SOURCES AND EFFECTS OF IONIZING RADIATION

United Nations Scientific Committee on the Effects of Atomic Radiation

UNSCEAR 2000 Report to the General Assembly, with Scientific Annexes

VOLUME I: SOURCES

UNITED NATIONS
New York, 2000

NOTE

The report of the Committee without its annexes appears as Official Records of the General Assembly, Fifty-fifth Session, Supplement No. 46 (A/55/46).

The designation employed and the presentation of material in this publication do not imply the expression of any opinion whatsoever on the part of the Secretariat of the United Nations concerning the legal status of any country, territory, city or area, or of its authorities, or concerning the delimitation of its frontiers or boundaries.

The country names used in this document are, in most cases, those that were in use at the time the data were collected or the text prepared. In other cases, however, the names have been updated, where this was possible and appropriate, to reflect political changes.

UNITED NATIONS PUBLICATION
Sales No. E.00.IX.3
ISBN 92-1-142238-8

ANNEX A

Dose assessment methodologies

CONTENTS

INTRODUCTION

1. The estimation of exposures of human populations from the various sources of radiation is an important and continuing goal of the Committee. In its previous assessments, the Committee took many different approaches to dose estimation, depending mainly on the availability of data. These methods have been documented in the UNSCEAR reports. To ensure that the methods are relevant for continued use, the assumptions and parameters must be reviewed from time to time and, if necessary, updated for improved accuracy. The objective of this Annex is to provide such a review of dose estimation procedures.

2. The initial work of the Committee involved evaluating the doses from natural background sources and from the testing of nuclear weapons in the atmosphere. In each case, the starting point of the calculations was where the fewest steps or assumptions would be needed, for example, the concentrations of radionuclides in the body or the deposition of radionuclides on the ground. To evaluate the exposures from nuclear power production, generic models had to be used to estimate the dispersion of radionuclides in the environment, the transfer to humans and the doses from various pathways, since the concentrations or depositions were not measurable at the point of interest. To evaluate the exposures resulting from the Chernobyl accident, some of the dose estimation procedures were modified to account for seasonal and other features indicated by available measurements.

3. In most cases, the Committee has been interested in evaluating the average annual doses from the naturally occurring levels of radionuclides in the environment and from the releases due to man-made practices or events. There has been little need for detailed, time-dependent dose modelling; the use of transfer coefficients or equilibrium modelling has been adequate for purposes of the Committee. Data compilations have been generalized to allow widespread use in both time and space. Although projections were needed to obtain committed doses, there has been little emphasis on prognostic modelling. In general, data-based methods of assessment with more direct and simpler dose estimation procedures have provided results of reliable accuracy and allowed scientists throughout the world to understand and apply or adapt these same methods. This historical viewpoint is significant and important to understand the evaluations of the Committee. In specific circumstances, more theoretical or more detailed models might have been more appropriately considered, but these have generally not been used by the Committee, nor will they be described or used in this Annex.

4. The Committee previously summarized its dose estimation procedures in Annex A, “Concepts and quantities in the assessment of human exposures”, of the UNSCEAR 1977 Report [U7] and in Annex A, “Dose assessment models”, of the UNSCEAR 1982 Report [U6]. These reviews are extended in this Annex with consideration of dose estimation procedures used in all earlier assessments of the Committee. The selection of models and the values of the parameters have been adjusted, based on best available estimates.

5. The procedures and models developed and used by the Committee are believed to be reasonably accurate in general application. They are largely based on empirical evaluations of available measurements. In the widest sense, the estimates of the average doses to the global population from radiation sources are certainly well within the wide

variations that are known to exist. It is clear, however, that more regionally appropriate values of environmental conditions or of human habits apply in specific circumstances. Thus, the calculational procedures described here should be used in other applications only with caution, and site-specific data should be used where appropriate.

I. GENERAL CONSIDERATIONS FOR DOSE ASSESSMENTS

A. DOSIMETRIC QUANTITIES

1. Definitions

6. For radiation assessment purposes, a number of specialized quantities are used. A historical review of the quantities used by the Committee was presented in the UNSCEAR 1988 Report [U4]. The Committee uses the system of radiation quantities and units adopted in 1980 by the International Commission on Radiation Units and Measurements (ICRU) [I8, I12] and the revised terminology and definitions proposed in 1990 by the International Commission on Radiological Protection (ICRP) [I1].

7. For assessments by the Committee, the fundamental dosimetric quantity used is the absorbed dose, D, averaged over a tissue or organ; its unit is joule per kilogram, which is given the special name gray (Gy). The relationship of this quantity to the risk of biological effect is described by the weighted dose quantities. Values of weighting factors have been recommended by ICRP for the various types and energies of radiation incident on the body or emitted from within the body and for selected tissues and organs [I1]. Equivalent dose, H_T, is the averaged absorbed dose in tissue or organ T, modified by the radiation weighting factor, w_R:

$$H_T = \sum_R w_R D_{T,R} \qquad (1)$$

where $D_{T,R}$ is the mean absorbed dose in tissue or organ T due to radiation R. The unit of equivalent dose is joule per kilogram, and it is given the special name sievert (Sv). Values of w_R are given in Table 1.

8. Effective dose, E, is the sum of the weighted equivalent doses in all the tissues and organs of the body. It is calculated from the following expression, where w_T is the weighting factor for tissue T:

$$E = \sum_T w_T \sum_R w_R D_{T,R} \qquad (2)$$

Effective dose has the unit joule per kilogram, which is given the name sievert (Sv). ICRP has selected values of w_T to assess health detriment arising from the irradiation of various organs and tissues. The recommended values of w_T are given in Table 2. The values have been selected for a reference population of equal numbers of both sexes and a wide range of ages. They apply to workers, to the general public and to either sex.

