investigation (Step 5.12.3). Otherwise further special procedures (Step 5.12.4 onwards) are needed for more detailed evaluation of the case.

***Step 5.12.3*** : The results in terms of intake and committed effective dose from Step 5.12 are recorded together with the corresponding parameter values from Step 5.12.

***Step 5.12.4*** : Check that there are sufficient data, and get more if necessary. This is similar to steps 5.7 and 5.7.1. Criteria for the “sufficient” number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose level. In this Step, the numbers for Dose ≥6 mSv are appropriate.

To get additional dose relevant data assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those

available are insufficient. When the additional data have been obtained, further special procedures (Step 5.14 onwards) are needed for more detailed evaluation of the case.

***Step 5.13*** : Assessment of dose by fitting a mixture of absorption Types. This is an extension of Step 5.11, to give greater flexibility in fitting by consider a mixture of absorption Types.

This Step may have been reached through Step 5.11.1, because an acceptable fit was not obtained with any single absorption Type. In that case combinations should be tried by inspection, trial and error etc. If more than one fits (Stage 5C Step 5.15), the mixture of absorption Types giving the best fit is chosen.

Alternatively, this Step may have been reached through Steps 5.11.1 and 5.11.2, because the estimated dose is ≥6 mSv, and more data may have been obtained. If so then as much of the procedure as necessary should be repeated: evaluate using in turn: the *a priori* default absorption Type; another absorption Type; and a combination of absorption Types, until an adequate fit is obtained.

***Step 5.14*** : Assessment of dose by simultaneous fitting of the time of intake and a mixture of absorption Types. This is an extension of Step 5.12, to give greater flexibility in fitting by considering a mixture of absorption Types.

This Step may have been reached through Step 5.12.1, because an acceptable fit was not obtained with any single absorption Type and time of intake. In that case combinations of absorption Type should be tried. If more than one fits (Stage 5C Step 5.15), the mixture of absorption Type giving the best fit is chosen. If an acceptable fit is found, it is likely that acceptable fits will be found for a range of times of intake, and therefore the combination of the mixture of absorption Types and time of intake giving the best fit is chosen.

Alternatively, this Step may have been reached through Steps 5.12.1 and 5.12.2, because the estimated dose is ≥6 mSv, and more data may have been obtained. If so then as much of the procedure as necessary should be repeated: evaluate using in turn: the a priori default absorption Type and default time of intake; all absorption Types and variable time of intake; and a combination of absorption Types and variable time of intake, until an adequate fit is obtained.

### G5.3 *Stage 5C [Fig. G3]*

In this stage, an advanced evaluation is carried out. It applies to cases where there are comprehensive data available. The fundamental approach is that the model parameter values are adjusted systematically, in a specific order, until the goodness of fit is acceptable (i.e. the fits obtained to all the data are not rejected by the specified criteria). If the fit is acceptable then the estimated intake is taken as the best estimate and the effective dose is calculated with the same model parameter values that were assumed in the assessment of intake. These results (intake and committed effective dose) are then recorded together with the corresponding parameter values (Step 5.15.1). Thus after each Step in which a parameter value is varied (5.17 to 5.22) there is a corresponding Step (5.17.1 to 5.22.1 respectively) to test the goodness of fit. Since these are all very similar to Step 5.15, explanatory text is not given.

It is recommended, in cases where multiple types of bioassay data sets are available, that the intake and dose are assessed by fitting predicted values to the different types of data simultaneously. If the time of intake is not known, the time of intake should be varied simultaneously with the other parameters (Steps 5.16 – 5.20).

***Step 5.15*** : Is the goodness of fit acceptable?

If the goodness of fit is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. In this step, the criteria are those appropriate to a dose of ≥ 6 mSv, because such doses may have been evaluated in Steps 5.11.2 and 5.12.2. The effective dose is then calculated with the same model parameter values that were assumed in the assessment of intake. However if the fit is rejected then proceed to next (step 5.16).

***Step 5.15.1:*** Record dose with all parameter values; if the effective dose is above 6 mSv, all organ doses have to be calculated and recorded.

***Step 5.16*** : Determine specific HRTM parameters

For materials that are moderately to very insoluble (typically absorption Types M or S), determine specific values for $f_r$ and $s_s$ by fitting $f_r$, $s_s$ and intake to the data with $s_r$ fixed at 100 $d^{-1}$. For most materials there is no evidence for binding to the respiratory tract so the bound fraction $f_b$ is taken to be zero. However, if relevant values of $s_r$ and/or of $f_b$ and $s_b$ have been determined from *in vivo* experimental data then use these values.

***Step 5.17*** : Determine specific $f_A$ value

Generally, it is not justifiable to change the $f_A$ value as well as the HRTM absorption parameter values. Occasionally, for inhaled materials that are relatively insoluble, it is necessary to reduce the value of $f_A$ so that the predicted systemic activities or urinary excretion rates are consistent with the data. Consideration should, however, be given to correlating changes to $f_r$ and $f_A$, to reduce the number of variables, e.g. setting $f_r = f_1$.

***Step 5.18*** : Determine specific HRTM particle transport values

The parameter values that describe particle transport from the respiratory tract in the HRTM were based so far as possible on human experimental data, which enable typical lung clearance rates to be determined for a year or so after particle deposition in the lungs. However, the values were chosen to be average values for healthy non-smokers. The experimental data from which they were derived show considerable inter-subject variation even among healthy subjects, and indicate that clearance would generally be slower in smokers and patients with lung disease (ICRP, 1994a). If there are comprehensive lung and/or faecal excretion data available, it may be necessary to vary particle transport rates to improve the fits to the data.

It should be noted that adjusting particle transport rates also affects the amount absorbed into blood, because clearance from the lung is competitive between absorption into blood and particle transport to the alimentary tract. Thus in some cases it is necessary to readjust HRTM absorption parameter values (i.e. repeat step 5.16) after varying the particle transport rates (see Section 3.7).

***Step 5.19*** : Determine specific HATM transit parameter values

The parameter values in the ICRP HATM again represent typical values, and there will be considerable inter (and intra-) subject variations. The transit time through the alimentary tract affects the amount in the whole body and the amount excreted in the faeces within the first few days following inhalation or ingestion. If there are comprehensive early data it may be necessary to alter the HATM parameter values to obtain a reasonable fit to the data.

***Step 5.20*** : Adjust systemic biokinetic model parameter values

Again, model parameters values were derived by ICRP to represent population averages, and there are likely to be individual variations, which will result in differences between predicted values and data, independently of the biokinetics of the respiratory or alimentary tract. This might well arise for very soluble materials, where particle transport rates have little effect. Individual whole body retention half-times have been reported for intakes of tritiated water, and caesium-137. However, for actinides, with sufficiently comprehensive data, individual differences from model predictions might be observed for retention in liver and skeleton, or in the ratio between deposition in such organs, and urinary excretion.

It is emphasised that this is the last step, so adjusting the systemic biokinetic model parameter values should only be considered after varying the HRTM and HATM parameter values, and $f_A$ value (Steps 5.18, 5.19 and 5.20). If the goodness of fit test results in the fit being rejected according to the specified criteria then consult other experts (Step 5.21). Otherwise the results (intake and committed effective dose) are then recorded together with the corresponding parameter values (Step 5.15.1).

## G6 Stage 6

### G6.1 *Stage 6A [Fig. G4]*

In this stage, a simple evaluation is carried out using parameter values chosen *a priori*: before the evaluation is carried out. The procedure is very similar to the "Standard procedure" (Stage 3). The main difference is that in a special procedure there should be more than one measurement.

**Step 6.1**: Identification and preparation of measurement data. It is expected that there will be more than one measurement available for a special assessment ($M_i$ for i = 1 to n).

It is therefore important that realistic uncertainties are assigned to the data ("scattering factor", SF, Step 2.1) There may be more than one type of measurement (urine, faeces, etc), and there may be measurements of more than one radionuclide involved in the exposure.

**Step 6.2**: (As Step 2.3 for a single measurement.)

The contributions ($P_i$) from all previous intakes of the radionuclide considered are calculated, taking into account all pathways of intake, and all intakes of mixtures where the
radionuclide was involved. The net values ($N_i$) of the radionuclide are calculated by subtracting $P_i$ from the measured value $M_i$.

**Step 6.3**: (As Step 3.2 in the Standard Procedure, Stage 3, except for time of intake).

Case or site specific parameter values should be assigned as far as they are available. Such *a priori* information needs to be well established and documented. Examples might include the fraction of the ingested activity that is absorbed into the systemic circulation: the "$f_A$ value" – if it has been determined by an appropriate *in vivo* experiment (although such experiments are uncommon), or the time of intake, if potential exposure was limited, or an incident was known to occur. Otherwise the following default parameter values should be used:

- Mode of intake: Single intake
- $f_A$ value: defaults according to the ICRP OIR Document.

**Step 6.4**: Time of intake known/unknown. If the special procedure was initiated as a result of a known incident (and hence the time of intake is known) then a simple assessment (Step 6.5) should be carried out which is consistent with the Standard evaluation (Stage 3). If the special procedure was initiated as a result of a routine measurement being inconsistent with previous assessment (Step 2.6) or a dose >1 mSv resulting from the Standard evaluation (Step 3.4) where the time of intake is probably not known, then further special procedures (Stage 6B) are needed for more detailed evaluation of the case.

**Step 6.5**: (As step 3.3 in the Standard Procedure, Stage 3, but for more than one measurement). Using the assigned *a priori* parameter values, an estimate of intake $I_i$ is obtained by dividing the net value $N_i$ by the appropriate retention or excretion function. The mean of the value of $I_i$ gives the "best estimate" of intake. (Usually the geometric mean – see "Best estimation of intake".) Using the same assigned *a priori* parameter values the committed effective dose is calculated by multiplying the "best estimate" of intake by the appropriate dose coefficient (dose per unit intake).

**Step 6.6**: If the effective dose estimated in Step 6.5 is less than 1 mSv, there is no need for further investigation (Step 6.6.1). (The dose from the intake under consideration, rather than the "annual dose" as in Step 3.4, is the criterion, because intakes requiring special assessment procedures should be unusual for any individual worker.) Otherwise further special procedures (Stage 6B) are needed for more detailed evaluation of the case.

**Step 6.6.1**: The results in terms of intake and committed effective dose from Step 6.6 are recorded together with the corresponding parameter values from Step 6.3.

### G6.2 *Stage 6B [Fig. G5]*

In this stage, procedures are described for varying the main factor related to the ingested material, the $f_A$ value, and also the time of intake, if not known, using the measurement data (*a posteriori*).

In this Stage, and in Stage 6C that follows, parameter values are selected on the basis of the "fit" of the model predictions to the observations (data). A check on whether the fit is adequate is used to decide whether to stop the evaluation, or to go on to further Steps. A measure of the "Goodness of fit" (GOF) and the criteria for deciding that the fit is good enough are therefore critical issues. There may be conflict between "harmonisation" and "optimisation". Generally the better the data (quality and quantity) the more likely it is that a statistical test will show that the data are inconsistent with the model. If the data are poor it is more likely that the model will fit – in the extreme case of a single measurement any model will fit. It is therefore important that there should be sufficient data available for assessment of a significant dose, and the higher the dose, the better the data should be. Proposals are therefore made for the minimum amounts of data that would be acceptable ("sufficient").

As seen in the flow chart, there are two alternative routes through this stage of the process, according to whether or not the time of intake is known.

**Step 6.7**: Are there are sufficient data? As noted in the introduction, criteria for the "sufficient" number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose. In this Step, the numbers for the range 1 mSv <Dose <6 mSv are appropriate, because a Special procedure is generally initiated on the assumption that the dose could exceed 1 mSv, and doses ≥ 6 mSv are considered in Steps 6.13 onwards.

**Step 6.7.1**: Get additional dose relevant data. This assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those available are insufficient. When the additional data have been obtained, a simple re-evaluation (Stage 6A) is made.

**Step 6.8**: Is the time of intake known? As noted in the introduction, there are two alternative routes through this stage of the process, according to whether or not the time of intake is known. Generally, Special Procedures follow from an identified incident for which the time is known (Step 6.9). However, previously unidentified intakes are sometimes found through e.g. routine monitoring, and so the time of intake is unknown, or known only to be within a certain interval. Step 6.10 is followed, but provides less opportunity for *a posteriori* characterisation of the material.

**Step 6.9**: Assessment of dose by selecting the default $f_A$ value. An *a priori* assignment of the $f_A$ value has been made in Step 6.3 above according to the ICRP OIR Document recommendations based on what is known of the chemical form of the ingested material. A check is made on the Goodness of fit (Step 6.11) using this default $f_A$ value. If it is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 6.12 etc. If it is not, then other $f_A$ values are tried, as follows.

For some elements (e.g. cobalt, strontium, uranium, plutonium) the ICRP OIR Document gives different $f_A$ values for different chemical forms. It is proposed that evaluations are made assuming each of the other default $f_A$ values available for that element. In each case a check is made on the Goodness of fit (Step 6.11). If the fit is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 6.12 etc. (If more than one $f_A$ value fits, the one giving the best fit is chosen).

**Step 6.10**: Assessment of dose by simultaneous fitting of the time of intake and the $f_A$ value. As can be seen this Step is reached through Step 6.8 when the time of intake is unknown.

An *a priori* assignment of the $f_A$ value has been made in Step 6.3 above according to the ICRP OIR Document recommendations based on what is known of the chemical form of the inhaled material. A check is made on the Goodness of fit (Step 6.11) using this default $f_A$ value and the default time of intake. (As in Step 3.2: Mid-point of the monitoring interval, i.e. the mid-point of the time range between the date of the measurement being considered and the date of either the previous measurement or the beginning of monitoring). If the fit is acceptable, then the dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 6.12 etc. If it is not, then other default $f_A$ values and times of intake are tried, as follows.

For some elements (e.g. cobalt, strontium, uranium, plutonium) the ICRP OIR Document gives different $f_A$ values for different chemical forms. It is proposed that evaluations are made assuming each of the other default values available for that element, for several times of intake spanning the period of possible intake. In each case a check is made on the Goodness of fit (Step 6.11).

If an acceptable fit is found, it is likely that acceptable fits will be found for a range of times of intake, and therefore the combination of $f_A$ value and time of intake giving the best fit is chosen. The dose is calculated with the same model parameter values that were assumed in the assessment of intake and the process moves to Step 6.12 etc.

**Step 6.11**: Is the Goodness of fit acceptable? If the goodness of fit is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. In this Step, the criteria are those appropriate to a dose of less than 6 mSv, because doses greater than 6 mSv are considered in Steps 6.13 onwards. Otherwise further special procedures (Step 6.14 onwards) are needed for more detailed evaluation of the case.

**Step 6.12:** Is the dose less than 6 mSv? If the effective dose estimated in Step 6.9 or 6.10 is less than 6 mSv, there is no need for further investigation (Step 6.12.1). Otherwise further special procedures (Step 6.13 onwards) are needed for more detailed evaluation of the case.

**Step 6.12.1**: The results in terms of intake and committed effective dose from Step 6.12 are recorded together with the corresponding parameter values from Step 6.9 or 6.10. If the effective dose is 6 mSv or above, all organ doses would have to be calculated and recorded.

### G6.3 *Stage 6C [Fig. G6]*

In this stage, an advanced evaluation is carried out. It applies to cases where there are comprehensive data available. The fundamental approach is that the model parameter values are adjusted systematically, in a specific order, until the goodness of fit is acceptable (i.e. the fits obtained to all the data are not rejected by the specified criteria). If the fit is acceptable, then the estimated intake is taken as the best estimate and the effective dose is calculated with the same model parameter values that were assumed in the assessment of intake. These results (intake and committed effective dose) are then recorded together with the corresponding parameter values (Step 6.12.1). Thus after each Step in which a parameter value is varied (6.14 to 6.16) there is a corresponding Step (6.14.1 to 6.16.1 respectively) to test the goodness of fit. Since these are all very similar, explanatory text is only given for Step 6.14.1.

If the time of intake is unknown, then by the start of this Stage it may have been assessed, based on simultaneous fitting of the model to the data with the $f_A$ value (Step 6.10). In that case, if any of the parameter values are changed in the Steps below, the time of intake should be re-assessed.

It is recommended, in cases where multiple types of bioassay data sets are available, that the intake and dose are assessed by fitting predicted values to the different types of data simultaneously.

**Step 6.13**: Check that there are sufficient data, and get more if necessary. This is similar to Steps 6.7 and 6.7.1 (Stage 6B). Criteria for the “sufficient” number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose level. In this Step, the numbers for Dose ≥ 6 mSv are appropriate.

To get additional dose relevant data assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those available are insufficient. (For historical cases, where it is not possible to obtain more measurements, it should be recorded that the data are insufficient, and therefore the result should be treated with caution.) When the additional data have been obtained, further special procedures (Step 6.14 onwards) are needed for more detailed evaluation of the case.

**Step 6.14:** Determine specific $f_A$ value

The $f_A$ value is the main variable related to the ingested material. The default values recommended by ICRP are generally typical values representing the wide ranges

that might arise in practice, especially when a single value is given for all chemical forms of an element. Alimentary tract absorption can also vary according to factors such as how recently a meal was taken. Hence it is reasonable to consider values different from the ICRP default. If sufficiently comprehensive data are available, especially if it is possible to estimate both the intake and the total amount absorbed into blood (e.g. if early faecal and urine data are available), then it may be necessary to change the $f_A$ value to obtain a reasonable fit to the data. If the time of intake is not known, the $f_A$ value and the time of intake should be varied simultaneously as in Step 6.10.

**Step 6.15:** Determine specific HATM transit parameter values

The parameter values in the ICRP HATM represent typical values, and there will be considerable inter (and intra-) subject variations. Moreover, at noted in Step 6.1, while for ease of computation transit through the alimentary tract is represented by a series of compartments that clear exponentially, in practice, the movement is more like "slug" flow. It is therefore unlikely that individual daily faecal clearance measurements in the first few days after intake will follow the predicted pattern. The transit time through the alimentary tract affects the amount in the whole body and the amount excreted in the faeces within the first few days following inhalation or ingestion. If there are comprehensive early data it may be necessary to alter the HATM parameter values to obtain a reasonable fit to the data. If the time of intake is not known, the HATM parameter values and the time of intake should be varied simultaneously in analogy to Step 6.10.

**Step 6.15.1**: Is the goodness of fit acceptable?

If the goodness of fit is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. In this step, the criteria are those appropriate to a dose of ≥ 6 mSv, because such doses may have been evaluated in Step 6.9 or 6.10. The effective dose is then calculated with the same model parameter values that were assumed in the assessment of intake. However if the fit is rejected then proceed to the next Step (6.15).

**Step 6.16:** Adjust systemic biokinetic model parameter values

Systemic model parameter values were derived by ICRP to represent population averages, and there are likely to be individual variations, which will result in differences between predicted values and data, independently of the biokinetics in the GI tract. Individual whole body retention half-times have been reported for intakes of tritiated water and caesium-137. For actinides, with sufficiently comprehensive data, individual differences from model predictions might be observed for retention in liver and skeleton, or in the ratio between deposition in such organs, and urinary excretion.

It is emphasised that this is the last Step, so adjusting the systemic biokinetic model parameter values should only be considered after varying the GI tract model parameter values, and $f_A$ value (Steps 6.14 and 6.15). If the goodness of fit test results in the fit being rejected according to the specified criteria then send the case to the IDEAS website. Otherwise the results (intake and committed effective dose) are then recorded together with the corresponding parameter values (Step 6.12.1). If the time of intake is not known, the systemic biokinetic parameter values and the time of intake should be varied simultaneously in analogy to Step 6.10.

## G7 Stage 7

### G7.1 *Stage 7A [Fig. G7]*

Initial evaluation carried out using parameter values chosen *a priori* before the evaluation is carried out. The main goal of this Stage is to get a first rough estimate of the potential exposure due to the wound deposition in order to have a basis for further decisions on the treatment of the case, i.e. excision of contaminated tissue and/or application of chelation therapy to reduce the potential exposure.

***Step 7.1***: Identification and preparation of measurement data. Typically, at this stage only *in vivo* data on the activity at the wound site are available. If the intake occurred some time before starting the evaluation it is expected that there will be more than one measurement available for a special assessment ($M_i$ for i = 1 to n). In this case there may be more than one type of measurement (urine, faeces, etc), and there may be measurements of more than one radionuclide involved in the exposure. All these measurements should be corrected for contributions from previous intakes.

***Step 7.2***: Case or site specific parameter values should be assigned as far as they are available. Such *a priori* information needs to be well established and documented. If no such information is available, conservative parameters should be assigned, i.e. those parameters which would result in the maximum dose according to the NCRP wound model.

[Note: need to clarify: is this "conservative" for prospective calculations? It is often found that parameters that are conservative prospectively are not retrospectively and that the latter change with time.

***Step 7.3***: Calculate the dose with the parameter values as defined by Step 7.2. Typically the time of intake is known for wound cases. If the time of the intake is not known, special evaluation procedures have to be applied similar to those described for inhalation cases.

***Step 7.4***: If the effective dose estimated in step 7.3 is less than 1 mSv, there may be no need for further investigation (Step 7.4.1).

***Step 7.4.1***: The results in terms of intake and committed effective dose from Step 7.3 are recorded together with the corresponding parameter values from Step 7.2.

***Step 7.5***: The dose assessment will be used by medical professionals to determine the need for any treatment and further follow up.