9. The above definition of effective dose replaces a previous similar definition of effective dose equivalent, H_E:

$$H_E = \sum_T w_T(1977) \sum_R w_R D_{T,R} \qquad (3)$$

which was promulgated by ICRP in 1977 [I11]. The difference between H_E and E is in the values of the weighting factors, w_T. In equation (3) this is noted by appending (1977) to the w_T expression. Values of $w_T(1977)$ are also indicated in Table 2. Normally, this now outdated concept would not be used by the Committee, but some very extensive calculations of external dose coefficients have been performed and reported as values of H_E rather than E, and there is no unambiguous way to convert from one value to another without access to the original calculations. It is presumed that eventually these calculations will be repeated so that values of E can be used in those few circumstances where it is not now possible. For high-energy gamma radiation the numerical values of E and H_E should be approximately the same. However, for low-energy gamma radiation, bremsstrahlung, and electrons, the dose to the skin is typically much higher than the dose to any other organ, and the skin was specifically excluded from consideration in H_E. To simulate the value of E where complete recalculation of E from H_E is not possible, the value of 0.01 H_{skin} has been added to H_E. This practice of adding a weighted component of skin dose to H_E was suggested by ICRP [I14] in 1978 and was first used by the Committee in the UNSCEAR 1982 Report [U6] to calculate doses from fission noble gases released from nuclear reactors.

10. The term exposure is often used in the general sense of being exposed to a radiation source, inferring that a dose is received, but it also has a more specific definition. Exposure is the total electrical charge of ions of one sign produced in air by electrons liberated by x or gamma rays per unit mass of irradiated air at NTP. The unit of exposure is coulomb per kilogram. An old unit, the roentgen, R, is still used, as noted, for example, in reporting after the Chernobyl accident. One roentgen is equal to $2.58\ 10^{-4}$ C kg^{-1}. In this sense, the term exposure applies to ionization of air by x or gamma rays, but the more common usage is also prevalent. Another dosimetric quantity is the kerma, which is the initial energy of charged particles liberated by uncharged particles in a unit mass of material. The unit is joule per kilogram, given the name gray (Gy). Under the assumption that charged particle equilibrium exists within the

volume of material, the kerma and absorbed dose may be assumed to be equivalent. This assumption is used by the Committee in most circumstances in specifying absorbed dose rates in air or tissue.

11. When radionuclides are released to the environment, they persist until they are lost through radioactive decay, causing radiation exposures into the future. To compare doses delivered over different time periods, the Committee introduced the concept of the dose commitment. The dose commitment, $H_{c,T}$ or E_c, is defined as the time integral of the average individual dose rate (per caput dose rate) delivered as a result of a specific practice:

$$H_{c,T} = \int_0^\infty \dot{H}_T dt \qquad \text{or} \qquad E_c = \int_0^\infty \dot{E}(t) dt \tag{4}$$

The integral is taken over infinite time to account for exposures occurring during all future time and may thus involve the average individual dose rates over generations. The dose commitment from one year of a practice is numerically equal to the equilibrium dose rate, if the practice continues indefinitely at constant rate. If the integration is carried out only to a specified time, this is then termed a truncated dose commitment.

12. When prolonged exposure to a single individual from a single intake of a radionuclide is being considered, committed dose quantities are used. The time distributions of the absorbed dose rates vary with the radionuclides, their form, mode of intake, and biokinetic behaviour. The committed equivalent dose, $H_T(\tau)$, is defined as the time integral of the equivalent dose rate, where τ is the integration time in years:

$$H_T(\tau) = \int_t^{t_0+\tau} \dot{H}_T(t) dt \tag{5}$$

The value of τ is taken to be 50 years for adults and from time of intake to age 70 years for children. The committed effective dose, $E(\tau)$, is the sum of the committed equivalent doses to tissues and organs multiplied by the appropriate tissue weighting factors, w_T. In general, the Committee considers doses to adults; doses to children are considered only when such doses are significantly different. ICRP has developed age-dependent models for the respiratory and gastrointestinal tract and for the systemic biokinetic behaviour of radionuclides that are of importance in the environment. These models have been used to compute values of committed effective dose per unit intake by members of the public by inhalation and ingestion. These values are compiled in ICRP publications [I2, I3, I4, I5], and general use of these values is made by the Committee.

13. Collective dose quantities have also been used by the Committee. These are aggregate quantities of dose and population size. The collective equivalent dose, S_T, is the average equivalent dose in an exposed group of individuals multiplied by the number of individuals in each group:

$$S_T = \sum_i \overline{H}_{T,i} N_i \tag{6}$$

where N_i is the number of individuals in population subgroup i receiving mean organ equivalent dose $H_{T,i}$. The collective effective dose, S, is defined in a similar manner. The population and the time period over which the dose is determined should be specified. The collective dose commitment may become rather uncertain if applied to very long time periods in which future environmental conditions and the populations affected cannot be reasonably anticipated.

2. Age groupings

14. In many instances, the effective doses in populations have been estimated by the Committee for the adult individual. Data on concentrations of radionuclides in tissues have not always been widely available for other age groups. In some cases, the uncertainties have been as great as the possible differences. For certain radionuclides and pathways, however, the differences may justify separate dose estimates. This is particularly true for ^{131}I. The availability of dose per unit intake estimates for other age groups means that calculated dose estimates can be derived from measured concentrations in foods, and more extensive reporting of age-specific results can be expected in the future.

15. Earlier estimates of doses from fallout ^{131}I were made for infants, using the age of 6 months as representative of the 0–1 year age group [U7, U8]. For releases of ^{131}I from nuclear reactors, parameters were given in the UNSCEAR 1977 Report [U7] for the ages 6 months, 4 years, 14 years and adult. For assessment of exposures from the Chernobyl accident, dose estimates for ^{131}I were made for 1-year-old infants and adults [U4]. In the UNSCEAR 1993 Report [U3], food consumption amounts were indicated for infants, children and adults. In that report age-weighted annual intakes of naturally occurring radionuclides were then derived, assuming the fractional distribution of adults, children and infants in the population to be 0.65, 0.3, and 0.05, respectively. An age-independent dose per unit intake (the adult value) was applied [U3]. Age-dependent dose coefficients are now available from ICRP, and the number of age groups considered could be expanded to six: 3 months (from 0 to 1 years), 1 year (from 1 year to 2 years), 5 years (>2 years to 7 years), 10 years (>7 years to 12 years), 15 years (>12 years to 17 years), and adult. For most purposes, the Committee will consider the age categories of infants, children, and adults and use the available dose coefficients corresponding to 1–2 years, 8–12 years, and >17 years, respectively, for these categories. The fractional distribution of the population within these categories is that mentioned above, namely, 0.05, 0.3, and 0.65 for infants, children and adults, respectively.

B. ENVIRONMENTAL BEHAVIOUR OF RADIONUCLIDES

1. Transfer processes

16. Radionuclides are generally released in trace quantities to the environment. They are then physically transported in the air or water media in which they are located. The measurements of radionuclide transfers from past releases have been used to study and infer large-scale atmospheric and hydrological movements on the earth. The fallout radionuclides ^{90}Sr and ^{137}Cs have been used to infer material removal or renewal times (residence times) in environmental regions. Tritium is a tracer for the world hydrological cycle and ^{14}C for the global carbon cycle. The specific removal or transfer processes of the various exposure pathways have been extensively studied.

17. Radioactive materials, either particles or gases, may be transported great distances by local and large-scale air movements. The time periods that the materials remain airborne depend on the latitude, time of year and height of injection into the atmosphere. The depletion processes include gravitational settlement and dry impaction, incorporation into rain drops and washout by falling precipitation. The physical and chemical characteristics of the materials themselves, such as particle size and chemical and physical forms, may influence the removal rates.

18. The predominant features of large-scale mixing processes and air movements in the atmosphere were presented in the UNSCEAR 1982 Report [U6] in connection with discussion of exposures from nuclear explosions. They were used to describe the occurrence of fallout. The measured deposition of ^{90}Sr could, however, be used as a starting point for the dose assessment, obviating the need to evaluate the deposition from the uncertain input amounts. With improved estimates recently available of the input of fission radionuclides to the atmosphere from nuclear tests, quantitative aspects of the general model can be pursued with seasonal values of residence times assigned to the various compartmental regions and latitudinal deposition estimated. This exercise is discussed in Annex C, "*Exposures to the public from man-made sources of radiation*".

19. Releases of radionuclides from nuclear fuel cycle installations occur at ground level or through stacks of assumed representative heights of 30 or 100 m. The long-term, sector-averaged Gaussian plume model can be used to calculate air concentration for limited distances following airborne releases. Estimates can be obtained directly from the model or from a simple analytical expression that gives a good fit to the model results. The air concentration at one kilometre per unit release is typically 5 10^{-7} s m^{-3} and decreases as a result of further dispersion at a rate inversely proportional to distance, expressed in kilometres, raised to the power 1.2–1.4. Derivation of these quantities is discussed in Section I.B.3. Integration to 50 or 100 km defines the local exposures. Further integration to a distance of 2,000 km defines the continental or regional component of exposure. Most particles from near-surface releases are deposited within this distance. Only fine aerosols and gases may become further dispersed in the troposphere.

20. Global modelling of atmospheric releases will be described with respect to the specific radionuclides. Mixing occurs first within the latitude band, then within the hemisphere. Gradual interhemispheric exchange occurs for gases such as ^{85}Kr, for which removal processes are minimal. Tritium and ^{14}C enter the global cycles of the respective elements.

21. Radioactive material released to the aquatic environment is transported and dispersed by advective and turbulent processes occurring in the water body. Interactions of radionuclides with suspended matter and sediments may remove radionuclides from the solution. Methods for modelling hydrological transport have been developed and applied, usually for specific categories of water bodies: lakes, rivers, estuaries, coastal seas and oceans.

22. UNSCEAR has needed hydrological transport estimates to evaluate the exposures from releases of radionuclides from nuclear fuel cycle installations. For fuel reprocessing plants, use has been made of dispersion estimates surrounding the plants at Sellafield and La Hague. In the general case for reactor releases, the Committee made use of relationships between water volumes, water usage and potential intake to estimate collective doses. The water uses considered included drinking water, fish and seafood production and irrigation. Some minor pathways might be involved in the local regions, such as immersion and exposure to shoreline contaminants. Some general considerations with regard to aquatic models and suggestions about which models to use have been published, e.g. [S2]. Details of the procedures used by UNSCEAR will be presented later in connection with ingestion exposures.

2. Parameters for dose estimation

23. The basic parameters used in models to describe environmental behaviour and transport of radionuclides and to make dosimetric calculations are transfer coefficients, P_{ij}. These describe the relationships of integrated concentrations or dose in successive environmental compartments, e.g. movement from compartment i to compartment j. The pathways of transfer of radionuclides through the environment commonly evaluated in UNSCEAR dose assessments are illustrated in Figure I along with designations of the transfer coefficients. As an example, P_{34} is the time-integrated activity concentration of a radionuclide in the body divided by the time-integrated concentration of the same radionuclide in the diet. This methodology for deriving relationships between measured quantities has been used by the Committee since 1962.