***Step 7.6***: Unlike inhalation and ingestion cases, the committed dose due to wound deposition can be reduced significantly by surgery, i.e. by excision of the contaminated tissue. This option would be considered by medical professionals.

### G7.2 *Stage 7B [Fig. G8]*

In this stage, a procedure is described for adjusting the wound absorption type and, if necessary, the systemic biokinetic model parameters using the measurement data (*a posteriori*). Parameter values are selected on the basis of the "fit" of the model predictions to the observations (data). A check on whether the fit is adequate is used to decide whether to stop the evaluation, or to go on to further steps. A measure of the "Goodness of fit" (GOF) and the criteria for deciding that the fit is good enough are therefore critical issues. Similar to Stage 5, there may be conflict between "harmonisation" and "optimisation". Generally the better the data (quality and quantity) the more likely it is that a statistical test will show that the data are inconsistent with the model. If the data are poor it is more likely that the model will fit – in the extreme case of a single measurement any model will fit. It is therefore important that there should be sufficient data available for assessment of a significant dose, and the higher the dose, the better the data should be.

***Step 7.7***: Are there sufficient data? Criteria for the "sufficient" number (and types) of relevant data, duration of monitoring etc, are proposed according to the dose (see Section… and Table 7.12; in Table 7.12 there should be a corresponding footnote for "Type of monitoring", such as "In wound cases: wound measurement instead of whole body or lung measurement"). In this Step, the numbers for the range 1 mSv <Dose <6 mSv are appropriate.

***Step 7.7.1***: Get additional dose relevant data. This assumes that the evaluation is being carried out in real time, so that the opportunity exists to obtain more measurements if those available are insufficient.

***Step 7.8***: Assessment of dose by fitting the wound absorption type according to the NCRP wound model.

***Step 7.9***: Is the goodness of fit acceptable? If it is acceptable (i.e. the fit obtained is not rejected by the specified criteria) then the estimated intake is taken as the best estimate. Otherwise further special procedures (Step 7.13 onwards) are needed for more detailed evaluation of the case.

***Step 7.10***: Is the dose less than 20 mSv? If the effective dose estimated in Step 7.9 is less than 20 mSv, there is no need for further investigation (Step 7.11). Otherwise the possibilities for reducing the dose by excision of the contaminated tissue should be checked out (Step 7.12 onwards).

***Step 7.11***: The results in terms of intake and committed effective dose from Step 7.8 are recorded together with the corresponding parameter values.

***Step 7.12***: As pointed out already at Step 7.6, the committed dose due to wound deposition can be reduced significantly by surgery, i.e. by excision of the contaminated tissue. This option should be considered if the committed dose would exceed the dose limit (**point for discussion)**.

***Step 7.13***: Adjust systemic biokinetic model parameter values and estimate dose. If there is clear evidence that the discrepancy between the measured values and the model prediction is due to some systemic biokinetic model parameters, these parameters should be adjusted accordingly.

***Step 7.14***: If the adjustment results in an acceptable fit, the results in terms of intake and committed effective dose are recorded together with the corresponding parameter values (Step 7.11).

***Step 7.15***: Wound cases are in general complicated and so it is good anyway to consult other experts.

### G7.3 *Stage 7C [Fig. G9]*

**Point for discussion: This Stage is very important for the dose estimation of wound cases with significant exposures. It is, however, closely related to monitoring issues. Thus, it is a question whether or not to include the Stage here. Maybe it would be better to condense Stage 7C to one box which could be implemented into the flowcharts of Stage 7A and 7B. The following text describes shortly the scope and the content of this Stage. This will be considered further.**

The efficiency of wound measurements depends appreciably on the composition and the distribution of the activity deposited at the wound site. In many cases a radionuclide with relative high gamma-emissions (i.e. Am-241) is used as tracer for

other nuclides with low gamma-emissions (i.e. Pu-238, Pu-239). In those cases knowledge of the nuclide composition is essential for both the calibration of the counting system and the evaluation of the counting results. Thus, in this stage the evaluation is repeated taking into account the real nuclide composition as derived from radiochemical analysis of the excided tissue.

**Step 7.16**: At first the excided tissue is measured with the counting system in the same wound measuring geometry as applied for the person's measurement.

**Step 7.17**: The nuclide composition and the total activity is determined by radiochemical analysis of the excised tissue.

**Step 7.18**: From the results obtained in Steps 7.16 and 7.17 the case specific efficiency of the counting system is derived. The initial wound counting data are adjusted according to the case specific counting efficiency.

**Step 7.19**: The residual activity at the wound site is measured taking into account the case specific counting efficiency. With the adjusted data the evaluation is repeated according to Stage 7B.