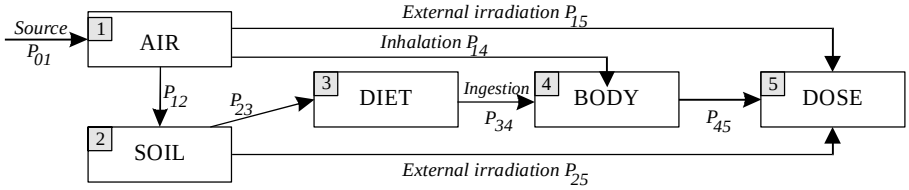

Figure I. Terrestrial pathways of transfer of radionuclides and dose to humans.

24. For a particular environmental transfer pathway, the amount of radionuclide released to the environment multiplied by the intervening transfer coefficients gives an estimate of the resulting effective dose. If measurement results are available at any point in the chain, the calculation may begin at that point. This minimizes the uncertainties that may exist in determining transfer coefficients for earlier steps in the transfer pathway. Thus, assessments of dose derived by the Committee have started with integrated concentrations of radionuclides in air, deposition densities, measured concentrations in foods or body burdens.

25. The measurements used to evaluate transfer coefficients have been made over a number of years by research and monitoring organizations in many locations. The transfer coefficients derived for estimation of effective doses from atmospheric nuclear testing were summarized in the UNSCEAR 1993 Report [U3] for a long listing of radionuclides for the pathways of external irradiation, inhalation and ingestion. The many measurement results acquired following the Chernobyl accident have shown that more seasonal or locally characteristic conditions should be taken into account in evaluating exposures from specific single releases of radioactive materials. Several programmes to compare results and validate models were instigated following the Chernobyl accident. Those activities are contributing results useful for deriving specific values for many transfer coefficients, e.g. [I16, I17].

26. Tritium and ^{14}C are modelled differently than other radionuclides, since they are mobile in the environment and are readily incorporated into living organisms. The transfer of tritium and ^{14}C is not modelled using transfer parameters but by a specific-activity approach. For tritium, it is assumed that the tritium to hydrogen atom ratio in the various environmental compartments is simply proportional to the ratio in moisture in air. For carbon, the ^{14}C activity per gram carbon in all compartments is assumed to be the same as that in air.

27. The specific methods used by the Committee to estimate doses to humans caused by releases of radioactive materials to the environment are described in the following Chapters. The rationale for the selection of the various parameters is presented, so that it will be clear when alternative selections might be desirable for specific local conditions. The methods are intended to be widely applicable, and since they are largely empirically based, they should provide realistic estimates of doses in most general circumstances of release of radionuclides.

3. Atmospheric dispersion from a near-surface release

28. Radionuclide concentrations in the environment downwind of an isolated source such as a nuclear reactor are usually undetectable at distances greater than a few kilometres. In such cases, the air concentrations needed as the starting point for dose assessments to the public must be estimated using a mathematical model.

29. Average air concentrations close to a specific source are traditionally calculated using the long-term sector-averaged Gaussian plume model [I15]. In this model, the plume is assumed to spread uniformly across a sector subtended by an angle $\Delta\theta$ (usually chosen to be 30°). Air concentrations at a given distance downwind are calculated for each of six atmospheric stability classes using average values of wind speed, inversion height and vertical dispersion parameter for each class. The long-term mean concentration is found by summing over classes, taking into account the frequency of occurrence of each class and the frequency with which the wind blows towards the site of interest. The model is able to account for reductions in air concentration due to wet and dry deposition. A general discussion of the processes governing atmospheric dispersion was presented in the UNSCEAR 1982 Report [U6].

30. The mathematical statement of the long-term sector-averaged Gaussian plume model is as follows:

$$C_{a,j} = \left(\frac{2}{\pi}\right)^{1/2} \frac{f_j Q D_w}{x \Delta\theta} \sum_{i=1}^{6} \frac{f_i F(\sigma_{z,i}, H, h_i) \exp(-\lambda x u_i^{-1}) D_{d,i}}{(u_i \sigma_{z,i})} \quad (7)$$

where $C_{a,j}$ is the long-term average air concentration (Bq m^{-3}) in sector j; f_j is the frequency with which the wind blows into sector j; Q is the release rate (Bq s^{-1}); x is the downwind distance (m); $\Delta\theta$ is the sector width (radians); f_i is the frequency of occurrence of stability class i; $F(\sigma_{zi},H,h_i)$ is the vertical shape function; $\sigma_{z,i}$ is the vertical dispersion parameter for stability class i (m); H is the effective release height (m); h_i is the mixed layer height for stability class i (m); λ is the radioactive decay constant for the radionuclide in question (s^{-1}); $D_{d,i}$ is the depletion factor for dry deposition; D_w is the depletion factor for wet

deposition; and u_i is the average wind speed for stability class i at the release height (m s^{-1}).

31. For ground-level concentrations, the vertical shape function is given by

$$F(\sigma_{z,i}, H, h_i) = \sum_{n=-\infty}^{\infty} \exp\left[-\frac{(H+2n\,h_i)^2}{2\sigma_{z,i}^2}\right] \qquad (8)$$

which accounts for reflection from the ground and from an elevated inversion through the method of virtual sources. The summation index n in equation (8) represents the number of reflections that the plume has undergone. The summation converges slowly in some applications. To simplify the calculation, Yamartino [Y1] proposed approximations as follows: for $\sigma_{z,i} / h_i \leq 0.63$, truncate equation (8) at n = 0, ± 1; for $0.63 < \sigma_{z,i}/h_i \leq 1.08$, $F(\sigma_{z,i}, H,h_i) = (2\pi)^{½}\, \sigma_{z,i}/h_i\, (1 - k^2)[1 + k^2 + 2k \cos(\pi H/h_i)]$ where $k = \exp[-½\, (\pi\, \sigma_{z,i}/h_i)^2]$; for $\sigma_{z,i}/h_i > 1.08$, $F(\sigma_{z,i}, H,h_i) = (2\pi)^{½}\, \sigma_{z,i}/h_i$. These approximations result in minimal error in evaluation of equation (8).

32. Plume depletion due to dry deposition is normally treated using the source depletion method, in which case the depletion factor takes the form

$$D_{d,i} = \exp\left[-\alpha_i \int_0^x \frac{\exp\left(-\frac{H^2}{2\sigma_{z,i}^2(x)}\right)}{\sigma_{z,i}(x)}\, dx\right] \qquad (9)$$

where $\alpha_i = (2/\pi)^{½}\, v_d/u_i$ and v_d is the dry deposition velocity (m s^{-1}). The depletion factor for wet deposition is given by $D_w = \exp(-\Lambda t_s)$, where Λ is the washout coefficient (s^{-1}) and t_s (s) is the time over which precipitation occurs during the travel of the plume from source to receptor.

33. A number of investigators [B13, P6, V1] have suggested forms for the vertical dispersion parameter. The following scheme of Smith [S1] and Hosker [H8] is used, since it is able to take account of the surface roughness, z_0(m), of the site:

$$\sigma_z = g(x)\, F(x, z_0) \qquad (10)$$

where $g(x) = ax^b/(1+cx^d)$ and $F(x,z_0) = \ln[px^q[1+(rx^s)^{-1}]]$ when $z_0 > 0.1$ m and $F(x,z_0) = \ln[px^q(1+rx^s)^{-1}]$ when $z_0 \leq 0.1$ m. The parameters a, b, c, and d depend on the atmospheric stability class, and the parameters p, q, r, and s depend on the surface roughness. Representative values are given in Table 3.

34. Equations (7) to (10) provide a relatively simple method for calculating long-term average air concentrations due to a specific source. Wherever possible, site-specific values should be used for the meteorological and release parameters appearing in the equations. In the absence of site-specific data, the representative values listed in Table 4 give reasonable estimates of air concentrations. Values of λ, v_d and Λ should be chosen for the radionuclide of interest.

35. One aim of applying the above method is to derive long-term average dilution factors, C_a/Q, for downwind distances between 1 and 2,000 km from the source. The results of the calculation are given in Table 5. A long-lived radionuclide was assumed so that radiological decay could be neglected. The parameter values in Table 4 were used and the deposition velocity v_d and washout coefficient Λ were set to representative values of 0.002 m s^{-1} and 0.0001 s^{-1}, respectively. Precipitation was assumed to occur 500 hours per year, 80% of the time during class D conditions and 20% during class C, at an average rate of 1.5 mm h^{-1}. The washout time t_s was assumed to be equal to the travel time t_t between source and receptor for $t_t < 4$ hours and equal to $t_t/2$ for $t_t > 24$ hours; in the range $4 < t_t < 24$, t_s was assumed to vary linearly between t_t and $t_t/2$.

36. The variation of air concentration with downwind distance beyond 1 km can be approximated by the following simple function, which was used in previous UNSCEAR assessments:

$$C_a(x) = D_1 Q\, x^{-n} \qquad (11)$$

where D_1 is the dilution factor at 1 km (s m^{-3}) and x is the downwind distance (km). Figure II shows that equation (11) gives a very good representation of the detailed results of Table 5 and can therefore be used to estimate air concentrations in place of equations (7) to (10) if, for example, site-specific data are not available. The best approximation to the calculated results is obtained with values for D_1 and n of 5.3 10^{-7} s m^{-3} and 1.42, respectively. The value for n is similar to the value of 1.5 used in previous UNSCEAR assessments. The value for D_1 is lower by a factor of 6 than the value of 3 10^{-6} s m^{-3} suggested in the UNSCEAR 1982 Report [U6]; this value reflects concentrations at a location toward which the wind blows about 50% of the time, whereas the currently recommended value of 5 10^{-7} s m^{-3} assumes a uniform wind rose at the point of release.

37. The variability in calculated results has been investigated by altering the parameter values used in equations (7) to (10). The meteorological parameters were varied to cover the range of conditions that could occur from time to time. The variability in deposition velocity and washout coefficient reflects the values associated with different radionuclides. Each parameter was varied in turn, holding all other parameters at the values given above. Results are shown in Table 6 in terms of D_1 and n, the parameters required to implement equation (11). D_1 is relatively insensitive to changes in the values of the parameters except for wind speed and release height; n is sensitive to these parameters, as well as to deposition

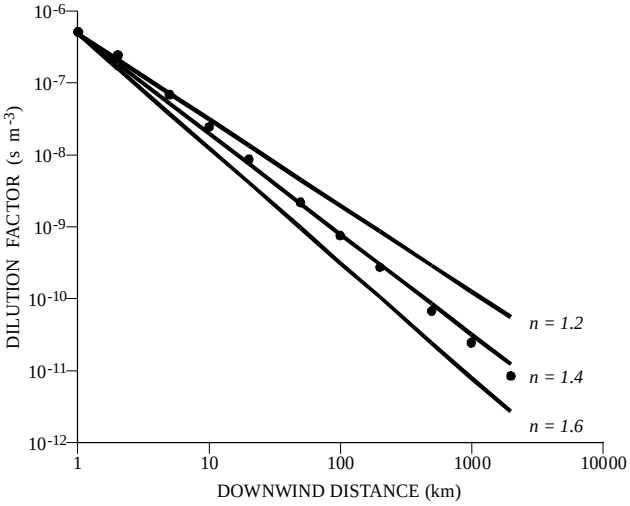

Figure II. Dilution factor for estimation of air concentrations at distances from a surface release determined from Gaussian plume model calculation (points) compared to power-function representations of the form $5\ 10^{-7}\ r^{-n}$.

velocity and inversion height. Release heights can vary from low-level building vents to stacks of 100 m or more. Building entrainment may reduce the effective release height to some extent. The representative height of 30 m has been retained as in earlier UNSCEAR assessments for estimating collective doses following releases from nuclear installations. Individual dose evaluations could depend more critically on the release height assumption. As an example, calculations indicate that for release from a stack of 150 m height, the collective dose in the local and regional area per unit release would be about 25% of that estimated for a 30 m stack. The model is not very sensitive to roughness length, washout coefficient, or the frequency of stability classes. Thus, adequate estimations can be made using the representative values suggested for these parameters without the need for site-specific data. Although wet deposition is very effective at removing material from the plume, precipitation occurs less than 10% of the time and has little effect on long-term average air concentrations. However, reliable values of the washout coefficient are needed to calculate accurately the amount of material deposited on the ground and on vegetation by wet deposition.