## Figures

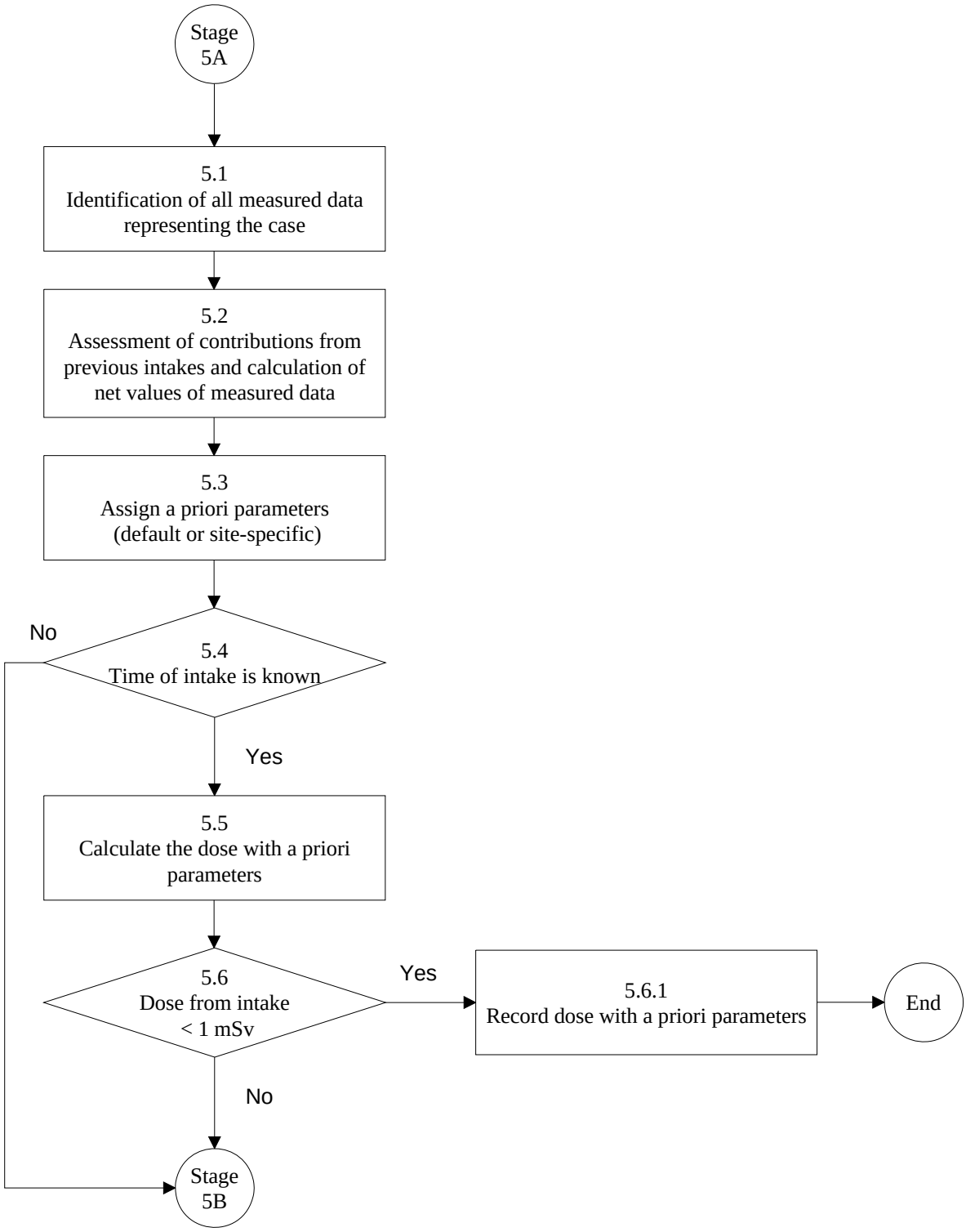


Fig. G1 Stage 5A – Structured approach to dose assessment

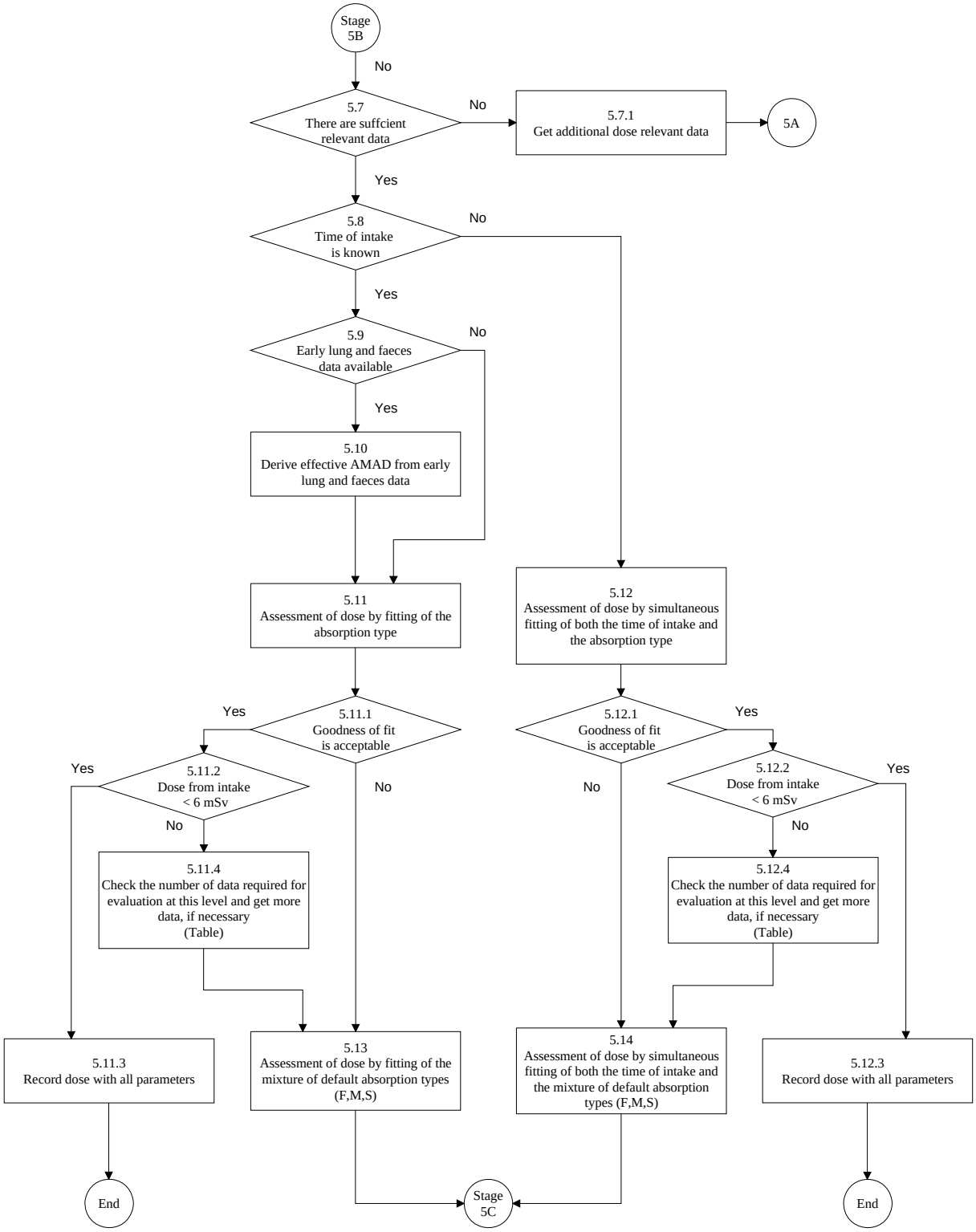


Fig. G2 Stage 5B – Structured approach to dose assessment

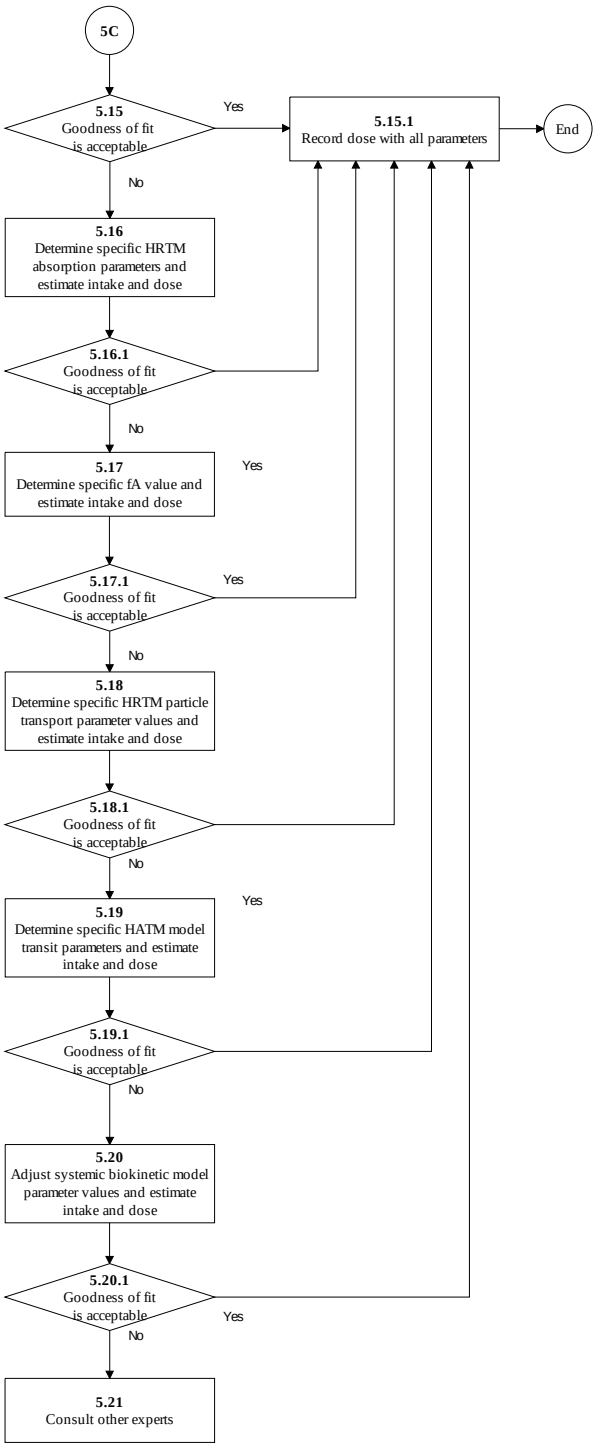


Fig. G3 Stage 5C – Structured approach to dose assessment

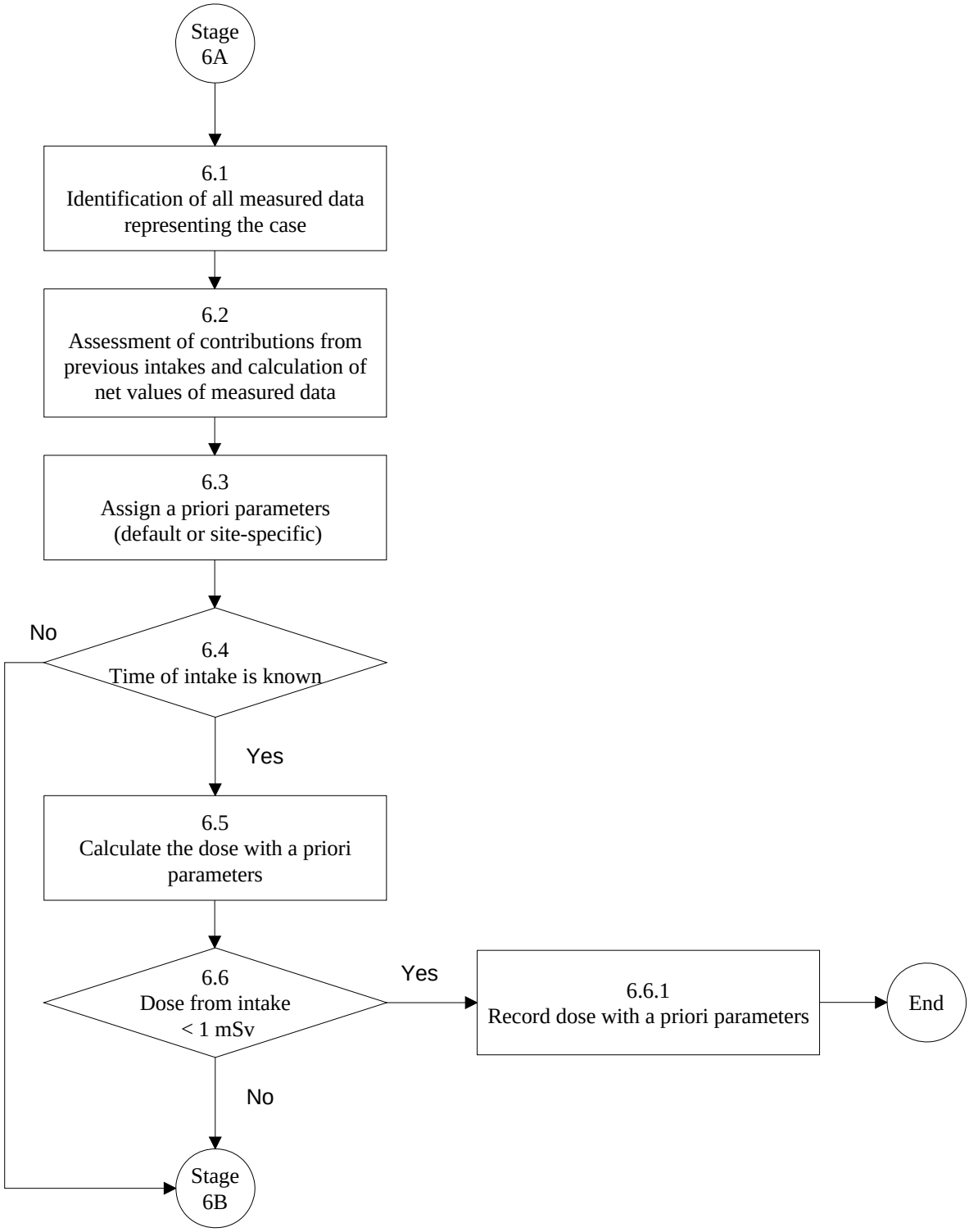


Fig. G4 Stage 6A – Structured approach to dose assessment

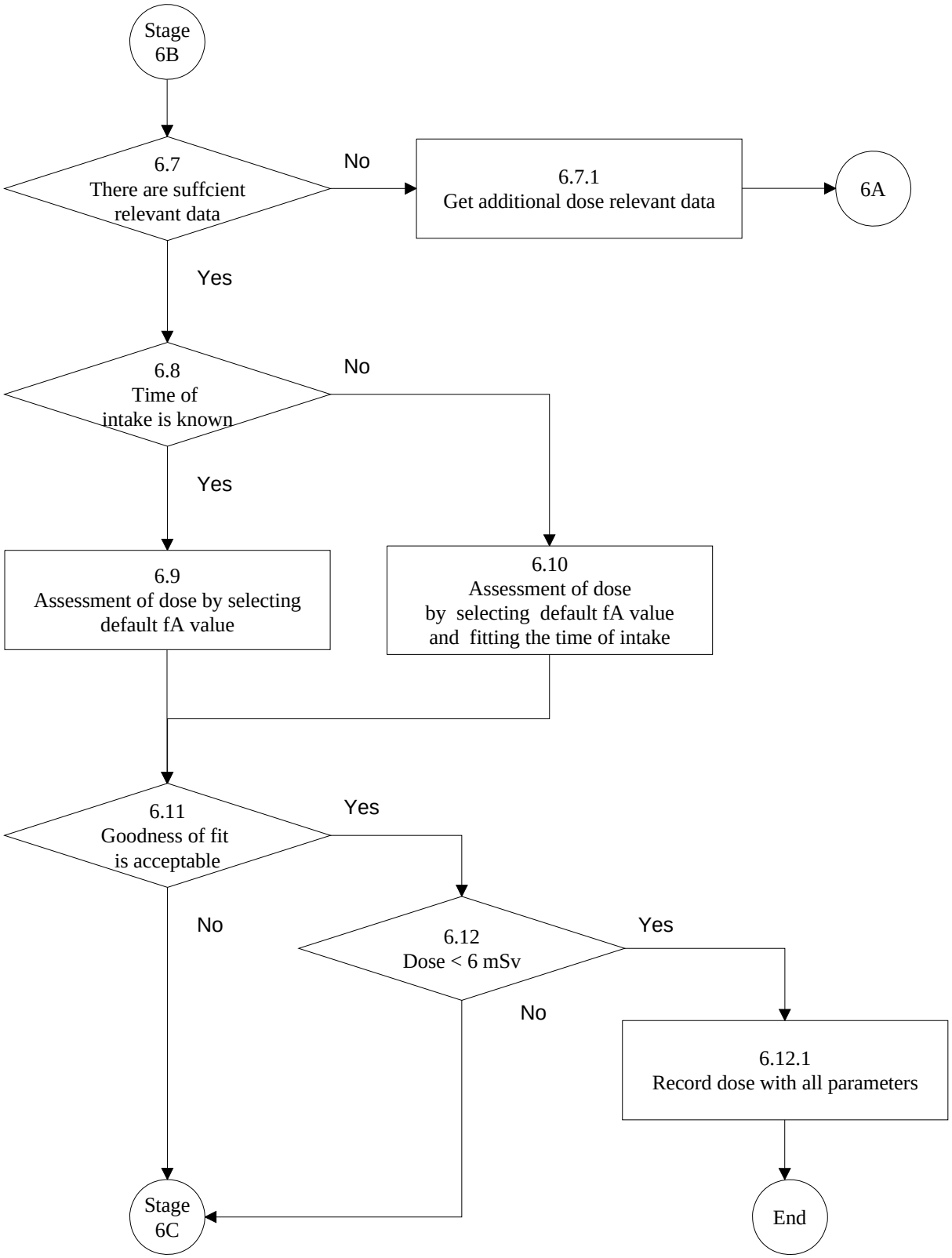


Fig. G5 Stage 6B – Structured approach to dose assessment

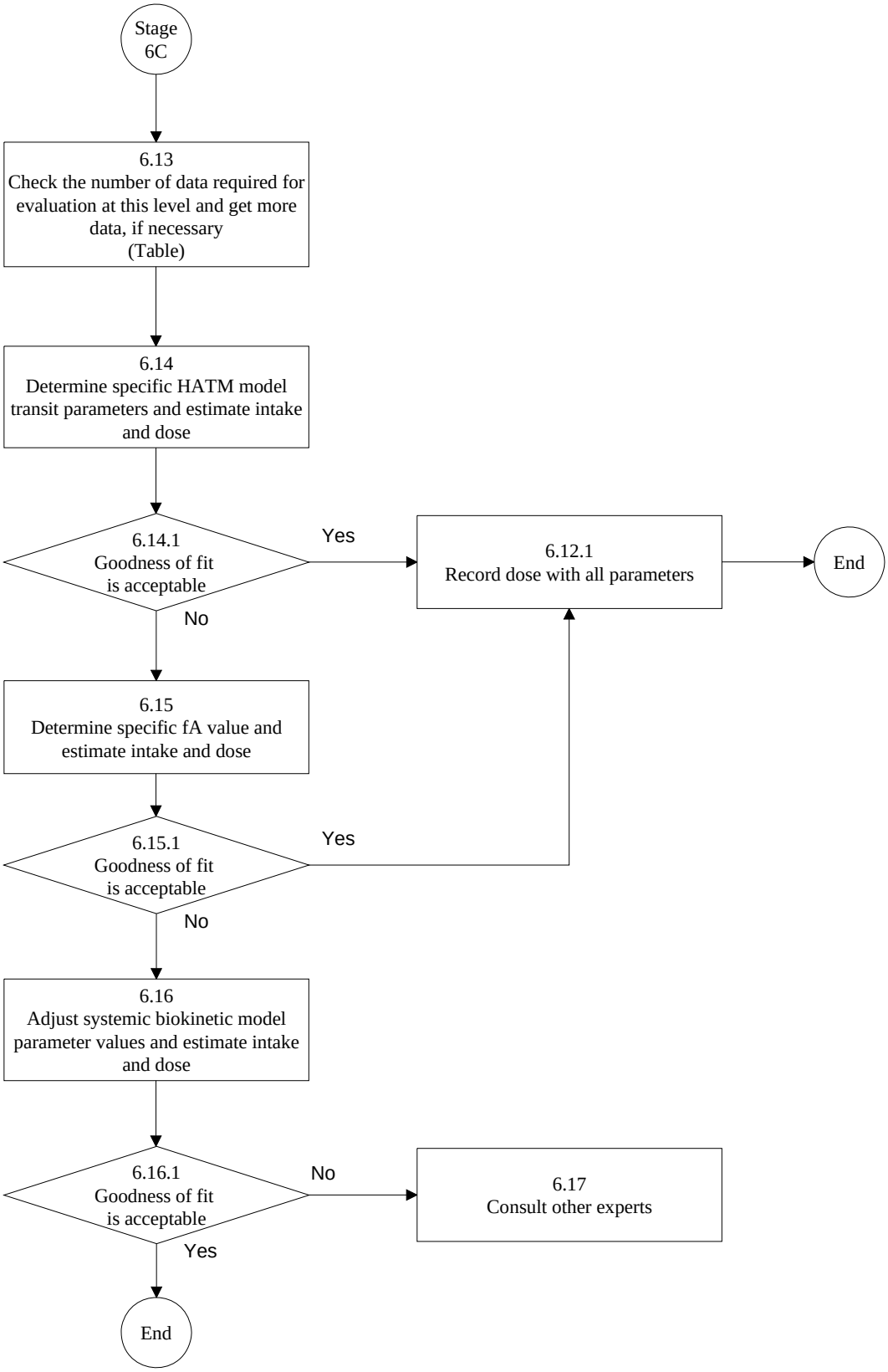


Fig. G6 Stage 6C – Structured approach to dose assessment

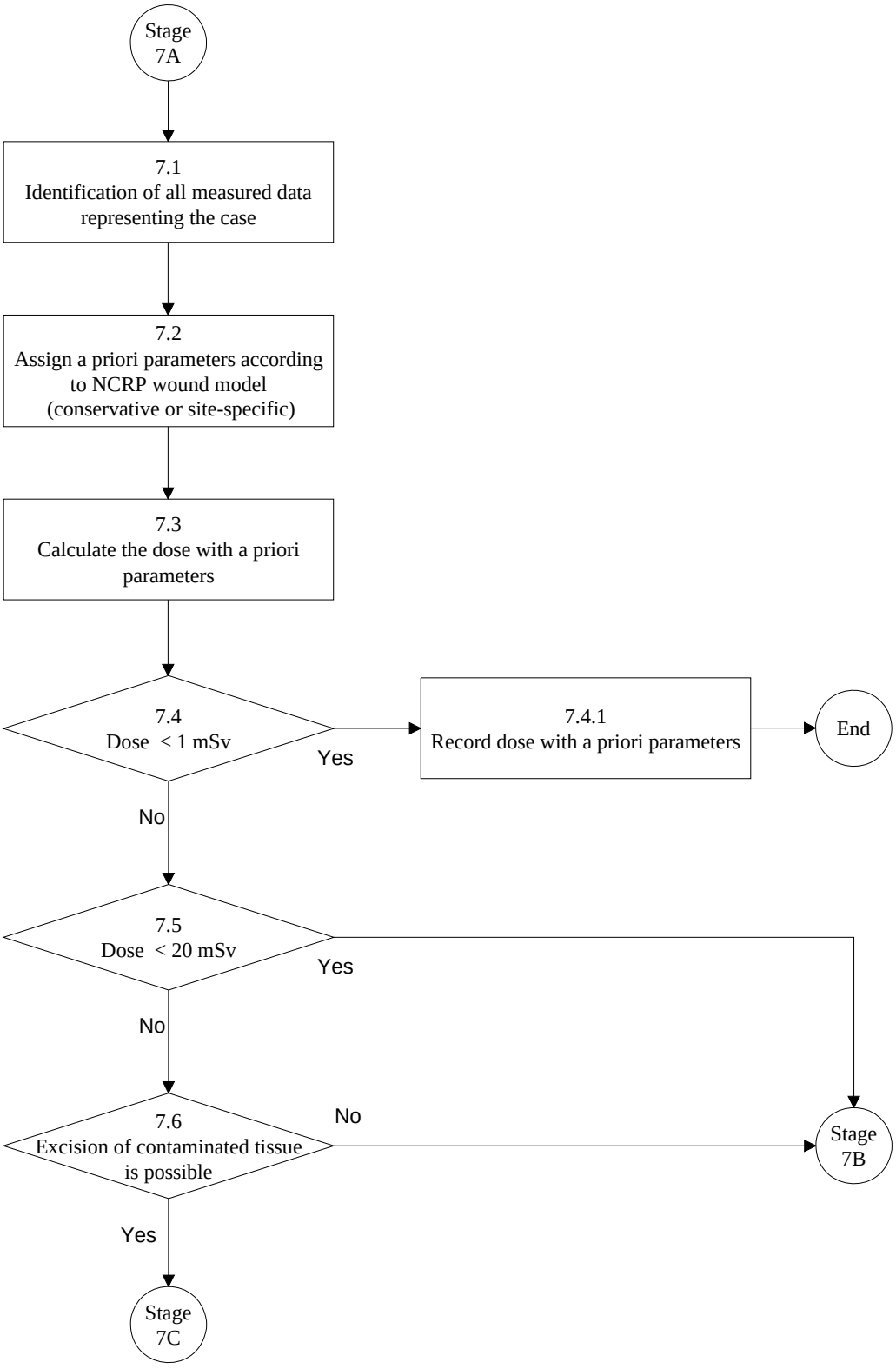


Fig. G7 Stage 7A – Structured approach to dose assessment

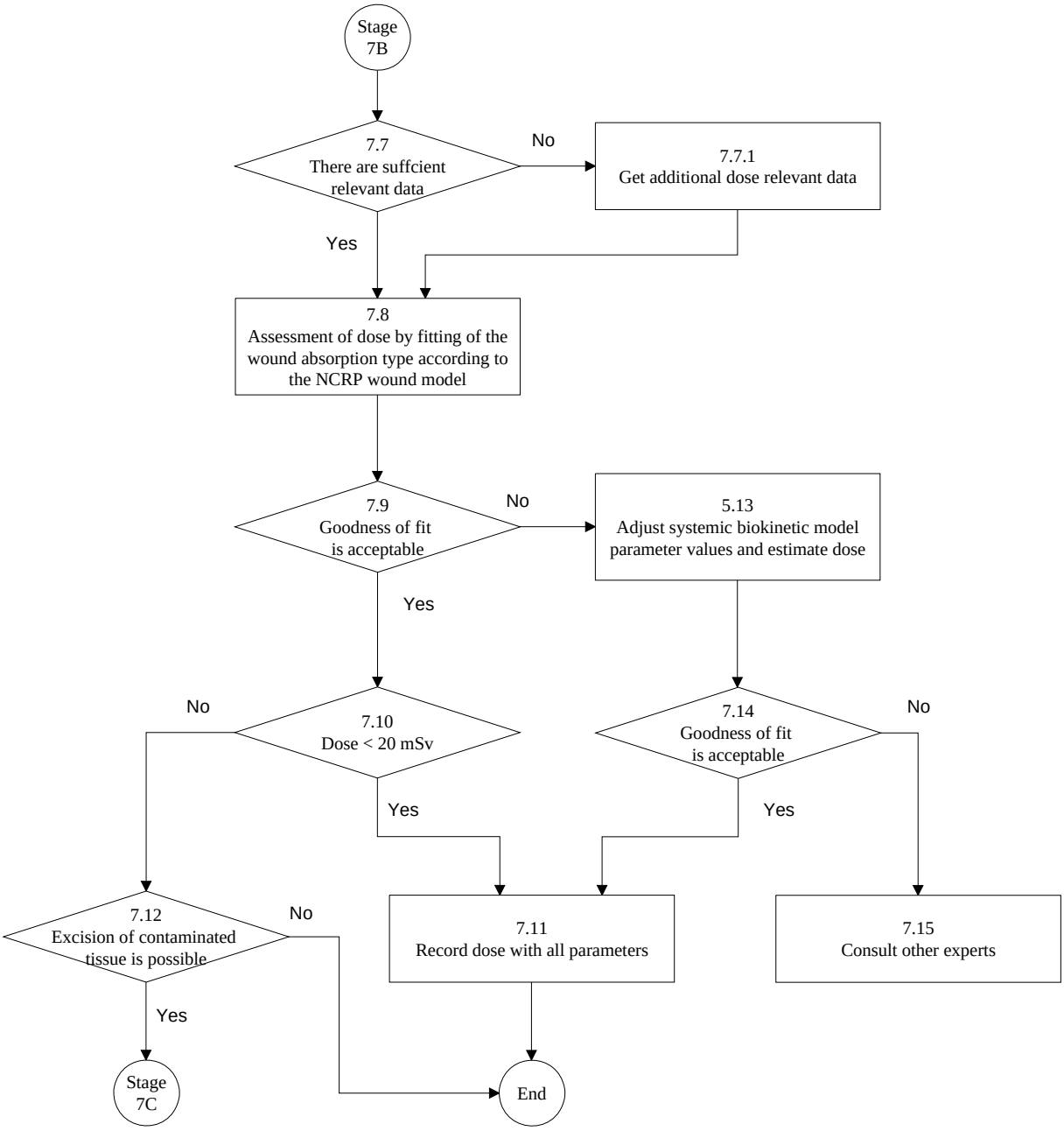


Fig. G8 Stage 7B – Structured approach to dose assessment

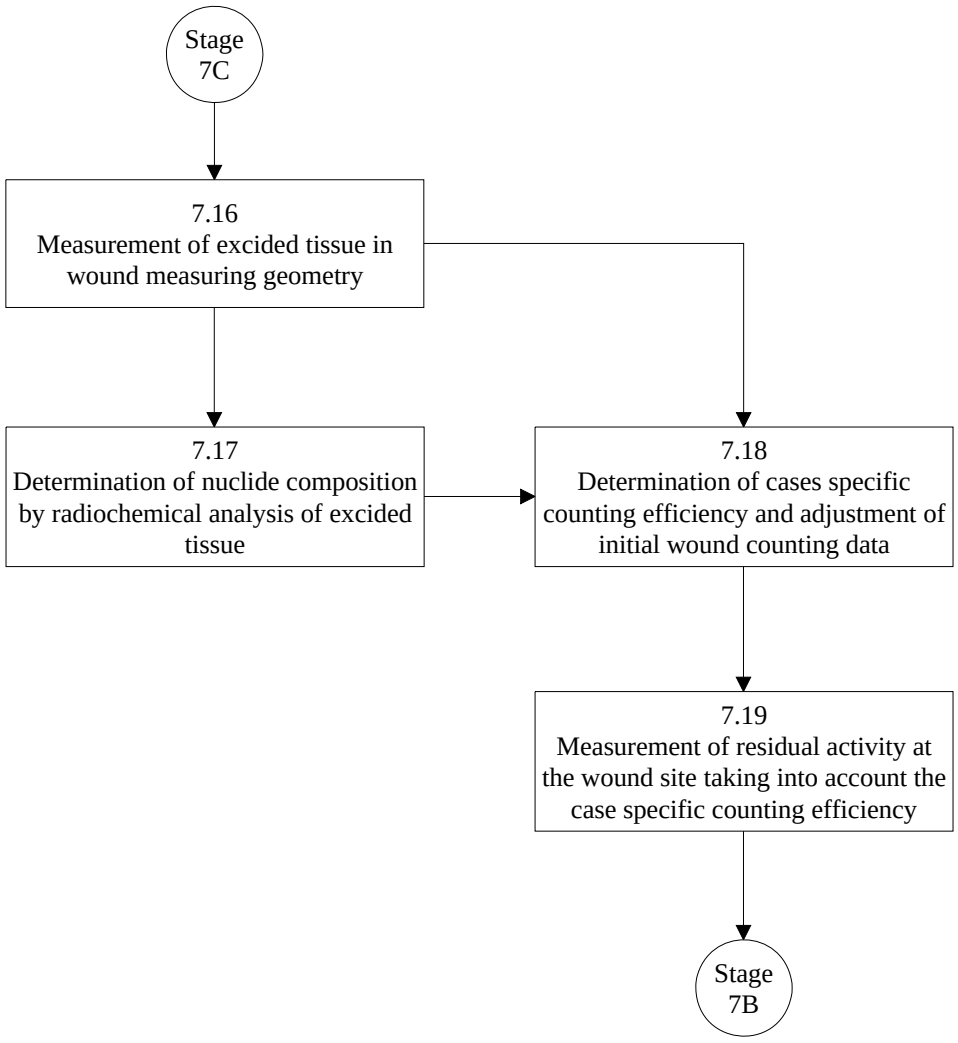


Fig. G9 Stage 7C – Structured approach to dose assessment

This Appendix on Uranium is included in the present draft version of the Supporting Guidance document on bioassay interpretation in order to illustrate the approach that will be used to provide data in the coming Occupational Intakes of Radionuclides (OIR) series of publications. The final Supporting Guidance document will not, however, contain specific data on radionuclides.

# 27 URANIUM (Z = 92)

## 27.1 Chemical forms in the workplace

Uranium occurs mainly in oxidation states IV and VI. Workers may be exposed to uranium in a variety of chemical and physical forms, including those discussed briefly in the following paragraphs. Some forms, notably the metal, carbides and oxides may be encountered as depleted uranium (~ 0.2% $^{235}$U), natural (0.7% $^{235}$U) or enriched (>0.7% $^{235}$U) uranium. The chemical behaviour of any given uranium compound will be similar irrespective of whether the metal is present in natural, depleted or enriched form.

Although this report is concerned with the calculation of doses it should be noted that intakes of the more rapidly absorbed uranium compounds are limited by considerations of chemical toxicity, rather than radiation dose (ICRP, 1997).

Table 27.1. The isotopes of uranium

| Isotope | Physical half-life |
|---|---|
| U-230 | 20.8 d |
| U-231 | 4.20 d |
| U-232 | 72 y |
| U-233 | 1.58E+05 y |
| U-234[a] | 2.44E+05 y |
| U-235[a] | 7.04E+08 y |
| U-236 | 2.34E+07 y |
| U-237 | 6.75 d |
| U-238[a] | 4.67E+09 y |
| U-239 | 0.392 h |
| U-240 | 14 h |

a Data for radionuclides given in publication. Remainder on accompanying CD-ROM.

## 27.2 Routes of Intake

### 27.2.1 Inhalation

There is extensive information available on the behaviour of uranium after deposition in the respiratory tract from animal experiments (mainly in rats), *in vitro* dissolution studies, and some accidental human intakes. Much of this information has been obtained since the issue of Publication 30 (ICRP, 1979). Absorption parameter values have been derived from the results of animal and *in vitro* studies for a wide range of compounds encountered in the nuclear fuel industry.

Proposed absorption parameter values and Types, and associated $f_A$ values for specific uranium compounds, derived from *in vivo* studies, are given in Table 27.2.

**Absorption Types and parameter values**

*Uranium hexafluoride* ($UF_6$). Uranium hexafluoride exists in vapour form, but in the presence of water in the atmosphere and in the respiratory tract it is converted to uranyl fluoride ($UO_2F_2$) aerosol. Generally, any exposure would be to both chemical forms simultaneously, and also to HF fumes. Hence, the mixture is treated here as an aerosol rather than a vapour. In experiments with beagle dogs (Morrow *et al*., 1982), 80% of the initial lung deposit (ILD) of uranium was absorbed into blood within 20 minutes. The rapid urinary excretion observed after accidental inhalation exposures by humans (Boback, 1975; Beau and Chalabreysse, 1989; Fisher *et al*., 1991) indicates assignment to default Type F. The rapid absorption half-time was estimated here to be 45 minutes ($s_r$ = 22 $d^{-1}$) from the data of Fisher *et al* (1991). Absorption parameter values derived here from urinary excretion data presented by Beau and Chalabreysse (1989) are $f_r$ = 1 and $s_r$ = 1.6 $d^{-1}$. Bailey and Davis (2002) derived absorption parameter values of $f_r$ = 1 and $s_r$ = 1.5 $d^{-1}$ from daily urinary excretion data presented by Moore and Kathren (1985) for an accidental intake by a worker (Case G) described by Boback (1975). However, the detailed data for the first two days after exposure reported by Boback (1975) show faster absorption ($s_r$ ~100 $d^{-1}$) of much of the uranium.

*Uranyl nitrate* ($UO_2(NO_3)_2$). Uranyl nitrate in aqueous solution is widely encountered in nuclear fuel fabrication and reprocessing. In rats exposed by inhalation to uranyl nitrate aerosols in aqueous solution (Ballou *et al*., 1986), 15–45% of the deposited material was retained in the lung at 30 d, depending on particle size, supporting assignment to default Type M. Measurements made after intratracheal instillation into rat lungs are consistent with assignment to default Type F (Cooper *et al*., 1982; Ellender, 1987; Stradling *et al*., 1991). Hodgson *et al*. (2000) derived absorption parameter values of $f_r$ = 0.93, $s_r$ = 3 $d^{-1}$ and $s_s$ = 5 $10^{-3}$ $d^{-1}$, consistent with assignment to default Type F.

*Uranyl Tri-Butyl-Phosphate (U-TBP).* Tri-n-Butyl-Phosphate (TBP) is used extensively as an extractant during fabrication of nuclear fuel and for the separation of uranium and plutonium during reprocessing. After administration of U-TBP to rats by intratracheal instillation, 80–90% of the U was absorbed into blood by about 1 d after exposure, and 70% of the amount absorbed by 7 d was excreted in the urine (Pellow *et al,* 1996). Absorption parameter values derived from results by Stradling *et al* (2002) were $f_r$ = 0.97, $s_r$ = 12 $d^{-1}$ and $s_s$ = 2.1 $10^{-3}$ $d^{-1}$, giving assignment to default Type F.