38. The results in Table 6 can be used to interpolate the values of D_1 and n that most closely represent meteorological conditions at the site and the radionuclides of interest. Equation (11) can then be used to estimate air concentrations at the downwind distance in question. For noble gases, which do not deposit, a value of n equal to about 1.2 should be used as long as other parameter values remain near the representative values defined here. Tritium should also be assigned a value of 1.2, since most tritium deposited under dry conditions is quickly re-emitted to the atmosphere. Carbon-14 is efficiently deposited and partially returned to the atmosphere through plant and soil respiration. On balance, it is recommended that the index value of 1.4 be used for this radionuclide. For calculation of radionuclide concentrations at a specific site, values of D_1 taken from Table 6 should be modified to reflect the frequency with which the wind blows towards the location of interest. For the purposes of calculating representative population doses using the method presented in this Annex, a uniform wind rose was assumed, with a frequency of 1/12, or 0.083, averaging over 12 sectors.

39. The long-term, sector-averaged Gaussian plume model has been extensively tested at local distances. When used with site-specific meteorological data, the uncertainties in its predictions are less than a factor of 2 within 10 km of the source and less than a factor of 4 between 10 and 100 km of the source [C10, H6, R4]. Use of the model is therefore adequate for local assessments. Validation of the model on regional scales is more difficult. Few point sources are strong enough or emit a unique enough contaminant to be detected unambiguously at downwind distances greater than 100 km. Thus few data from routine releases can be used to test the model. Regional-scale tracer studies have been carried out, but only over short periods of time. These must be considered case studies that provide information only for the meteorological conditions prevailing at the time of the release. They cannot be used to infer long-term average air concentrations.

40. The problem of acid precipitation has driven the development of a number of models that simulate the long-range transport of air pollutants [J3]. These models are much more sophisticated than the Gaussian model described above in their treatment of plume transport and deposition and can track pollutants through space- and time-varying meteorological conditions. They are moderately successful in predicting the broad features of the concentration field on regional scales. However, they require considerable expertise, computer resources, and input data to run and are therefore unsuited to the types of assessments performed by the Committee. Comparisons of their predictions with those of the Gaussian model would help to establish the validity of the latter, but such studies have not yet been done.

41. Although the accuracy of the predictions of the Gaussian model beyond 100 km is difficult to quantify, a number of factors suggest that the model overestimates true concentrations at this range:

(a) The model assumes that the plume travels in a straight line from source to receptor. In reality, variations in wind direction will generally lead to quite complicated trajectories that increase the travel time between source and receptor and provide the opportunity for enhanced mixing.
(b) The model assumes that the stability class in effect at the start of the release remains in effect until the plume reaches the receptor. In reality, a plume travelling over hundreds or thousands of kilometres

will experience several diurnal cycles and a full range of atmospheric stabilities. A plume that has undergone one or more unstable periods will be mixed through a deep vertical layer. It will remain well mixed through subsequent night-time stable periods and not be confined beneath a low capping inversion, as is assumed in the model;

(c) The model assumes that the plume is transported with the wind speed at the effective release height. In reality, as the plume mixes to greater heights, the effective transport velocity must be averaged over deeper layers. Since wind speeds generally increase away from the ground, the result is extra dilution and lower concentrations;

(d) The model assumes that the terrain over which the plume passes is flat, a supposition unlikely to hold over regional distances. Complex topography will tend to increase turbulence levels and deflect the plume trajectory, thereby reducing concentrations.

42. In contrast to the model features mentioned above, the procedure for estimating plume depletion due to dry deposition is not conservative. Material is assumed to be lost uniformly over the entire depth of the plume when in reality it is lost only at the deposition surface. Horst [H7] showed that this approach underestimates airborne concentrations by an amount that increases with increasing atmospheric stability, greater downwind distances, and larger deposition velocities.

43. Thus, the accuracy of the Gaussian model at regional scales is unknown, and uncertainties are large, but probably within a factor of 10 for relatively simple situations. The uncertainties would be somewhat smaller for population doses since the concentration averaged over all distances and directions is probably better known than the concentration at a point.

44. In summary, the value of the dilution factor, D_1, of 5 10^{-7} s m^{-3} is assumed by the Committee to be representative for evaluating collective doses per unit release when site-specific data are not available. The value is not very sensitive to variations in meteorological or deposition parameters. The release height can be of greater influence and, if known to be different from the representative value of 30 m, should be taken into account. The index parameter, n, is more variable than the dilution factor with respect to meteorological and deposition conditions, but a value of 1.2 for noble gases and tritium and 1.4 for other radionuclides should provide reasonable estimates of air concentrations.

II. EXTERNAL IRRADIATION

45. External irradiation from radionuclides naturally present in the environment or released from man-made practices or events is usually an important component of the exposure of human populations. These exposures derive primarily from gamma radiation arising from the decay of these radionuclides at locations outside the human body. Secondarily, exposures to the skin from beta radiation may be considered. The methods used by the Committee to estimate external exposures from the various sources are reviewed in this Chapter.