*Ammonium diuranate (ADU)* $(NH_4)_2U_2O_7$. ADU is a basic product in the uranium fuel cycle, a component of “yellow cake” (a generic term for material which may comprise ADU, $U_3O_8$ or a mixture of both). After inhalation of ADU by rats (Stradling *et al*, 1987), at 7 d, 11% ILD remained in the lung and 70% ILD was absorbed into blood. From the results Hodgson et al (2002) derived parameter values of $f_r$ = 0.85 and $s_r$ = 0.78 $d^{-1}$: the value of $s_r$ was too low to be determined and was taken to be 0.005 $d^{-1}$. Reviews of *in vivo* studies in rats (Ansoborlo *et al*, 2002; Stradling *et al*, 2002) gave a range of absorption parameter values: $f_r$ = 0.71–0.85, $s_r$ = 0.61–0.78 $d^{-1}$ and $s_s$ = 1.9 $10^{-2}$ $d^{-1}$ giving assignment to default Type M.

*Uranium peroxide hydrate* ($UO_4.nH_2O$). Uranium peroxide hydrate is present at one stage of the enriched uranium fuel cycle. This compound, also expressed as $UO_3.H_2O_2.H_2O$, is very similar to uranium trioxide $UO_3.nH_2O$. The dissolution and biokinetic behaviour of both compounds are very sensitive to the hydration state (*n* can vary between 0 and 2.5). One main characteristic of $UO_4.nH_2O$ is that it consists of small needles with an average AMAD of about 1.1 µm. Assessments of the physico-chemical and biokinetic properties of $UO_4$, both *in vitro* and *in vivo*, have been carried out (Ansoborlo *et al*, 1998a). The biokinetics of uranium were followed for 90 days after intratracheal administration to rats. By 7 d after exposure 3–10% of uranium remained in the lungs, whereas about 65% was absorbed into blood. The calculated absorption parameter values were: $f_r$ = 0.87, $s_r$ = 0.93 $d^{-1}$ and $s_s$ = 2.4 $10^{-2}$ $d^{-1}$ (Ansoborlo *et al,* 1998a*)* giving assignment to default Type F.

*Uranium tetrafluoride* ($UF_4$). Uranium tetrafluoride is an intermediate product in the uranium fuel cycle. It can be reduced to uranium metal or oxidized by fluorine to form $UF_6$. The reported biokinetic behaviour of $UF_4$ is complex. Measurement of urinary excretion after inhalation by workers (Chalabreysse *et al*., 1989) and experiments in rats and baboons (Stradling *et al*., 1985; André *et al*, 1989; Ansoborlo *et al*., 1990) show that a large fraction (35–40%) of the lung deposit was rapidly absorbed to the blood by 7 d. However, considerable variations in behaviour were observed, with some experiments indicating assignment to default Type F and others to default Type M. Reviews of *in vivo* studies (Chazel *et al*, 2000a; Ansoborlo *et al*, 2002, Stradling *et al*, 2002) on different industrial forms of $UF_4$ gave a range of absorption parameter values $f_r$ = 0.51 – 0.58, $s_r$ = 0.11 – 0.21 $d^{-1}$ and $s_s$ = 2.6 $10^{-3}$ – 7.4 $10^{-3}$ $d^{-1}$, giving assignment to default Type M.

*Uranyl Acetylacetonate.* Uranyl acetylacetonate is an organic complex of uranium with military applications. *In vitro* dissolution tests in simulated lung fluid led to the classification of 50% Class D and 50% Class W (Fisher and Briant, 1994). Absorption parameter values calculated here are $f_r$ = 0.52, $s_r$ = 2.5 $d^{-1}$ and $s_s$ = 2.6 $10^{-2}$ $d^{-1}$, giving assignment to default Type M.

*Uranium trioxide* ($UO_3.nH_2O$). Uranium trioxide is formed by heating uranyl nitrate, which in the fuel fabrication cycle is then reduced to form $UO_2$. The biokinetic behaviour of $UO_3$ is very sensitive to the hydration state and its solubility depends on the parameter *n*. Studies carried out on rats (Morrow *et al*, 1972; Ansoborlo *et al*, 2002, Stradling *et al*, 2002) gave ranges of absorption parameter values: $f_r$ = 0.71–0.92, $s_r$ = 0.28–14 $d^{-1}$ and $s_s$ = 3.6 $10^{-3}$ – 2 $10^{-2}$ $d^{-1}$, giving assignment to default Type F in some cases and Type M in others.

*Uranium octoxide* ($U_3O_8$). Uranium octoxide can be present in the ore concentrate ("yellow cake", see ADU above) and also occurs at later stages in the uranium fuel cycle. Human data from accidental intakes of $U_3O_8$ (West *et al*., 1979; Eidson, 1990), and from monitoring data for workers in processing facilities (Barber and Forrest, 1995; Chalabreysse *et al*, 1989), the many animal studies in rats, dogs and monkeys (Métivier *et al*, 1992; Stradling *et al*, 1989a), and extensive *in vitro* studies (Eidson, 1994; Ansoborlo *et al*, 1998; Chazel *et al*, 1998) show that the biokinetic behaviour of this compound depends on the particular process of manufacture. A study of the influence of specific surface area (SSA) (Chazel *et al*, 1998) also demonstrated the important effect of this parameter on the dissolution behaviour. When the SSA increased from 0.7 to 16 $m^2$ $g^{-1}$ the rapidly dissolved fraction, $f_r$, increased from 0.01 to 0.20. At 30 d after intake by rats and baboons, the lung retention and total urinary excretion were 50-90% ILD and 2–10% ILD, respectively. An analysis of the results obtained on different forms of $U_3O_8$ collected in the French or UK industry (Ansoborlo *et al*, 2002; Stradling *et al*, 2002) gave ranges of absorption parameter values: $f_r$ = 0.03–0.04, $s_r$ = 0.49–2.1 $d^{-1}$ and $s_s$ = 3.5 $10^{-4}$ – 3.8 $10^{-4}$ $d^{-1}$, giving assignment to default Type M.

*Uranium dioxide* ($UO_2$). Uranium dioxide is the final product in the manufacture of nuclear fuel pellets, and is also present as depleted uranium in mixed oxide fuel (MOX). Human studies have shown that $UO_2$ can be very insoluble (Pomroy and Noel, 1981; Price, 1989; Schieferdecker *et al*., 1985). Experiments in rats, dogs, monkeys and baboons (Leach *et al*., 1973; Stradling *et al*., 1989b; Métivier *et al*., 1992) also support the assignment of $UO_2$ to default Type S. Manufacturing processes of $UO_2$ differ from one industry to another and, as for $U_3O_8$, SSA plays a role in the dissolution of $UO_2$ (Chazel *et al*, 2000b). For compounds with SSA varying from 1.0 to 4.4 $m^2$ $g^{-1}$, $f_r$ varied from 0.003 to 0.004. At 30 d after intake by rats and baboons, the total urinary excretion was 1–4% ILD and lung retention was 60-90% ILD. An analysis of results obtained on different forms of $UO_2$ collected in the French and UK industry (Ansoborlo *et al*, 2002; Stradling et al, 2002) gave average absorption parameters $f_r$ = 0.01–0.02, $s_r$ = 0.9–1.3 $d^{-1}$ and $s_s$ = 2.6 $10^{-4}$–8.6 $10^{-4}$ $d^{-1}$, giving assignment to default Type S.

*Vaporised uranium metal.* A new method for uranium enrichment, based on laser isotopic separation, can produce three different types of aerosol identified as variable mixtures of $U_{metal}$ + $UO_2$ + $U_3O_8$, with different particle size distributions. *In vivo* instillation in rats gave absorption parameter values in the following ranges: $f_r$ = 0.12–0.36, $s_r$ = 0.47–1.45 $d^{-1}$ and $s_s$ = 0.9–7.3 $10^{-3}$ $d^{-1}$ (Ansoborlo *et al*, 1998b), giving assignment to default Type M.

*Depleted uranium (DU).* Depleted uranium, a by-product of the manufacture of enriched uranium for nuclear reactor fuel, has found a number of applications resulting mainly from its high density, in particular, in anti-tank munitions, counterweights for aircraft control surfaces and radiation shielding. DU, typically alloyed with 0.75% titanium is used in 'kinetic energy penetrators', rods of the metal fired at very high speed (~1.5 km $s^{-1}$). On impact with a hard object such as armour plate, a significant fraction of the penetrator mass may be converted to an aerosol that could be inhaled by persons in the vicinity or downwind. *In vitro* tests have shown considerable variability in the that 1–50% of the respirable material dissolves rapidly, and the rest very slowly, while X-ray analyses indicate that the uranium is present as a mixture of oxides including $U_3O_7$, $U_3O_8$, $U_4O_9$, and $UO_2$, but also combinations with other metals (Glissmeyer and Mishima, 1979; Scripsick *et al.*,1985a; 1985b; Chazel *et al,* 2003, Mitchel and Sunder, 2004). *In vitro* dissolution tests carried out by Chazel *et al* (2003), gave dissolution parameter values in the following ranges: $f_r$ = 0.47–0.57, $s_r$ = 0.06 – 0.07 $d^{-1}$ and $s_s$ = 1.8–3.4 $10^{-4}$ $d^{-1}$, giving assignment to Type M.

In by far the most comprehensive study so far, the Capstone DU Aerosol Study, in which aerosols formed when DU rounds penetrated armoured vehicles, Parkhurst et al (2004a, 2004b) measured dissolution in simulated lung fluid for 46 days on 27 samples, mainly from cascade cyclone stages and their back-up filters. Retention of undissolved DU was fit by two- or three-component exponential functions. Based on the two-component fits, there was a rapidly dissolving fraction of 1-28% (geometric mean, GM, 12.5%), with an associated rapid dissolution rate of 0.1-30 $d^{-1}$ (GM 6 $d^{-1}$; corresponding half-time, $t_{1/2}$ = 0.12 d). The remaining fraction dissolved at a slow rate of 0.0004-0.0095 $d^{-1}$ (GM 0.0026 $d^{-1}$: $t_{1/2}$ = 268 d). Thus there was considerable variation between samples, especially in the fraction that dissolved rapidly. There appeared to be some correlation between the initial and final dissolution rates: the greater the dissolution in the first day, the faster the long term dissolution rate. Based on extrapolation of the three-component exponential function where available (two-component otherwise), 24 samples would be assigned to Type M and three to Type S. Several sets of measurements were made on different stages from the same cascade cyclone. However, there was no clear trend of dissolution with particle size and in some cases the back-up filter, with the smallest particles, showed the slowest dissolution. Two confounding factors were noted: (1) cyclone cut-offs are not sharp, so there was considerable overlap in size distribution between stages (2) scanning electron microscope examination showed great heterogeneity of particle composition, shape etc.

Mitchel and Sunder (2004) followed urinary excretion of uranium for 7 days after intratracheal instillation into rats of the <50-µm fraction of dust obtained from impact of DU munitions on armour plate. Results indicate that about 10% ILD dissolved during 7 days, about half of it within 1 day. However, the large size suggests that the material was from surface deposits rather than air samples, and may not be representative of dust that might be inhaled.

If large pieces of uranium metal are subjected to fire (*e.g.* in a burning vehicle or aircraft crash) they will gradually oxidise and some of the oxide may be dispersed and inhaled. *In vitro* tests have shown that 0.5–10% of the respirable material dissolves rapidly, and the rest very slowly, while X-ray analyses indicate that most of the uranium is present as $U_3O_8$ (Mishima et al., 1985; Elder and Tinkle, 1980; Scripsick *et al.*, 1985a; OSAGWI,

2000).

*Irradiated fuel fragments.* Following an accidental release from a nuclear reactor, fission and activation products may be present in fragments of irradiated fuel, of which the matrix is predominantly uranium dioxide (Devell, 1988; Begichev *et al*., 1989; Toivonen *et al*., 1992). In studies of the *in vitro* dissolution of particles released from the Chernobyl accident, seven out of ten of which consisted mainly of uranium (Cuddihy *et al*., 1989), the data obtained were consistent with assignment of all the $\gamma$-emitting radionuclides to Type M.

**Rapid dissolution rate for uranium**

Most studies on uranium compounds show that the rapid rate of absorption is much less than 100 $d^{-1}$. In the absence of specific data, $s_r$ is taken to be 10 $d^{-1}$ for Type F compounds and 1 $d^{-1}$ for Type M and S compounds.

**Extent of binding of uranium to the respiratory tract**

Experimental evidence suggests that there is little binding of uranium to the respiratory tract. Cooper *et al*. (1982) and Ellender (1987) followed the behaviour of $^{233}U$ after instillation of uranyl nitrate and bicarbonate into the pulmonary region of the lungs of rats. Cooper *et al*. (1982) found that less than 2% of the initial lung deposit (ILD) remained at 7 days. Ellender (1987) gave more information for the nitrate, for which about 8% ILD remained at 1 d and 3% at 30 d. Detailed analysis, however, indicates that clearance over this period was mainly by particle transport, and that the results did not provide evidence for binding of uranium (Hodgson, *et al.,* 2000). It is therefore assumed that for uranium the bound state can be neglected, *i.e*., $f_b = 0.0$.

### 27.2.2 Ingestion

Data on the absorption of U have been reviewed by Wrenn et al. (1985), Harrison (1991), Leggett and Harrison (1995) and in ICRP Publication 69 (1994).

In the first controlled human study involving more than one subject, Hursh et al. (1969) administered uranyl nitrate to 4 hospital patients. The data obtained were taken to suggest fractional absorption in the range 0.005 - 0.05 (0.5 - 5%). Leggett and Harrison (1995) have interpreted the data as suggesting absorption of 0.004, 0.01, 0.02 and 0.06 for the 4 subjects. Wrenn *et al*., (1989) estimated absorption in 12 normal healthy adult volunteers given drinking water high in U. On the basis that 40 - 60% of absorbed U was excreted in the urine in the first three days, rather than the authors assumption of 79%, Leggett and Harrison (1995) concluded that mean absorption was 0.01 - 0.015, maximum absorption was in the range 0.02 - 0.04, and that six subjects absorbed less than $2.5 \times 10^{-3}$. Harduin et al., (1993) reported results for the absorption of U from drinking water either administered on one day or over 15 days. The data for acute administration suggested absorption of 0.005 - 0.05 with an average value of 0.015 - 0.02. The data for 15-day administration suggested absorption of 0.003 - 0.02 and average absorption of 0.01 - 0.015. Similar results have also been obtained in a number of dietary balance studies (Larsen and Orlandini, 1984; Spencer et al. 1990; Wrenn et al. 1989; Leggett and Harrison, 1995).

Data from animal studies provide information on the relative uptake of U ingested in different chemical forms, showing that absorption is strongly dependent on the solubility of the compound. Measurements have been made in rats, hamsters, rabbits, dogs and baboons (reviewed by Wrenn *et al*., 1985; Harrison, 1991; Leggett and Harrison, 1995). Absorption appears to be greatest for U ingested as $UO_2(NO_3)_2.6H_2O$, U-TBP, $UO_2F_2$ or $Na_2U_2O_7$, roughly half as great for $UO_4$ or $UO_3$, and 1 - 2 orders of magnitude lower for $UCl_4$, $U_3O_8$, $UO_2$ and $UF_4$. It should be noted, however, that the solubility of some poorly

soluble U compounds can vary substantially with thermal history as well as particle size (Cooke and Holt, 1974). Thus, greater absorption as $UO_2$ in hamsters than rats and dogs, could reflect solubility of the preparation of $UO_2$ rather than just species differences. A number of studies have shown that absorption is substantially greater in fasted than fed animals. For example, Bhattacharyya *et al*., (1989) found that uptake was increased by an order of magnitude in mice and baboons deprived of food for 24 h prior to U administration. Sullivan *et al*. (1980) reported a 2 – 4 fold increase in U absorption in rats given U nitrate after a 24 hour fast.

In Publication 30 (ICRP, 1979), an $f_1$ of 0.05 was recommended for water soluble inorganic forms of U(VI) and a value of 0.002 for U(IV) in relatively insoluble compounds such as $UF_4$, $UO_2$ and $U_3O_8$. In Publication 69 (ICRP, 1994), an $f_1$ of 0.02 was adopted for dietary intakes of U on the basis of human data as reviewed by Wrenn *et al*., (1985), Harrison (1991) and Leggett and Harrison (1995). The available human and animal data indicate that a value of 0.02 is also appropriate for occupational exposures to more soluble inorganic forms, including $UO_2(NO_3)_2.6H_2O$, $UO_2F_2$ and $Na_2U_2O_7$.

**Values of $f_A$ adopted for ingestion**

In this report, an $f_A$ value of 0.002 is adopted for the fractional absorption of relatively insoluble compounds (e.g. $UF_4$, $UO_2$, $U_3O_8$) and an $f_A$ value of 0.02 is adopted for all other more soluble chemical forms (Table 27.2).

### 27.2.3 Skin and wounds

Few quantitative data on the absorption of uranium metal or uranium compounds from human wounds are available. In subjects with wounds containing unknown amounts of embedded DU metal, the urinary excretion 3-5 years after injury was reported to be 150-210 times that in non-exposed persons (Hooper, *et al*., 1999). Since the amounts of uranium in the bodies of these persons are not known it is not possible to calculate dissolution rates for the embedded uranium metal.

Toohey (2003) reported data for the urinary excretion of uranium by seven persons who in 1997 had between 130 and 840 mg DU shrapnel embedded in their bodies. The individual excretion rates appeared to be very variable; the mean urinary fractional excretion rate measured in these seven individuals in 1997 was $(2.4 \pm 2.8) \times 10^{-5}$ $g^{-1}$ creatinine; in 1999 the mean fractional excretion in five of these subjects was $(1.1 \pm 0.6) \times 10^{-5}$ $g^{-1}$ creatinine. Toohey (2003) concluded that the mean fractional excretion values were not significantly different from each other and that there was no correlation between the excretion rate and the mass of the uranium measured in the bodies of these individuals.

Pellmar et al., 1999 studied the migration of uranium from DU pellets implanted in rats. Leggett and Pellmar (2003) reviewed these rat data and concluded that the results provided limited support for the application of the ICRP (ICRP, 1995) uranium systemic model to persons with embedded DU pellets. Hahn et al. (2000) studied the urinary excretion of uranium following the implantation of DU into the hind limbs of rats and calculated that about $10^{-6}$ of the implanted mass was excreted per day.

Animal studies have shown that uranium compounds can be absorbed through the skin in sufficient amounts to cause severe poisoning, and even death (Orcutt, 1949, Yuile, 1973, López *et al*., 2000). However, these studies do not provide quantitative data on the biokinetics, or the extent of absorption. Briefly, in animals, soluble compounds, including uranyl nitrate, uranyl fluoride, uranium pentachloride, uranium trioxide, sodium diuranate and ammonium diuranate, are sufficiently well absorbed to cause severe toxicity. In contrast, the insoluble oxides $UO_2$, $UO_4$ and $U_3O_8$, as well as the tetrafluoride $UF_4$, do not appear to be absorbed to a sufficient extent to cause symptoms of toxicity

(Yuile, 1973). In rats following dermal exposure to uranyl nitrate, significant quantities of uranium were demonstrated in the blood stream (Yuile, 1973). The tetravalent tetrachloride $UCl_4$ also caused poisoning, but it was believed that oxidation to $UCl_6$ was necessary for absorption to occur (Yuile, 1973).

Table 27.2. Absorption parameter values for inhaled and ingested uranium

| Inhaled particulate materials | Absorption parameter values[a] | | | Absorption Type[b] | Absorption from the alimentary tract, $f_A$[c] |
|---|---|---|---|---|---|
| | $f_r$ | $s_r$ ($d^{-1}$) | $s_s$ ($d^{-1}$) | | |
| Specific parameter values[d] | | | | | |
| Uranyl nitrate, $UO_2(NO_3)_2$ | 0.9 | 3 | 0.005 | (F) | 0.02 |
| Uranyl Tri-Butyl-Phosphate (U-TBP) | 0.97 | 12 | 0.002 | (F) | 0.02 |
| Uranium peroxide hydrate $UO_4$ | 0.9 | 0.9 | 0.024 | (F) | 0.02 |
| Ammonium diuranate, ADU | 0.8 | 0.7 | 0.020 | (M) | 0.002 |
| Uranium tetrafluoride $UF_4$ | 0.6 | 0.15 | 0.005 | (M) | 0.002 |
| Uranium trioxide $UO_3$ | 0.8 | 1 | 0.01 | (M) | 0.002 |
| Triuranium octoxide $U_3O_8$ | 0.04 | 1 | 0.004 | (M) | 0.002 |
| Uranium dioxide $UO_2$ | 0.015 | 1 | 0.0005 | (S) | 0.002 |
| Default parameter values[e,f] | | | | | |
| Uranium hexafluoride, $UF_6$ | 1 | 10 | - | F | 0.02 |
| Uranyl acetylacetonate; DU aerosols from use of kinetic energy penetrators; all unspecified forms | 0.1 | 1 | 0.005 | M | 0.002 |
| | 0.001 | 1 | 0.0001 | S | 0.002 |
| Ingested materials | | | | | |
| Soluble forms (Type F) | – | – | – | - | 0.02 |
| Relatively insoluble forms (as assigned to Types M and S for inhalation) | – | – | – | – | 0.002 |

a It is assumed that for uranium the bound state can be neglected, *i.e.*, $f_b = 0.0$.

b For those materials for which specific absorption parameter values are given, the corresponding default absorption Type is given in parentheses. This is used to assign $f_A$ where a specific value is not available from experimental data.

c $f_A$ values for ingested forms are also applied to material entering the alimentary tract after inhalation, using 0.02 for more soluble forms (Types F and M) and 0.002 for less soluble forms (Type S).

d See text for summary of information on which parameter values are based, and on ranges of parameter values observed for individual materials

e Materials (e.g. $UF_6$) are listed here where there is sufficient information to assign to a default absorption Type, but not to give specific parameter values (see text).

f Default Type M is recommended for use in the absence of specific information.

## 27.3 Systemic Distribution, Retention and Excretion

### 27.3.1 Systemic retention

In experimental studies in humans, typically two-thirds of U injected intravenously as the nitrate was excreted in urine in the first 24 hours and about a further 10% over the following 5 days (ICRP, 1995). Human and animal data indicate that most of the remaining U is excreted over a period of a few months but a small proportion is retained for a period of years (Stevens *et al.*, 1980; Sontag, 1984; ICRP, 1995). Faecal excretion of U is low, accounting for less than 1% of total excretion in human subjects over the first few days after intravenous injection of uranyl nitrate (Bernard and Struxness, 1957).

Results from animal experiments indicate that by a few days after absorption to blood, the majority of retained U is confined to the kidneys and skeleton. For example, the kidneys and skeleton of beagles accounted for 90% of retained U at 2 – 6 days after injection (Morrow *et al*., 1982). Measurements of the systemic distribution of U at autopsy after injection of uranyl nitrate showed that the skeleton, kidneys and other soft tissues accounted for 10%, 14% and 6%, respectively, of injected activity in a subject dying after 2.5 days, 4-13%, 6% and 4%, respectively, after 18 days and 1.4%, 0.3% and 0.3%, respectively, after 566 days.

A substantial proportion of U filtered by the kidneys is temporarily retained in the renal tubules before being excreted in urine. Durbin (1984) reviewed human and animal data and concluded that 92- 95% of the renal content is lost rapidly but that the remainder has a half-time of 30 – 340 days. Retention of U by the kidneys appears to increase with the mass absorbed to blood, complicating the interpretation of experimental studies, many of which involved high masses.

The behaviour of U in the skeleton shows some qualitative similarities to that of the alkaline earth elements, even if the chemical analogy, using affinity constants for mineral ligands, between $UO_2^{2+}$ and $Ca^{2+}$ is not very well close (Ansoborlo et al, 2006). It has been shown that uranyl ions exchange with $Ca^{2+}$ on the surface of bone mineral crystals although they do not participate in crystal formation or enter existing crystals. The early distribution of U in the skeleton is similar to that of Ca. There is evidence from studies using dogs that U enters bone mineral by diffusion as well as burial (Stevens *et al*., 1980). As is the case for Ca, a substantial proportion of U deposited in bone is lost to the circulation by processes that occur more rapidly than bone resorption.

Because of the qualitative similarities in the biokinetic behaviour of U and the alkaline earth elements, the model for U adopted in *ICRP Publication 69* (1995) was based on the model for the alkaline earth elements in *ICRP Publication 67* (1993). This model is also adopted in this report. Some features of the model (Figure 27.1) were included to enable its extension to U and do not apply to the alkaline earth elements. Thus, retention in red blood cells (RBC), kidneys and liver are included. The transfer rates used for adults (Table 27.3) apply to age $\geq$ 25 y when skeletal maturity has been reached and adult bone formation rates apply (see below).

Data on the early behaviour of uranium in the human circulation can be represented using an overall removal rate of 35 $d^{-1}$ (one half the transfer rate used for radium; *ICRP* (1993)) and assuming rapid exchange with a soft tissue compartment, ST0, receiving 30% of U leaving plasma; the assumed removal half-time from ST0 to plasma is 2 h. Based on data for baboons (Lipzstein, 1981), it is assumed that 0.7% of U leaving plasma is taken up by circulating red blood cells; the removal half-time from red cells to plasma is assumed to be 2 d.
From blood plasma, 15% of U is taken to deposit on bone surfaces. By analogy with the alkaline earth elements (ICRP, 1993), the ratio of the proportion of uranium deposited on trabecular surfaces to that deposited on cortical surfaces is assumed to be 1.25 for the mature skeleton. The removal half-time from bone surfaces is assumed to be 5 d. It is assumed that one-half of the uranium leaving bone surfaces returns to the plasma and one-half goes to an exchangeable bone volume compartment. Uranium is assumed to leave the bone volume compartment with a half-time of 30 d; this value was derived for radium (ICRP, 1993) and is consistent with the limited data on loss of uranium from bone. Based on the small amount of uranium retained in the skeleton over a long term (as indicated by human chronic exposure data) 75% of uranium leaving exchangeable bone is assumed to return to the bone surface compartment and 25% is assigned to non-exchangeable bone. Removal from non-exchangeable bone volume is assumed to result from the relatively slow process of bone turnover, involving resorption and remodelling.

For consistency with data for occupationally exposed persons, the liver is treated as two

compartments. Uptake accounts for 1.5% of U leaving blood, and retention half-times are 7 d for 93% and 10 y for 7%. To fit the available data for retention in soft tissues other than liver and kidneys, two other compartments are used: ST2 is assumed to receive 0.3% of U leaving blood and has a retention half-time of 100 years and ST1 receives 6.65% and has a half-time of 20 days.

Urinary excretion is assumed to arise from (1) direct transfer to the urinary bladder, accounting for 63 % of U leaving blood, and (2) delayed transfer after retention in the renal tubules, accounting for 12% of U leaving blood. The half-time of retention in the tubules is assumed to be 7 d and retention in other kidney tissues is set to correspond with the available human data. Information on uranium excretion is given in Ting *et al* (1999); Ough *et al* (2002); and Roth *et al* (2003).

*In vivo* experiments describing the distribution of uranium in rats implanted with 4 to 20 depleted uranium pellets, showed significant concentration of uranium in the skull and in the brain for high dose rate, at 1 to 6 months after implantation (Pellmar *et al* 1999): at 6 months the concentration in the skull was 1250 ng U $g^{-1}$ tissue compared to 1800 ngU $g^{-1}$ tissue in tibia, and the concentration in brain was 30 ng U $g^{-1}$ tissue compared to 150 ngU $g^{-1}$ tissue in liver or 80 ngU $g^{-1}$ tissue in spleen.

### 27.3.2 Gender-related differences in biokinetics

There are insufficient data, either human or animal, to identify any clear gender related differences in organ retention functions or longer-term excretion.

### 27.3.3 Behaviour of decay products

The treatment of decay-products is similar to that described in this document for thorium. For the treatment of thorium, actinium or protactinium produced from uranium *in vivo*, compartments for testes, ovaries, trabecular bone marrow and cortical bone marrow are added to the model structure shown in Fig. 27.1. Thorium or radium produced in red blood cells is assumed to be transferred to blood plasma with a half-time of 1 d.

Table 27.3. Transfer rates for the uranium systemic model shown in Figure 27.1. The values are for adult workers

| | $d^{-1}$ |
|---|---|
| Plasma to ST0 | 1.05E+01 |
| Plasma to RBC | 2.45E–01 |
| Plasma to urinary bladder | 1.543E+01 |
| Plasma to urinary path | 2.94E+00 |
| Plasma to other kidney tissue | 1.22E–02 |
| Plasma to ULI contents | 1.22E–01 |
| Plasma to liver 1 | 3.67E–01 |
| Plasma to ST1 | 1.63E+00 |
| Plasma to ST2 | 7.35E–02 |
| Plasma to trabecular surfaces | 2.04E+00 |
| Plasma to cortical surfaces | 1.63E+00 |
| ST0 to plasma | 8.32E+00 |
| RBC to plasma | 3.47E–01 |
| Urinary path to urinary bladder | 9.90E–02 |
| Other kidney tissue to plasma | 3.80E–04 |
| Liver 1 to plasma | 9.20E–02 |
| Liver 1 to liver 2 | 6.93E–03 |
| ST1 to plasma | 3.47E–02 |
| ST2 to plasma | 1.90E–05 |
| Bone surfaces to plasma | 6.93E–02 |
| Bone surfaces to exchangeable volume | 6.93E–02 |
| Liver 2 to plasma | 1.90E–04 |
| Non-exchangeable trabecular volume to plasma | 4.93E–04 |
| Non-exchangeable cortical volume to plasma | 8.21E–05 |
| Exchangeable bone volume to bone surfaces | 1.73E–02 |
| Exchangeable bone volume to non-exchangeable volume | 5.78E–03 |

Parameter values are given with sufficient precision for computational purposes. This may be more precise than the biological data would support.

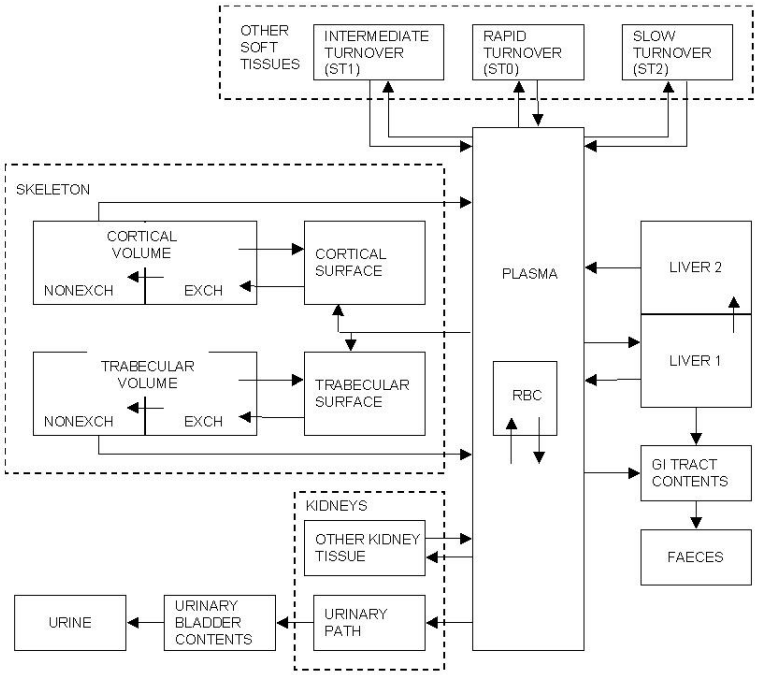


Fig. 27.1. ICRP systemic model for uranium, which also applies to the alkaline earth elements, and to lead.

## 27.4 Dose Coefficients

For inhaled materials, studies of common chemical forms showing characteristics of Types F, M and S have been found in the literature. Type M is recommended as the default in the absence of specific information permitting use of specific parameter values, or assignment to Type F, M or S. The dose coefficients for inhalation of particulate aerosols, ingestion in the workplace, and direct injection into blood for three uranium isotopes ($^{234,235,238}$U) given in Tables 27.4, were calculated using the parameter values in Tables 27.2 and 27.3.

## 27.5 Interpretation of Monitoring Data

The biokinetic model for uranium given above describes the kinetics of uranium retention, the main sites of deposition, as well as routes of excretion. It can be used to predict urinary and faecal excretion at any given time after uptake to blood (ICRP, 1997), and in conjunction with the HRTM and HATM, excretion at any given time after intake. Brief information on the Tables and Figures provided is given below. More details are given in Chapter 6.

The principal emissions from the three uranium isotopes of most importance ($^{234}$U, $^{235}$U and $^{238}$U) are given in Table 27.5. All three isotopes have physical half-lives that are long compared with time scales of retention in body tissues and emit alpha particles of similar energy, and hence have similar dose coefficients.

Table 27.6 gives detection limits for the monitoring techniques generally applied to uranium: lung retention by external gamma-ray spectrometry, and assay of urine and feces. In addition to typical limits that most laboratories should be able to achieve,

estimates are given of those achievable using current state of the art methods.

Values of the critical monitoring quantity $M_c$ for $^{235}U$ are listed in Table 27.7. Values of $M_c$ for $^{234}U$ and $^{238}U$ are still to be added. Material specific data are also to be provided in the final text.

The predicted lung retention and daily excretion in urine and faeces (Bq per Bq unit intake, as in Publication 78), following inhalation or ingestion were calculated using the biokinetic parameter values in Tables 27.2 and 27.3. Results, which apply to all uranium isotopes with physical half-lives that are long compared with biokinetic time scales are to be given on an accompanying CD-ROM.

Tables 27.8 and 27.9 give predicted values (Bq per Bq intake) for special monitoring for $^{234}U$, $^{235}U$ and $^{238}U$ for inhalation of Types F, M and S and uranyl nitrate, uranium tetrafluoride and uranium dioxide. Values are also given for ingestion and direct entry into the blood.

Predicted values for routine monitoring (Bq per Bq intake) following inhalation of $^{234}U$, $^{235}U$ and $^{238}U$ as Types F, M and S are given in Table 27.10 together with data for uranyl nitrate, uranium tetrafluoride and uranium dioxide.

Predicted values of committed effective dose from the inhalation of natural uranium per unit content of $^{235}U$ in the lungs and excreta (Sv per Bq content) are given in Table 27.11 for intakes of inhalation Types F, M and S and for inhalation of various chemical forms. The values were calculated from the content per unit intake tables (Tables 27.8 to 27.10) and the dose coefficients given in Table 27.4.

Predicted values of the effective dose corresponding to unit activity of natural uranium measured in 24 h urine samples are given in Table 27.12 for selected times between 1 day and 1000 days after the intake. The data were calculated in the same way as for Table 27.11.

The tables of dose per unit content (Table 27.11 and 27.12) are a new development for ICVP as described in Sections 1.3 and Annex D.

In practice $^{235}U$ occurs as one component of a mixture of uranium isotopes, depending on the degree of enrichment. Generally the dose from $^{234}U$ exceeds that from $^{235}U$, and except at high enrichments the dose from $^{238}U$ will too. To illustrate this, in Tables 27.12 and 27.13, intake of natural uranium (composition by activity 49% $^{234}U$; 2% $^{235}U$; 49% $^{238}U$) is considered. Thus the relevant dose is that from all three isotopes – about 50 times higher than that from $^{235}U$.

Note: This is most relevant to lung measurements, for which $^{235}U$ is likely to be measured. Urine and feces measurements may well include all three isotopes. Hence we could instead give tables for urine and feces that reflect this e.g. dose per Bq of uranium alpha-emitting isotopes. However, there is a general issue here. The presence of radionuclides in addition to that measured is more likely to be overlooked by the inexperienced analyst going directly from measurement to dose than going via intake.

Figures 27.2 to 27.7 give levels of activity in urine and faeces as a function of time after the intake (Bq per unit intake) for inhalation intakes of $^{234}U$, $^{235}U$ and $^{238}U$ following inhalation of Types F, M and S and of uranyl nitrate, uranium tetrafluoride and uranium dioxide. Figures 27.8 and 27.9 give excretion in urine following intakes by ingestion ($f_A$ = 0.02 and 0.002).

The Annex gives examples of additional tables of equivalent dose to tissues as a function

of time after the intake (Table A1 to A5). These tables are expected to be included on an accompanying CD-ROM together with Figures A27.1 to A27.8 giving the activity in urine, faeces or lungs to give a committed effective dose of 1 mSv. The values relate to committed doses of 1 mSv since the smaller unit is more relevant to monitoring considerations. Again examples are given for unit dose of "pure" $^{235}U$, and of the dose from natural uranium based on measurement of the $^{235}U$ present.

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Table 27.4. Committed effective dose coefficients (Sv/Bq) for the inhalation (5 µm AMAD aerosols), ingestion or direct uptake to blood of $^{234}U$, $^{235}U$, and $^{238}U$ uranium compounds

| | Effective dose coefficients (Sv $Bq^{-1}$) | | |
|---|---|---|---|
| Inhaled particulate materials | $^{234}U$ | $^{235}U$ | $^{238}U$ |
| Compound | | | |
| Uranyl nitrate, $UO_2(NO_3)_2$ | 4.4E-07 | 4.0E-07 | 3.7E-07 |
| Uranyl Tri-Butyl-Phosphate, U-TBP | 3.8E-07 | 3.4E-07 | 3.3E-07 |
| Uranium peroxide hydrate, $UO_4$ | 3.8E-07 | 3.4E-07 | 3.1E-07 |
| Ammonium diuranate, ADU | 5.2E-07 | 4.6E-07 | 4.2E-07 |
| Uranium tetrafluoride, $UF_4$ | 1.2E-06 | 1.1E-06 | 9.7E-07 |
| Uranium trioxide, $UO_3$ | 5.9E-07 | 5.1E-07 | 4.7E-07 |
| Triuranium octoxide, $U_3O_8$ | 2.3E-06 | 2.0E-06 | 1.8E-06 |
| Uranium dioxide $UO_2$ | 5.0E-06 | 4.5E-06 | 4.2E-06 |
| Type F | 6.4E-07 | 6.0E-07 | 5.8E-07 |
| Type M (all unspecified compounds) | 2.1E-06 | 1.8E-06 | 1.6E-06 |
| Type S | 6.8E-06 | 6.1E-06 | 5.7E-06 |
| Ingested materials | | | |
| Relatively insoluble compounds (e.g. $UF_4$, $UO_2$, $U_3O_8$): $f_A$ = 0.002 | 8.3E-09 | 8.3E-09 | 7.6E-09 |
| All other chemical forms of uranium: $f_A$ = 0.02 | 4.9E-08 | 4.6E-08 | 4.4E-08 |
| Direct uptake to blood | 2.3E-06 | 2.1E-06 | 2.1E-06 |

Table 27.5. Principal emissions of $^{234}U$, $^{235}U$, and $^{238}U$

| Radionuclide | Radiation | Energy (MeV) | Intensity (%) |
|---|---|---|---|
| $^{234}U$ | α | 4.72 | 27 |
| | α | 4.77 | 72 |
| $^{235}U$ | α | 4.22 | 6 |
| | α | 4.33 | 5 |
| | α | 4.37 | 18 |
| | α | 4.40 | 56 |
| | α | 4.56 | 4 |
| | α | 4.60 | 5 |
| | γ | 0.14 | 11 |
| | γ | 0.19 | 54 |
| | γ | 0.21 | 5 |
| $^{238}U$ | α | 4.15 | 23 |
| | α | 4.20 | 77 |

*Note, will need to be consistent with new radiation data (revision of* Publication 38*)*

**[Decay products to add]**

Table 27.6. Measurement techniques for $^{234}U$, $^{235}U$ and $^{238}U$

| Method of measurement | | Typical detection limit | Achievable |
|---|---|---|---|
| $\gamma$-ray spectrometry in vivo | Lung[a] | 20 Bq | 2 Bq |
| Radiochemical separation and $\alpha$-spectrometry on biological samples | Urine[b] | 1 mBq | 0.1 mBq |
| | Faeces[b] | 10 mBq | 1 mBq |

a For $^{235}U$ only
b Daily excretion
[Note: 234 Th can be used to measure 238U – to be included]

Table 27.7. Critical Monitoring quantity, $M_C$ (dose <0.1 mSv) for routine monitoring of inhalation of $^{234}U$, $^{235}U$ and $^{238}U$

| Absorption Type | Type of measurement | Monitoring interval (d) | Critical Monitoring quantity, $M_C$ | | |
|---|---|---|---|---|---|
| | | | $^{234}U$ | $^{235}U$ | $^{238}U$ |
| F | Urine[a] | 90 | **to add** | 1.4 mBq | **to add** |
| | | 180 | | 1.0 mBq | |
| | | 360 | | 0.5 mBq | |
| M | Lungs | 90 | | (0.14 Bq) | |
| | | 180 | | (0.2 Bq) | |
| | | 360 | | (0.2 Bq) | |
| | Urine[a] | 90 | | 0.8 mBq | |
| | | 180 | | 1.0 mBq | |
| | | 360 | | 1.1 mBq | |
| S | Lungs | 90 | | (0.2 Bq) | |
| | | 180 | | (0.3 Bq) | |
| | | 360 | | (0.5 Bq) | |
| | Urine[a] | 90 | | (0.02 mBq) | |
| | | 180 | | (0.04 mBq) | |
| | | 360 | | (0.05 mBq) | |

a Daily excretion

**[Note: to be extended for material special parameter values ]**

Table 27.8. Special monitoring: predicted values (Bq per Bq intake) for inhalation of $^{234}U$, $^{235}U$ or $^{238}U$

| Time after intake (d) | Type F | Type M | | | Type S | | |
|---|---|---|---|---|---|---|---|
| | Urine[a] | Lungs[b] | Urine | Faeces[a] | Lungs | Urine[a] | Faeces[b] |
| 1 | 1.8E-01 | 5.8E-02 | 2.3E-02 | | 6.4E-02 | 7.0E-04 | 1.1E-01 |
| 2 | 6.4E-03 | 5.6E-02 | 1.1E-03 | | 6.3E-02 | 4.4E-05 | 1.6E-01 |
| 3 | 5.1E-03 | 5.5E-02 | 8.5E-04 | | 6.2E-02 | 2.6E-05 | 8.4E-02 |
| 4 | 4.6E-03 | 5.4E-02 | 7.9E-04 | | 6.1E-02 | 2.4E-05 | 3.5E-02 |
| 5 | 4.2E-03 | 5.3E-02 | 7.3E-04 | | 6.1E-02 | 2.2E-05 | 1.4E-02 |
| 6 | 3.8E-03 | 5.3E-02 | 6.9E-04 | | 6.0E-02 | 2.0E-05 | 5.7E-03 |
| 7 | 3.5E-03 | 5.2E-02 | 6.5E-04 | | 6.0E-02 | 1.9E-05 | 2.5E-03 |
| ..... | | | | | | | |
| 20,000 | | | | | | | |

a Daily urinary or faecal excretion
b Lung monitoring is feasible only for $^{235}U$