A. COSMIC RAYS

46. Cosmic rays originate in outer space; they consist primarily of protons and alpha particles. Interactions in the upper layers of the earth's atmosphere create secondary components; the more important secondary particles from a dose-assessment view are muons, neutrons, electrons, positrons, and photons. Exposure to cosmic rays is strongly dependent on altitude and weakly dependent on latitude. Dose assessments are based on both measurements and calculations of the radiation transport to infer the dependence on altitude. At lower levels of the atmosphere and at sea level, the dependence on the 11−year solar cycle is small compared to the uncertainty in the estimates and is currently ignored.

47. The method used by the Committee to assess doses from the photon and directly ionizing component of cosmic radiation at sea level has not changed substantially in many years. In the UNSCEAR 1977 Report [U7] the basic value was considered to be the ion-pair production rate, for which a value of 2.1 cm^{-3} s^{-1} was adopted. This value was converted to a dose rate of 32 nGy h^{-1} and has been assumed to be numerically equal to the effective dose rate [U3, U4]. A mean shielding factor of 0.8 has been applied to derive an indoor effective dose rate of 26 nSv h^{-1}. With the further assumption that the average fraction of time spent indoors is 0.8 [U3, U4], the annual effective dose from the ionizing component of cosmic rays at sea level is judged to be 240 μSv. Estimates of cosmic ray dose rates at elevations above sea level are obtained using a procedure published by Bouville and Lowder [B12]:

$$\dot{E}_I(z) = \dot{E}_I(0)\,[0.21\, e^{-1.649z} + 0.79\, e^{0.4528z}] \qquad (12)$$

where $\dot{E}_I(0)$ is the dose rate at sea level, 240 μSv a^{-1}, and z is the altitude in km. The dose rate from the photon and ionizing component is known to vary with latitude, but the variation is small. The dose rate is about 10% lower at the geomagnetic equator than at high latitudes.

48. For the neutron component of the cosmic radiation exposure, the radiation field and the estimates of effective dose have been more uncertain owing to a lack of measurements. Recent measurements and calculations are beginning to provide clarification. Because earlier instrumentation had a low response to high-energy neutrons, which are an important component of the spectrum, some increases in the fluence rate and effective dose are being suggested. Measurements made using a Bonner sphere spectrometer [R3, S8] at the top of the Zugspitze mountain in Germany (altitude 2,963 m, atmospheric depth 718 g cm^{-2}) and associated calculations give a fluence rate of 0.126 ± 0.01 cm^{-2} s^{-1} [S9]. Attenuation with altitude was described using the function $e^{-0.00721p}$, where p (g cm^{-2}) is the atmospheric depth. From this, a fluence rate at sea level (p = 1,033 g cm^{-2}) of 0.013 ± 0.001 cm^{-2} s^{-1} can be derived. Measurements also with Bonner sphere spectrometers gave a value of 0.0133 ± 0.001 cm^{-2} s^{-1} at about sea level for a geomagnetic latitude of 53°N near Braunschweig in Germany [A6], and a value of 0.0123 cm^{-2} s^{-1} at sea level for a geomagnetic latitude of 45°N in Hampton, Virginia in the United States [G3]. The effective dose (isotropic) corresponding to a fluence rate of 0.013 cm^{-2} s^{-1} obtained by applying a neutron fluence energy distribution weighting factor of 200 pSv cm^2 [S9] (equal to 720 nSv h^{-1} per neutron cm^{-2} s^{-1}) is 9 nSv h^{-1}. Birattari et al. [B14], using an extended range remmeter, reported a value of 9 nSv h^{-1} (±5%) in agreement.

49. The shape of the neutron energy spectrum at habitable altitudes is considered to be relatively invariant, and therefore the fluence to effective dose (isotropic) conversion coefficient is expected to be generally valid. On this basis, the annual effective dose rate from neutrons at sea level would be estimated to be 80 μSv a^{-1}. This is substantially larger than the value of 30 μSv a^{-1} used in the UNSCEAR 1993 Report [U3] and is still subject to great uncertainty; the main factor in the increase in the calculated dose is the inclusion of high-energy neutrons. With the application of a shielding factor of 0.8 and an occupancy factor of 0.8, the annual average effective dose at sea level is estimated to be 65 μSv at geographic latitudes between about 40° and 50°.

50. For calculations of outdoor cosmic ray neutron dose rates at other altitudes, the relation between height, h_v, in km, above sea level and atmospheric depth [R3] is, for p > 230 g cm^{-2},

$$h_v = 44.34 - 11.86\, p^{0.19} \qquad (13)$$

Both altitude and latitude variations in the cosmic ray neutron dose rate must be known to determine the population-weighted average exposure of the world population. Calculations of dose from cosmic rays to airline crews and passengers are based on measurements and on detailed calculations using radiation-transport codes tailored to follow the altitude and latitude of a particular flight.

51. The fluence of neutrons, which arise from collisions of high-energy protons within the upper atmosphere, is strongly influenced by geomagnetic latitude. This variation at habitable altitudes has not been satisfactorily quantified, as measurements at different latitudes have not always been comparable. Recent measurements at high altitudes have shown a variation by a factor of about 4 [G3], with the lower values near the equator. These results support the calculations of Florek et al. [F3], who used the Los Alamos Lahet Code System (LCS) to simulate neutron fluence as a function of latitude. Their results are expressed in terms of k_φ, a latitude coefficient, as follows:

$$\dot{E}_N(\text{lat}) = \dot{E}_N(90)\, k_\phi(\text{lat}) \qquad (14)$$

with k_φ ranging from 1.0 at 90° to 0.8 at 47°, 0.6 at 42°, 0.4 at 35° and 0.2 at the equator. The application of this relationship to available measurement results is discussed in Annex B, *"Exposures from natural radiation sources"*.