Table 27.8. (cont.)

| Time after intake (d) | Uranyl nitrate, $UO_2(NO_3)_2$ | Uranium tetrafluoride, $UF_4$ | | | Uranium dioxide, $UO_2$ | | |
|---|---|---|---|---|---|---|---|
| | Urine | Lungs[a] | Urine | Faeces[b] | Lungs | Urine[a] | Faeces[b] |
| 1 | 4.9E-02 | 5.9E-02 | 8.6E-03 | | 6.4E-02 | 9.5E-04 | 1.1E-01 |
| 2 | 5.3E-03 | 5.3E-02 | 3.5E-03 | | 6.2E-02 | 2.4E-04 | 1.6E-01 |
| 3 | 1.7E-03 | 4.8E-02 | 3.0E-03 | | 6.1E-02 | 1.2E-04 | 8.4E-02 |
| 4 | 1.4E-03 | 4.4E-02 | 2.7E-03 | | 6.0E-02 | 7.7E-05 | 3.5E-02 |
| 5 | 1.3E-03 | 4.1E-02 | 2.4E-03 | | 6.0E-02 | 6.0E-05 | 1.4E-02 |
| 6 | 1.2E-03 | 3.8E-02 | 2.1E-03 | | 5.9E-02 | 5.3E-05 | 5.6E-03 |
| 7 | 1.1E-03 | 3.6E-02 | 1.9E-03 | | 5.9E-02 | 4.9E-05 | 2.5E-03 |
| ..... | | | | | | | |
| 20,000 | | | | | | | |

a Daily urinary or faecal excretion
b Lung monitoring is feasible only for $^{235}U$

**[Table incomplete]**

Table 27.9. Special monitoring: predicted values (Bq per Bq intake) for ingestion and injection of $^{234}U$, $^{235}U$ or $^{238}U$

| Time after intake (d) | Ingestion, $f_A$ = 0.02 | Ingestion, $f_A$ = 0.002 | Injection |
|---|---|---|---|
| | Urine | Urine | Urine |
| 1 | 1.3E-02 | 1.3E-03 | 6.5E-01 |
| 2 | 6.9E-04 | 7.0E-05 | 2.2E-02 |
| 3 | 3.7E-04 | 3.7E-05 | 1.8E-02 |
| 4 | 3.3E-04 | 3.3E-05 | 1.6E-02 |
| 5 | 3.0E-04 | 3.0E-05 | 1.5E-02 |
| 6 | 2.7E-04 | 2.7E-05 | 1.3E-02 |
| 7 | 2.5E-04 | 2.5E-05 | 1.2E-02 |
| ... | | | |
| 20000 | | | |

Table 27.10. Routine monitoring: predicted values (Bq per Bq intake) for inhalation of $^{234}U$, $^{235}U$ or $^{238}U$

| Monitoring interval (d) | Type F | Type M | | Type S | | |
|---|---|---|---|---|---|---|
| | Urine[a] | Lungs[b] | Urine | Lungs | Urine | Faeces[a] |
| 7 | 3.0E-02 | 5.5E-02 | 4.0E-03 | 6.2E-02 | 1.2E-04 | 5.9E-02 |
| 14 | 1.6E-02 | 5.2E-02 | 2.2E-03 | 6.0E-02 | 6.9E-05 | 3.0E-02 |
| 30 | 8.2E-03 | 4.7E-02 | 1.2E-03 | 5.6E-02 | 3.7E-05 | 1.4E-02 |
| 60 | 4.3E-03 | 4.0E-02 | 7.2E-04 | 5.1E-02 | 2.2E-05 | 7.3E-03 |
| 90 | 2.9E-03 | 3.5E-02 | 5.3E-04 | 4.7E-02 | 1.6E-05 | 4.9E-03 |
| 120 | 2.2E-03 | 3.1E-02 | 4.2E-04 | 4.4E-02 | 1.3E-05 | 3.7E-03 |
| 180 | 1.5E-03 | 2.5E-02 | 3.1E-04 | 4.1E-02 | 9.9E-06 | 2.5E-03 |
| 360 | 7.5E-04 | 1.6E-02 | 1.7E-04 | 3.5E-02 | 6.4E-06 | 1.3E-03 |

a Daily urinary or faecal excretion
b Lung monitoring is feasible only for $^{235}U$

Table 27.11. Predicted values of dose from inhalation of natural U per unit content of $^{235}U$ in lungs and for daily excretion of $^{234}U$ or $^{238}U$ in urine and faeces (Sv per Bq content) (5 µm AMAD aerosols inhaled by a Reference Worker at Light Work)

| Time after intake (d) | Mon Int. | Type F | Type M | | | Type S | | |
|---|---|---|---|---|---|---|---|---|
| | | Urine | Lungs $^{235}U$ | Urine[a,b] $^{234/238}U$ | Faeces[a,b] $^{234/238}U$ | Lungs $^{235}U$ | Urine[a,b] $^{234/238}U$ | Faeces[a,b] $^{234/238}U$ |
| 1 | | 6.79E-06 | 1.43E-03 | 1.63E-04 | 3.54E-05 | 4.32E-03 | 1.81E-02 | 1.12E-04 |
| 2 | | 1.96E-04 | 1.47E-03 | 3.36E-03 | 2.50E-05 | 4.43E-03 | 2.90E-01 | 7.85E-05 |
| 3 | 6 | 2.43E-04 | 1.49E-03 | 4.43E-03 | 4.86E-05 | 4.48E-03 | 4.92E-01 | 1.52E-04 |
| 4 | | 2.69E-04 | 1.52E-03 | 4.80E-03 | 1.16E-04 | 4.53E-03 | 5.40E-01 | 3.63E-04 |
| 5 | | 2.96E-04 | 1.54E-03 | 5.15E-03 | 2.91E-04 | 4.57E-03 | 5.82E-01 | 9.11E-04 |
| 6 | | 3.25E-04 | 1.56E-03 | 5.50E-03 | 7.26E-04 | 4.62E-03 | 6.24E-01 | 2.26E-03 |
| 7 | 14 | 3.57E-04 | 1.59E-03 | 5.86E-03 | 1.68E-03 | 4.66E-03 | 6.66E-01 | 5.18E-03 |
| 8 | | 3.90E-04 | 1.61E-03 | 6.22E-03 | 3.31E-03 | 4.71E-03 | 7.09E-01 | 1.00E-02 |
| 9 | | 4.26E-04 | 1.63E-03 | 6.58E-03 | 5.22E-03 | 4.75E-03 | 7.53E-01 | 1.56E-02 |
| 10 | | 4.64E-04 | 1.65E-03 | 6.95E-03 | 6.74E-03 | 4.80E-03 | 7.97E-01 | 1.98E-02 |
| 15 | 30 | 6.93E-04 | 1.77E-03 | 8.84E-03 | 9.19E-03 | 5.01E-03 | 1.02E+00 | 2.59E-02 |
| 30 | 60 | 1.83E-03 | 2.14E-03 | 1.42E-02 | 1.39E-02 | 5.62E-03 | 1.66E+00 | 3.65E-02 |
| 45 | 90 | 3.52E-03 | 2.53E-03 | 1.87E-02 | 2.07E-02 | 6.17E-03 | 2.12E+00 | 5.06E-02 |
| 60 | 120 | 5.50E-03 | 2.92E-03 | 2.27E-02 | 3.03E-02 | 6.63E-03 | 2.47E+00 | 6.88E-02 |
| 90 | 180 | 1.04E-02 | 3.77E-03 | 3.04E-02 | 6.08E-02 | 7.37E-03 | 2.98E+00 | 1.20E-01 |
| 180 | 360 | 4.00E-02 | 6.89E-03 | 5.79E-02 | 2.66E-01 | 8.68E-03 | 3.88E+00 | 3.43E-01 |
| 300 | | 1.41E-01 | 1.41E-02 | 1.19E-01 | 7.26E-01 | 9.85E-03 | 4.54E+00 | 5.26E-01 |
| 1000 | | 5.41E-01 | 8.08E-01 | 3.39E+00 | 4.67E+01 | 1.83E-02 | 8.38E+00 | 5.65E-01 |

Table 27.11 (cont.)

| Time after intake (d) | Mon Int. | Uranyl nitrate, $UO_2(NO_3)_2$ | Uranium tetrafluoride, $UF_4$ | | | Uranium dioxide, $UO_2$ | | |
|---|---|---|---|---|---|---|---|---|
| | | Urine | Lungs $^{235}U$ | Urine[a,b] $^{234/238}U$ | Faeces[a,b] $^{234/238}U$ | Lungs $^{235}U$ | Urine $^{234/238}U$ | Faeces $^{234/238}U$ |
| 1 | | 1.68E-05 | 8.20E-04 | 2.59E-04 | 1.98E-05 | 3.21E-03 | 9.89E-03 | 8.25E-05 |
| 2 | | 1.56E-04 | 9.14E-04 | 6.26E-04 | 1.39E-05 | 3.30E-03 | 3.89E-02 | 5.78E-05 |
| 3 | 6 | 4.80E-04 | 1.00E-03 | 7.41E-04 | 2.71E-05 | 3.35E-03 | 7.99E-02 | 1.12E-04 |
| 4 | | 5.86E-04 | 1.09E-03 | 8.34E-04 | 6.46E-05 | 3.38E-03 | 1.23E-01 | 2.67E-04 |
| 5 | | 6.49E-04 | 1.18E-03 | 9.36E-04 | 1.63E-04 | 3.42E-03 | 1.57E-01 | 6.71E-04 |
| 6 | | 7.13E-04 | 1.27E-03 | 1.05E-03 | 4.13E-04 | 3.46E-03 | 1.79E-01 | 1.67E-03 |
| 7 | 14 | 7.81E-04 | 1.36E-03 | 1.17E-03 | 9.88E-04 | 3.49E-03 | 1.94E-01 | 3.82E-03 |
| 8 | | 8.53E-04 | 1.44E-03 | 1.31E-03 | 2.10E-03 | 3.53E-03 | 2.05E-01 | 7.45E-03 |
| 9 | | 9.29E-04 | 1.53E-03 | 1.45E-03 | 3.70E-03 | 3.56E-03 | 2.15E-01 | 1.16E-02 |
| 10 | | 1.01E-03 | 1.62E-03 | 1.62E-03 | 5.34E-03 | 3.59E-03 | 2.24E-01 | 1.48E-02 |
| 15 | 30 | 1.49E-03 | 2.00E-03 | 2.65E-03 | 9.57E-03 | 3.76E-03 | 2.63E-01 | 1.95E-02 |
| 30 | 60 | 3.76E-03 | 2.77E-03 | 8.12E-03 | 1.78E-02 | 4.25E-03 | 3.51E-01 | 2.75E-02 |
| 45 | 90 | 6.84E-03 | 3.33E-03 | 1.51E-02 | 2.72E-02 | 4.69E-03 | 4.09E-01 | 3.84E-02 |
| 60 | 120 | 1.02E-02 | 3.86E-03 | 2.12E-02 | 3.99E-02 | 5.07E-03 | 4.52E-01 | 5.25E-02 |
| 90 | 180 | 1.76E-02 | 4.97E-03 | 3.18E-02 | 8.00E-02 | 5.71E-03 | 5.19E-01 | 9.25E-02 |
| 180 | 360 | 5.21E-02 | 9.10E-03 | 6.81E-02 | 3.50E-01 | 6.96E-03 | 6.48E-01 | 2.74E-01 |
| 300 | | 1.39E-01 | 1.86E-02 | 1.47E-01 | 9.57E-01 | 8.29E-03 | 7.77E-01 | 4.41E-01 |
| 1000 | | 1.09E+00 | 1.07E+00 | 3.02E+00 | 5.91E+01 | 2.04E-02 | 1.88E+00 | 1.28E+00 |

a Measurement of $^{234}U$ or $^{238}U$
b Daily excretion in urine and faeces
c More information on specific isotopes is given on the CD

Table 27.12. Special monitoring: predicted values of dose from natural U per unit content (Sv per Bq content) following ingestion or uptake to blood of natural U

| Time after intake (d) | Mon Int. | Ingestion, $f_A$ = 0.02 | Ingestion, $f_A$ = 0.002 | Direct uptake to blood |
|---|---|---|---|---|
| | | Urine[a] $^{234/238}$U | Urine $^{234/238}$U | Urine $^{234/238}$U |
| 1 | | 1.05E-12 | 1.83E-12 | 9.57E-13 |
| 2 | | 1.91E-11 | 3.26E-11 | 2.79E-11 |
| 3 | 6 | 3.58E-11 | 6.22E-11 | 3.43E-11 |
| 4 | | 3.98E-11 | 6.92E-11 | 3.79E-11 |
| 5 | | 4.38E-11 | 7.63E-11 | 4.18E-11 |
| 6 | | 4.82E-11 | 8.38E-11 | 4.59E-11 |
| 7 | 14 | 5.29E-11 | 9.20E-11 | 5.04E-11 |
| 8 | | 5.78E-11 | 1.01E-10 | 5.51E-11 |
| 9 | | 6.31E-11 | 1.10E-10 | 6.01E-11 |
| 10 | | 6.88E-11 | 1.20E-10 | 6.55E-11 |
| 15 | 30 | 1.03E-10 | 1.79E-10 | 9.79E-11 |
| 30 | 60 | 2.73E-10 | 4.76E-10 | 2.58E-10 |
| 45 | 90 | 5.27E-10 | 9.18E-10 | 4.97E-10 |
| 60 | 120 | 8.26E-10 | 1.44E-09 | 7.76E-10 |
| 90 | 180 | 1.57E-09 | 2.72E-09 | 1.47E-09 |
| 180 | 360 | 6.02E-09 | 1.05E-08 | 5.64E-09 |
| 300 | | 2.12E-08 | 3.69E-08 | 1.99E-08 |
| 1000 | | 8.16E-08 | 1.42E-07 | 7.63E-08 |

a Daily excretion in urine

**Fig. 27.2. Inhalation Type F: predicted activity of U-234, U-235 or U-238 following unit acute intake**

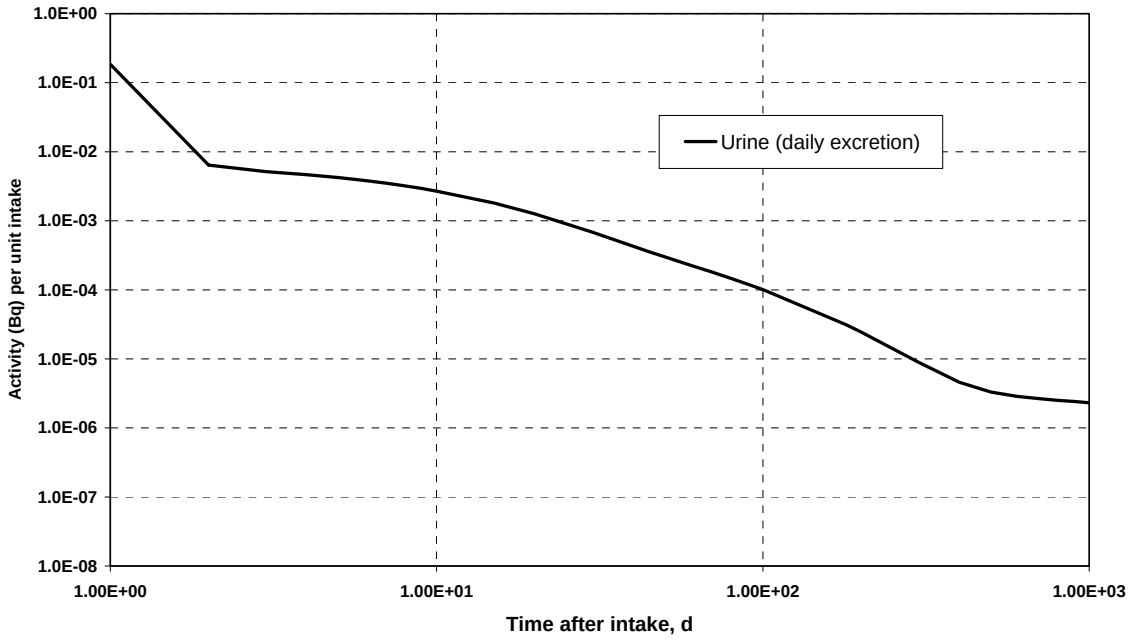


**Fig. 27.3. Inhalation Type M: predicted activity of U-234, U-235 or U-238 following unit acute intake**

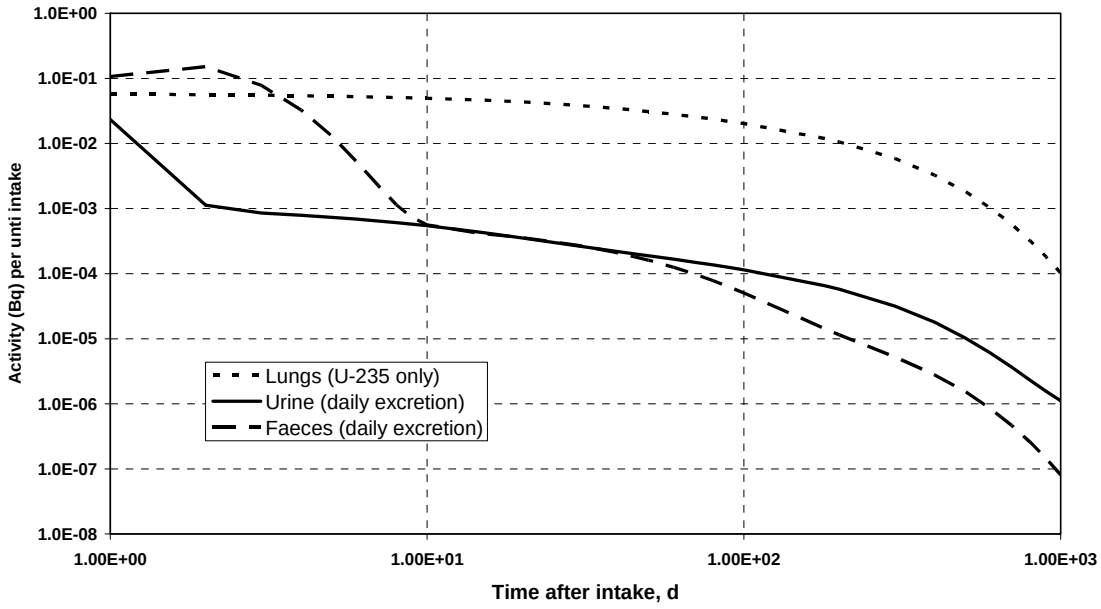


**Fig. 27.4. Inhalation Type S: predicted activity of U-234, U-235 or U-238 following unit acute intake**

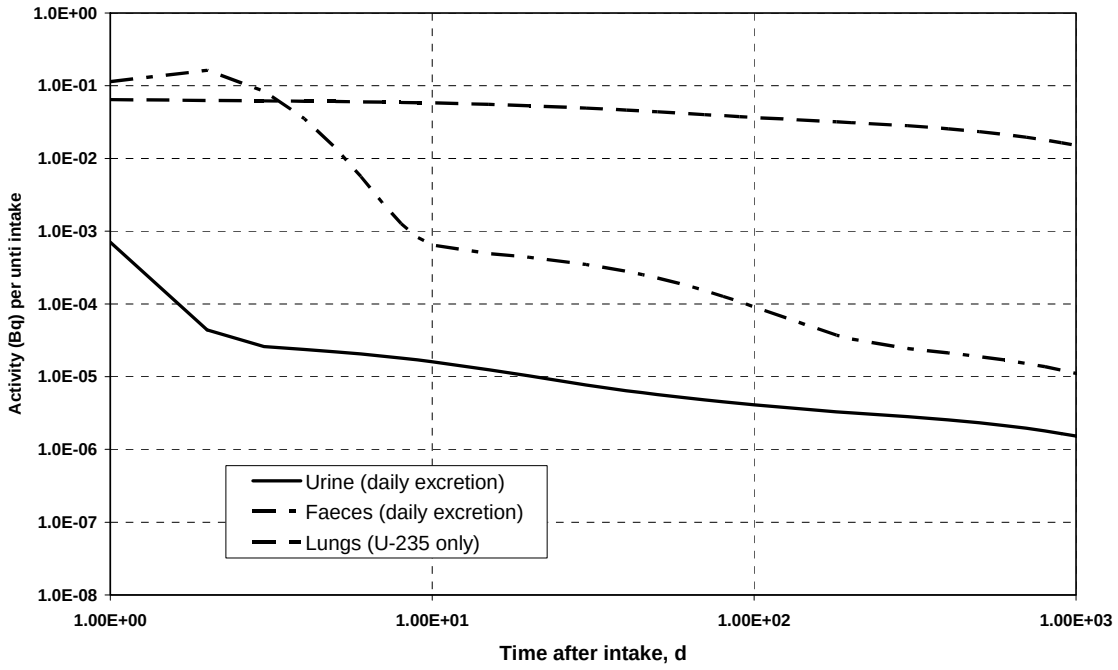



**Fig. 27.5. Inhalation of uranyl nitrate: predicted activity of U-234, U-235 or U-238 following unit acute intake**

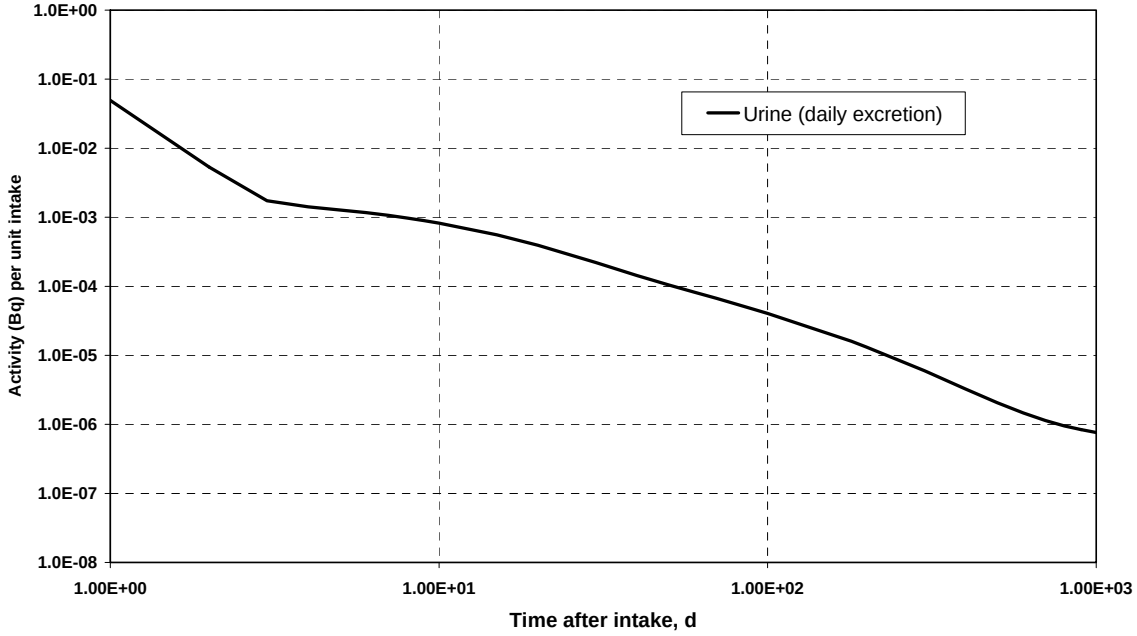


**Fig. 27.6. Inhalation uranium tetrafluoride: predicted activity of U-234, U-235 or U-238 following unit acute intake**

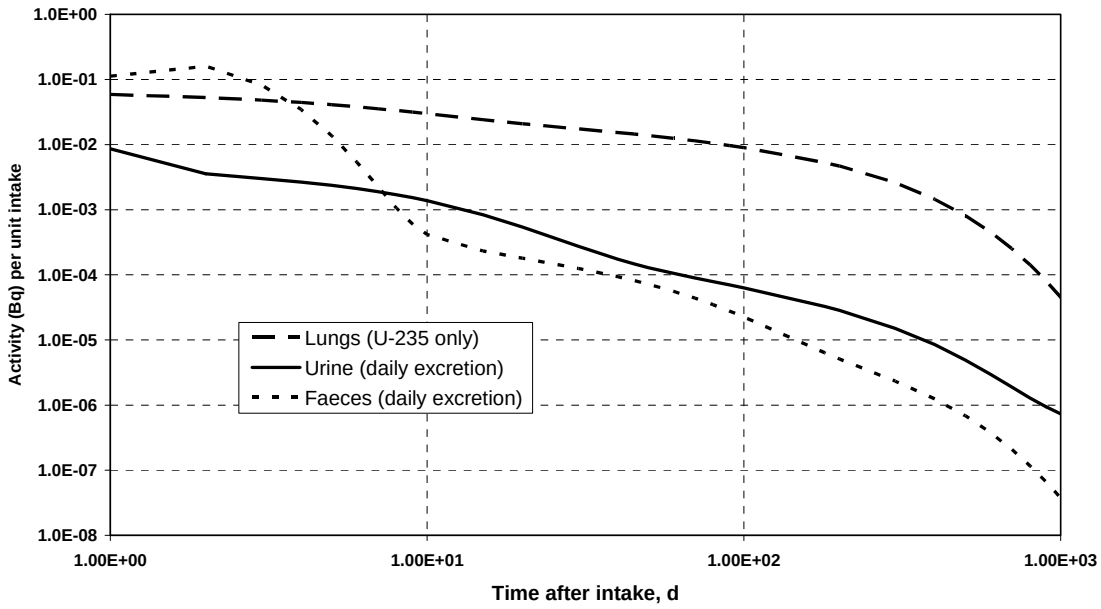



**Fig. 27.7. Inhalation uranium dioxide: predicted activity of U-234, U-235 or U-238 following unit acute intake**

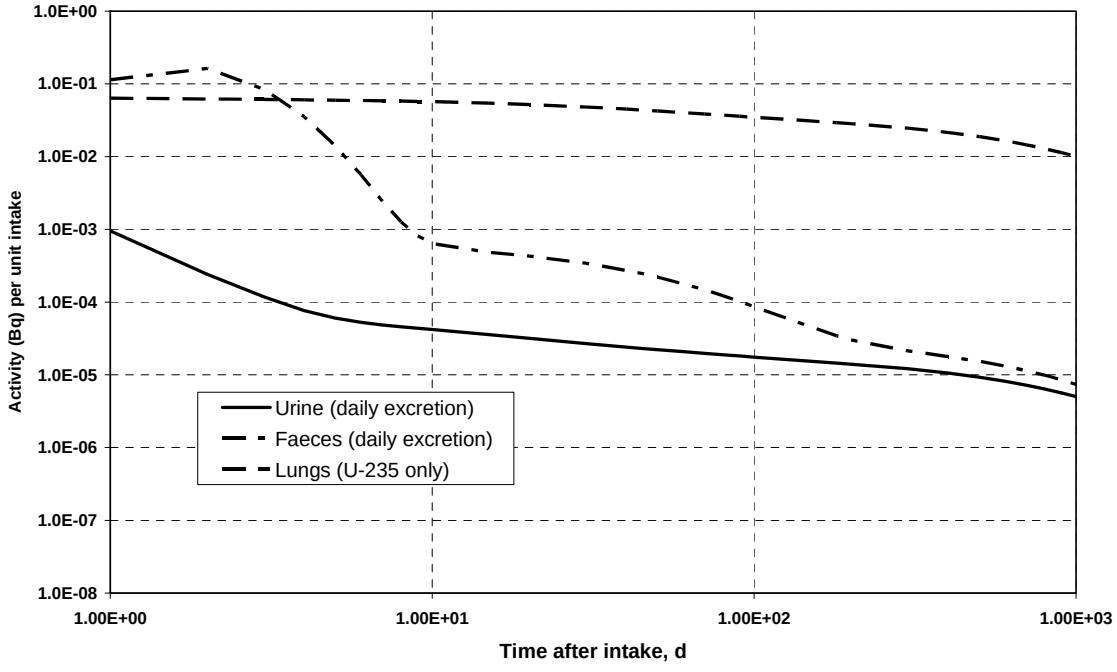


**Fig. 27.8. Ingestion (f1 = 0.02): predicted activity of U-234, U-235 or U-238 following unit acute intake**

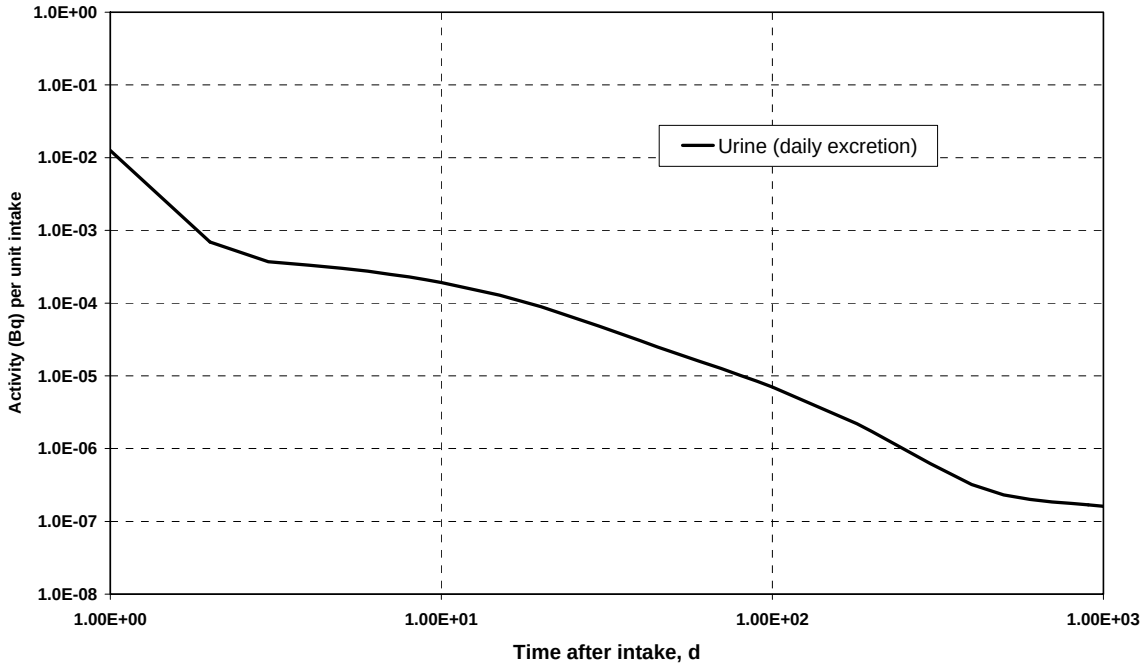
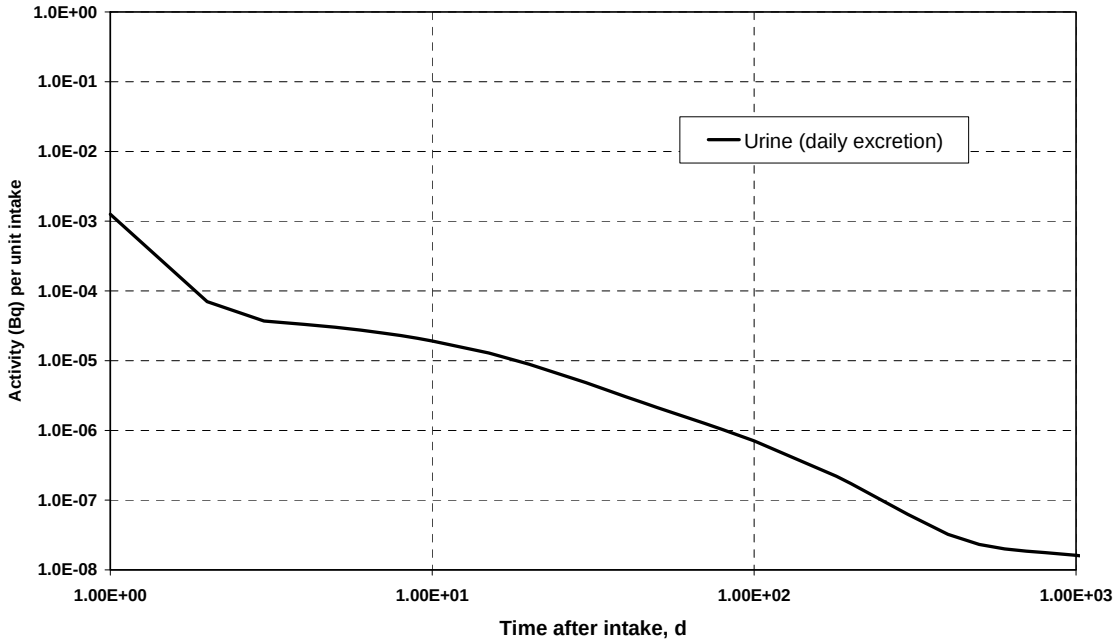



**Fig. 27.9. Ingestion (f1 = 0.002): predicted activity of U-234, U-235 or U-238 following unit acute intake**



**ANNEX**

## Example electronic data for uranium to go on CD-ROM

### *A1 Dose coefficients*

Table A1. U-235, adult worker Inhalation of particulate aerosol: AMAD = 5.000 micron, absorption Type F, $f_A$ = 0.02

| Time after intake | 7 days | 30 days | 1 year | 10 years | 50 years |
|---|---|---|---|---|---|
| Adrenals | 3.2E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Bladder Wall | 4.2E-09 | 1.0E-08 | 2.4E-08 | 9.3E-08 | 3.6E-07 |
| Bone Surface | 5.2E-07 | 1.1E-06 | 2.5E-06 | 6.2E-06 | 1.1E-05 |
| Brain | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Breast | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| GI-Tract | | | | | |
| Oesophagus | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| St Wall | 3.5E-09 | 9.5E-09 | 2.3E-08 | 9.2E-08 | 3.6E-07 |
| SI Wall | 3.8E-09 | 9.8E-09 | 2.3E-08 | 9.3E-08 | 3.6E-07 |
| ULI Wall | 6.7E-09 | 1.3E-08 | 2.6E-08 | 9.6E-08 | 3.6E-07 |
| LLI Wall | 1.4E-08 | 2.0E-08 | 3.3E-08 | 1.0E-07 | 3.7E-07 |
| Colon | 9.7E-09 | 1.6E-08 | 2.9E-08 | 9.9E-08 | 3.6E-07 |
| Kidneys | 7.1E-07 | 1.5E-06 | 2.0E-06 | 3.2E-06 | 3.7E-06 |
| Liver | 1.6E-08 | 3.7E-08 | 1.2E-07 | 7.2E-07 | 1.4E-06 |
| Muscle | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Ovaries | 3.3E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Pancreas | 3.2E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Red Marrow | 5.0E-08 | 1.1E-07 | 2.7E-07 | 7.0E-07 | 1.1E-06 |
| Respiratory Tract | | | | | |
| ET Airways | 7.1E-09 | 1.3E-08 | 2.6E-08 | 9.6E-08 | 3.6E-07 |
| Lungs | 1.2E-08 | 1.8E-08 | 3.1E-08 | 1.0E-07 | 3.7E-07 |
| Skin | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Spleen | 3.2E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Testes | 3.2E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Thymus | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Thyroid | 3.2E-09 | 9.2E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Uterus | 3.2E-09 | 9.3E-09 | 2.2E-08 | 9.2E-08 | 3.6E-07 |
| Remainder | 1.0E-08 | 2.4E-08 | 4.2E-08 | 1.2E-07 | 3.9E-07 |
| Effective dose | 1.7E-08 | 3.6E-08 | 8.5E-08 | 2.6E-07 | 6.0E-07 |

Table A2 U-235, adult worker. Inhalation of particulate aerosol: AMAD = 5.000 micron, absorption Type M, $f_A$ = 0.02

| Time after intake | 7 days | 30 days | 1 year | 10 years | 50 years |
|---|---|---|---|---|---|
| Adrenals | 4.4E-10 | 1.4E-09 | 5.1E-09 | 2.2E-08 | 8.8E-08 |
| Bladder Wall | 5.7E-10 | 1.5E-09 | 5.2E-09 | 2.2E-08 | 8.8E-08 |
| Bone Surface | 6.8E-08 | 1.6E-07 | 5.5E-07 | 1.5E-06 | 2.6E-06 |
| Brain | 4.2E-10 | 1.3E-09 | 4.9E-09 | 2.2E-08 | 8.7E-08 |
| Breast | 4.3E-10 | 1.4E-09 | 5.0E-09 | 2.2E-08 | 8.7E-08 |
| GI-Tract | | | | | |
| Oesophagus | 4.4E-10 | 1.4E-09 | 5.1E-09 | 2.2E-08 | 8.7E-08 |
| St Wall | 8.7E-10 | 1.8E-09 | 5.4E-09 | 2.3E-08 | 8.8E-08 |
| SI Wall | 1.6E-09 | 2.5E-09 | 6.1E-09 | 2.3E-08 | 8.9E-08 |
| ULI Wall | 7.3E-09 | 8.3E-09 | 1.2E-08 | 2.9E-08 | 9.5E-08 |
| LLI Wall | 2.1E-08 | 2.3E-08 | 2.7E-08 | 4.4E-08 | 1.1E-07 |
| Colon | 1.3E-08 | 1.4E-08 | 1.9E-08 | 3.6E-08 | 1.0E-07 |
| Kidneys | 9.2E-08 | 2.2E-07 | 4.5E-07 | 7.8E-07 | 9.1E-07 |
| Liver | 2.1E-09 | 5.3E-09 | 2.6E-08 | 1.7E-07 | 3.4E-07 |
| Muscle | 4.4E-10 | 1.3E-09 | 5.0E-09 | 2.2E-08 | 8.7E-08 |
| Ovaries | 5.0E-10 | 1.4E-09 | 5.0E-09 | 2.2E-08 | 8.8E-08 |
| Pancreas | 4.4E-10 | 1.4E-09 | 5.0E-09 | 2.2E-08 | 8.7E-08 |
| Red Marrow | 6.5E-09 | 1.6E-08 | 5.9E-08 | 1.7E-07 | 2.6E-07 |
| Respiratory Tract | | | | | |
| ET Airways | 4.7E-07 | 1.9E-06 | 1.0E-05 | 1.1E-05 | 1.2E-05 |
| Lungs | 2.7E-06 | 7.8E-06 | 1.4E-05 | 1.4E-05 | 1.4E-05 |
| Skin | 4.2E-10 | 1.3E-09 | 4.9E-09 | 2.2E-08 | 8.7E-08 |
| Spleen | 4.3E-10 | 1.4E-09 | 5.0E-09 | 2.2E-08 | 8.7E-08 |
| Testes | 4.3E-10 | 1.3E-09 | 4.9E-09 | 2.2E-08 | 8.7E-08 |
| Thymus | 4.4E-10 | 1.4E-09 | 5.1E-09 | 2.2E-08 | 8.7E-08 |
| Thyroid | 4.3E-10 | 1.3E-09 | 5.0E-09 | 2.2E-08 | 8.7E-08 |
| Uterus | 4.6E-10 | 1.4E-09 | 4.9E-09 | 2.2E-08 | 8.7E-08 |
| Remainder | 1.6E-09 | 4.5E-09 | 1.4E-08 | 3.5E-08 | 1.0E-07 |
| Effective dose | 3.2E-07 | 9.4E-07 | 1.7E-06 | 1.7E-06 | 1.8E-06 |

Table A3. U-235, adult worker. Inhalation of particulate aerosol: AMAD = 5.000 micron, absorption Type S, $f_A$ = 0.002

| Time after intake | 7 days | 30 days | 1 year | 10 years | 50 years |
|---|---|---|---|---|---|
| Adrenals | 2.9E-11 | 8.9E-11 | 5.2E-10 | 2.7E-09 | 9.7E-09 |
| Bladder Wall | 4.5E-11 | 7.4E-11 | 2.2E-10 | 1.5E-09 | 8.2E-09 |
| Bone Surface | 2.1E-09 | 4.8E-09 | 1.9E-08 | 1.1E-07 | 2.6E-07 |
| Brain | 1.8E-11 | 4.7E-11 | 1.9E-10 | 1.5E-09 | 8.2E-09 |
| Breast | 2.6E-11 | 8.5E-11 | 5.0E-10 | 2.6E-09 | 9.6E-09 |
| GI-Tract | | | | | |
| Oesophagus | 3.2E-11 | 9.9E-11 | 5.8E-10 | 2.9E-09 | 1.0E-08 |
| St Wall | 4.9E-10 | 5.5E-10 | 8.6E-10 | 2.6E-09 | 9.4E-09 |
| SI Wall | 1.2E-09 | 1.3E-09 | 1.5E-09 | 3.0E-09 | 9.6E-09 |
| ULI Wall | 7.4E-09 | 7.6E-09 | 8.2E-09 | 9.9E-09 | 1.7E-08 |
| LLI Wall | 2.2E-08 | 2.3E-08 | 2.4E-08 | 2.7E-08 | 3.4E-08 |
| Colon | 1.4E-08 | 1.4E-08 | 1.5E-08 | 1.7E-08 | 2.4E-08 |
| Kidneys | 2.8E-09 | 6.6E-09 | 1.6E-08 | 5.9E-08 | 9.5E-08 |
| Liver | 8.0E-11 | 2.1E-10 | 1.2E-09 | 1.3E-08 | 3.6E-08 |
| Muscle | 3.2E-11 | 7.4E-11 | 3.4E-10 | 2.0E-09 | 8.8E-09 |
| Ovaries | 1.0E-10 | 1.3E-10 | 2.8E-10 | 1.6E-09 | 8.2E-09 |
| Pancreas | 3.0E-11 | 8.1E-11 | 4.2E-10 | 2.3E-09 | 9.2E-09 |
| Red Marrow | 2.2E-10 | 5.1E-10 | 2.2E-09 | 1.3E-08 | 2.7E-08 |
| Respiratory Tract | | | | | |
| ET Airways | 5.3E-07 | 2.2E-06 | 2.3E-05 | 6.8E-05 | 6.9E-05 |
| Lungs | 3.0E-06 | 9.2E-06 | 2.0E-05 | 3.2E-05 | 3.6E-05 |
| Skin | 1.9E-11 | 5.2E-11 | 2.4E-10 | 1.7E-09 | 8.4E-09 |
| Spleen | 2.8E-11 | 7.9E-11 | 4.2E-10 | 2.3E-09 | 9.2E-09 |
| Testes | 2.0E-11 | 4.7E-11 | 1.8E-10 | 1.5E-09 | 8.1E-09 |
| Thymus | 3.2E-11 | 9.9E-11 | 5.8E-10 | 2.9E-09 | 1.0E-08 |
| Thyroid | 2.2E-11 | 6.1E-11 | 3.0E-10 | 1.9E-09 | 8.7E-09 |
| Uterus | 5.1E-11 | 8.0E-11 | 2.2E-10 | 1.5E-09 | 8.2E-09 |
| Remainder | 2.7E-07 | 1.1E-06 | 1.1E-05 | 3.4E-05 | 3.5E-05 |
| Effective dose | 3.7E-07 | 1.2E-06 | 2.9E-06 | 5.5E-06 | 6.1E-06 |

Table A4 U-235, adult worker. Ingestion: $f_A$ = 0.02

| Time after intake | 7 days | 30 days | 1 year | 10 years | 50 years |
|---|---|---|---|---|---|
| Adrenals | 2.3E-10 | 6.6E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Bladder Wall | 3.5E-10 | 7.9E-10 | 1.7E-09 | 6.6E-09 | 2.5E-08 |
| Bone Surface | 3.6E-08 | 7.6E-08 | 1.7E-07 | 4.4E-07 | 7.4E-07 |
| Brain | 2.2E-10 | 6.4E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Breast | 2.2E-10 | 6.5E-10 | 1.6E-09 | 6.4E-09 | 2.5E-08 |
| GI-Tract | | | | | |
| Oesophagus | 2.2E-10 | 6.5E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| St Wall | 1.3E-09 | 1.8E-09 | 2.7E-09 | 7.6E-09 | 2.6E-08 |
| SI Wall | 3.1E-09 | 3.5E-09 | 4.4E-09 | 9.3E-09 | 2.8E-08 |
| ULI Wall | 1.7E-08 | 1.8E-08 | 1.9E-08 | 2.4E-08 | 4.2E-08 |
| LLI Wall | 5.2E-08 | 5.3E-08 | 5.4E-08 | 5.9E-08 | 7.7E-08 |
| Colon | 3.2E-08 | 3.3E-08 | 3.4E-08 | 3.9E-08 | 5.7E-08 |
| Kidneys | 4.9E-08 | 1.0E-07 | 1.4E-07 | 2.2E-07 | 2.6E-07 |
| Liver | 1.1E-09 | 2.6E-09 | 8.7E-09 | 5.1E-08 | 9.8E-08 |
| Muscle | 2.4E-10 | 6.7E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Ovaries | 4.2E-10 | 8.5E-10 | 1.8E-09 | 6.7E-09 | 2.5E-08 |
| Pancreas | 2.4E-10 | 6.7E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Red Marrow | 3.5E-09 | 7.6E-09 | 1.9E-08 | 4.9E-08 | 7.5E-08 |
| Respiratory Tract | | | | | |
| ET Airways | 2.2E-10 | 6.4E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Lungs | 2.2E-10 | 6.5E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Skin | 2.3E-10 | 6.5E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Spleen | 2.4E-10 | 6.6E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Testes | 2.4E-10 | 6.6E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Thymus | 2.2E-10 | 6.5E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Thyroid | 2.2E-10 | 6.4E-10 | 1.6E-09 | 6.5E-09 | 2.5E-08 |
| Uterus | 3.1E-10 | 7.3E-10 | 1.7E-09 | 6.5E-09 | 2.5E-08 |
| Remainder | 7.9E-10 | 1.8E-09 | 3.0E-09 | 8.7E-09 | 2.8E-08 |
| Effective dose | 5.1E-09 | 6.4E-09 | 9.9E-09 | 2.2E-08 | 4.6E-08 |

Table A5. U-235, adult worker. Ingestion: $f_A$ = 0.002

| Time after intake | 7 days | 30 days | 1 year | 10 years | 50 years |
|---|---|---|---|---|---|
| Adrenals | 3.3E-11 | 7.5E-11 | 1.7E-10 | 6.6E-10 | 2.5E-09 |
| Bladder Wall | 9.5E-11 | 1.4E-10 | 2.3E-10 | 7.2E-10 | 2.6E-09 |
| Bone Surface | 3.6E-09 | 7.6E-09 | 1.7E-08 | 4.4E-08 | 7.4E-08 |
| Brain | 2.2E-11 | 6.4E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Breast | 2.3E-11 | 6.6E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| GI-Tract | | | | | |
| Oesophagus | 2.3E-11 | 6.5E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| St Wall | 1.1E-09 | 1.2E-09 | 1.3E-09 | 1.8E-09 | 3.6E-09 |
| SI Wall | 2.9E-09 | 3.0E-09 | 3.1E-09 | 3.6E-09 | 5.4E-09 |
| ULI Wall | 1.7E-08 | 1.8E-08 | 1.8E-08 | 1.8E-08 | 2.0E-08 |
| LLI Wall | 5.3E-08 | 5.3E-08 | 5.3E-08 | 5.4E-08 | 5.6E-08 |
| Colon | 3.3E-08 | 3.3E-08 | 3.3E-08 | 3.4E-08 | 3.5E-08 |
| Kidneys | 4.9E-09 | 1.0E-08 | 1.4E-08 | 2.2E-08 | 2.6E-08 |
| Liver | 1.2E-10 | 2.7E-10 | 8.8E-10 | 5.1E-09 | 9.8E-09 |
| Muscle | 4.4E-11 | 8.7E-11 | 1.8E-10 | 6.7E-10 | 2.5E-09 |
| Ovaries | 2.3E-10 | 2.7E-10 | 3.6E-10 | 8.5E-10 | 2.7E-09 |
| Pancreas | 4.4E-11 | 8.7E-11 | 1.8E-10 | 6.7E-10 | 2.5E-09 |
| Red Marrow | 3.8E-10 | 7.9E-10 | 1.9E-09 | 5.0E-09 | 7.6E-09 |
| Respiratory Tract | | | | | |
| ET Airways | 2.2E-11 | 6.5E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Lungs | 2.4E-11 | 6.7E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Skin | 2.9E-11 | 7.1E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Spleen | 4.0E-11 | 8.2E-11 | 1.7E-10 | 6.6E-10 | 2.5E-09 |
| Testes | 3.9E-11 | 8.2E-11 | 1.7E-10 | 6.6E-10 | 2.5E-09 |
| Thymus | 2.3E-11 | 6.5E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Thyroid | 2.2E-11 | 6.5E-11 | 1.6E-10 | 6.5E-10 | 2.5E-09 |
| Uterus | 1.1E-10 | 1.6E-10 | 2.5E-10 | 7.4E-10 | 2.6E-09 |
| Remainder | 1.5E-10 | 2.5E-10 | 3.8E-10 | 9.5E-10 | 2.8E-09 |
| Effective dose | 4.2E-09 | 4.4E-09 | 4.7E-09 | 6.0E-09 | 8.3E-09 |

**Fig. A27.1. Inhalation Type F: predicted activity of U-234 or U-238 following acute intake of natural U**

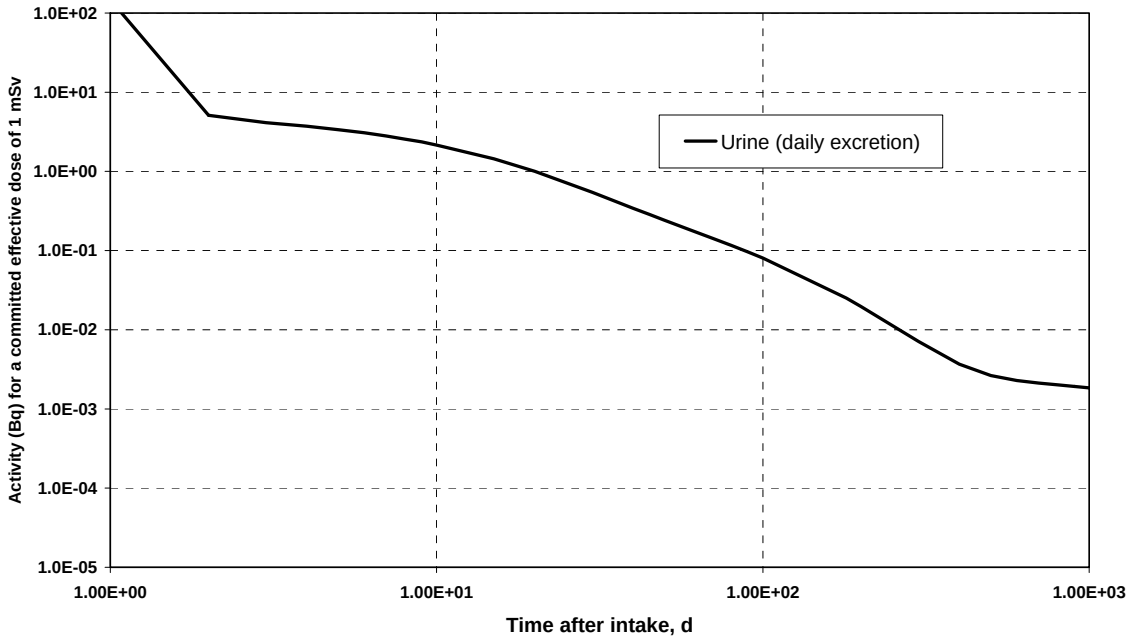



**Fig. A27.2. Inhalation Type M: predicted activity of U-235 in lungs and U-234 or U-238 in urine and faeces following acute intake of natural U**

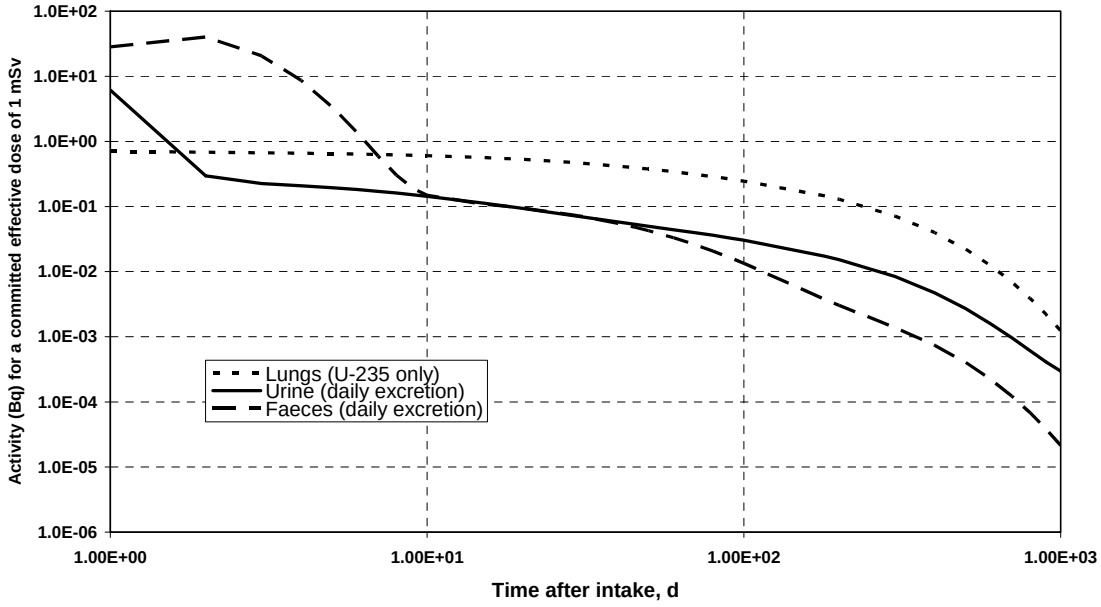


**Fig. A27.3. Inhalation Type S: predicted activity of U-235 in lungs and U-234 or U-238 in urine and faeces following acute intake of natural U**

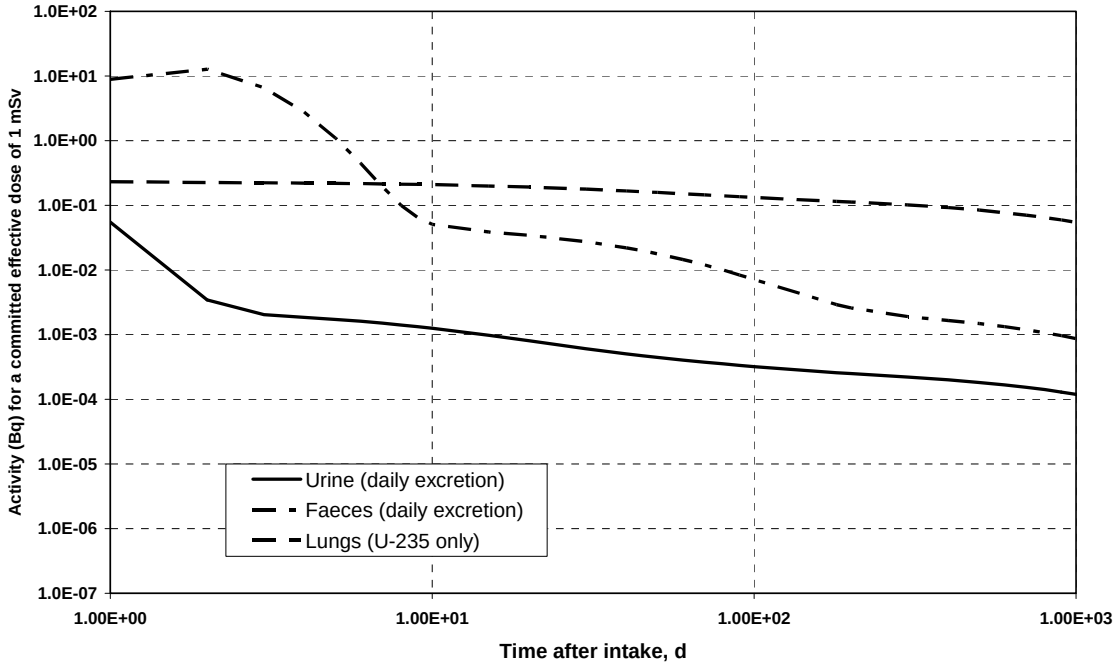



**Fig. A27.4. Inhalation of uranyl nitrate: predicted activity of U-234 or U-238 following acute intake of natural U**

**Fig. A27.5. Inhalation uranium tetrafluoride: predicted activity of U-235 in lungs and U-234 or U-238 in urine and faeces following acute intake of natural U**

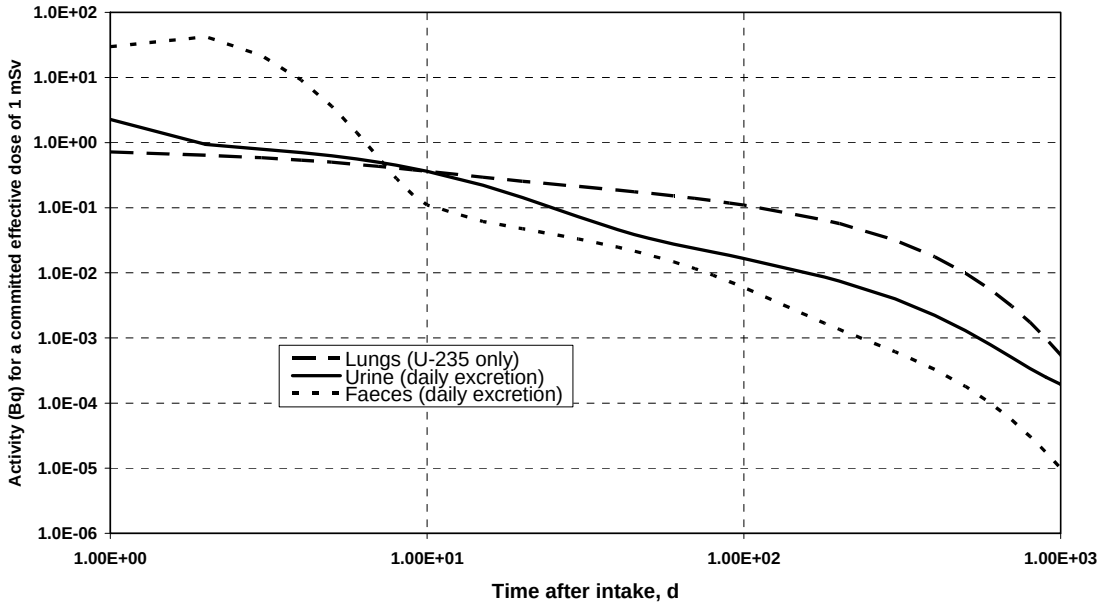



**Fig. A27.6. Inhalation uranium dioxide: predicted activity of U-235 in lungs and U-234 or U-238 in urine and faeces following acute intake of natural U**

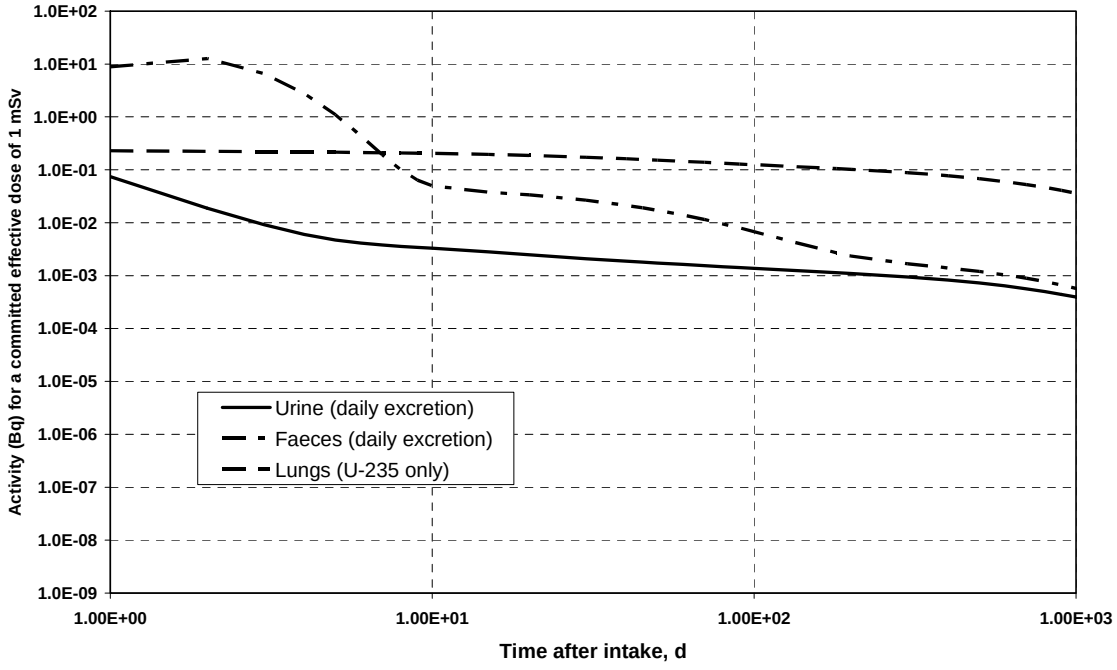


**Fig.A27.7. Ingestion (f1 = 0.02): predicted activity of U-234 or U-238 following acute intake of natural U**

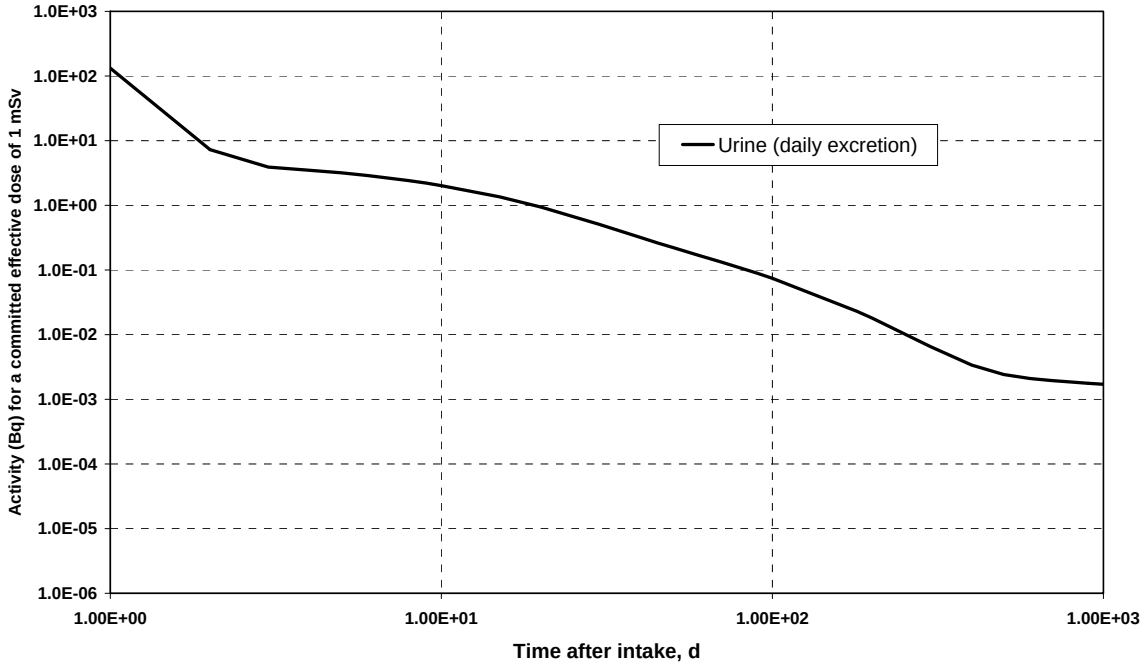



**Fig. A27.8. Ingestion (f1 = 0.002): predicted activity of U-234 or U-238 following acute intake of natural U**