B. NATURALLY OCCURRING RADIONUCLIDES

1. Exposure processes

52. Many radionuclides occur naturally in terrestrial soils and rocks and in building materials derived from them. Upon decay, these radionuclides produce an external radiation field to which all human beings are exposed. In terms of dose, the principal primordial (half-lives comparable to the age of the earth) radionuclides are ^{40}K, ^{232}Th, and ^{238}U. Both ^{232}Th and ^{238}U head series of radionuclides that produce significant human exposures. The two series are listed and discussed fully in Annex B, *"Exposures from natural radiation sources"*.

53. The decay of naturally occurring radionuclides in soil produces a gamma-beta radiation field in soil that also crosses the soil–air interface to produce exposures to humans. The main factors that determine the exposure rate to a particular individual are the concentrations of radionuclides in the soil, the time spent outdoors, and the shielding by buildings. However, as the materials of which most buildings are built also contain radionuclides, the shielding by buildings of the outdoor radiation field is often more than offset by the presence of additional radionuclides in the building materials.

2. Methods for estimating exposures

54. Two methods of evaluating external exposures from naturally occurring radionuclides have been used by the Committee. The first is simply to summarize directly measured external gamma dose rates in air outdoors and indoors, subtracting the dose rate due to cosmic rays. The second is to calculate the external gamma dose rates in air from measurements of the concentrations of the relevant radionuclides in soil. The two methods have provided generally consistent estimates of exposure.

55. Surveys with direct measurements of dose rate in air from naturally occurring terrestrial radionuclides have been made in most inhabited regions of the world. In the UNSCEAR 1993 Report [U3], data were included for countries or regions in which three fifths of the world population resides. Country average dose rates ranged from 24 to 160 nGy h^{-1}, with a population-weighted average of 57 nGy h^{-1}. The population-weighted average derived from this large sample was assumed to provide a representative global value of outdoor external exposure.

56. Surveys to determine the concentrations of radionuclides in soil have also been made. These results can be related to exposures by using estimates of the dose rates in air per unit concentration of radionuclide in soil. The Committee has relied on the calculations of Beck [B8] for many years. Extensive Monte Carlo calculations of kerma in air and of organ dose for terrestrial gamma rays have been reported by Petoussi et al. [P4], Saito et al. [S5], and Eckerman and Ryman [E7]. Results from three separate calculations are included in Table 7; the values are quite similar and can be considered equal. Uncertainty in the assumed average composition of soil could lead to differences of greater magnitude [E7].

57. Absorbed dose rates in air indoors have also been extensively measured. The values reported in the UNSCEAR 1993 Report [U3] covered areas in which over a third of the world population lives. Country averages ranged from 20 to 190 nGy h^{-1}, with a population-weighted average of about 80 nGy h^{-1}. The population-weighted average of the ratio of indoor to outdoor dose was 1.4. Some of the outdoor measurements may have been influenced by the presence of buildings nearby. The value of the indoor–to–outdoor ratio is very sensitive to the structural properties of buildings (materials and thickness).The building materials act as sources of radiation and also as shields against outdoor radiation. In wooden and lightweight houses, the source effect is negligible, and the walls are an inefficient shield against the outdoor sources of radiation, so that the absorbed dose rate in air could be expected to be somewhat lower indoors than outdoors. In contrast, in massive houses made of brick, concrete or stone, the gamma rays emitted outdoors are efficiently absorbed by the walls, and the indoor absorbed dose rate depends mainly on the activity concentrations of natural radionuclides in the building materials. Under these circumstances, the indoor absorbed dose rate is generally higher as a result of the change in source geometry, with the indoor–outdoor ratio of absorbed dose rates in air between 1 and 2.

58. The Committee has used a coefficient of 0.7 Sv Gy^{-1} to convert absorbed dose in air to effective dose equivalent and effective dose. This result was based on an analysis in the UNSCEAR 1982 Report [U6], and more recent calculations have confirmed the validity of this value for adults. However, newer calculations [P5, S11] using Monte Carlo radiation-transport codes indicate that higher values should be used for infants and children. These values, given in Table 8 for average energies of gamma rays, are 0.9 Sv Gy^{-1} for infants and 0.8 Sv Gy^{-1} for children.

59. In order to combine indoor and outdoor dose rates to compute total doses, the Committee continues to use an indoor occupancy factor of 0.8, which implies that people spend 20% of the time outdoors, on average, around the world. The estimated 80% of time spent indoors is considered likely to be low for industrialized countries in temperate climates and high for agricultural countries in warm climates.

C. RADIONUCLIDES IN AIR: CLOUD SHINE AND IMMERSION EXPOSURE

1. Exposure processes

60. Following the release of radionuclides to the atmosphere and before their deposition onto the ground, human beings may receive external exposure. Two situations are usually distinguished: external exposure from the cloud passing overhead (referred to as “cloud shine”) and external exposure from radionuclides in air surrounding the human body (referred to as “immersion”). The radiation dose from immersion is nearly always much larger than that from cloud shine. The dose from immersion can be readily calculated from the measured, integrated concentrations of radionuclides in air. The dose from cloud shine is rarely calculated; its importance would be significant only if other exposure pathways were absent. One such example would be for persons underneath an elevated, passing plume.

61. Effective doses from immersion are typically calculated for gamma-emitting radionuclides, but beta and even alpha particles can also produce external doses to the skin. Some radionuclides, notably ^{85}Kr, which emits a weak beta particle, produce nearly all of their dose via the pathway of immersion.

2. Methods for estimating exposures

62. Because of their relative insignificance, the Committee has seldom considered external exposures from cloud shine or immersion. Exceptions were made for the Chernobyl accident and for the release of noble gases from reactor operations. Since initial estimates of such exposures were made, tissue-weighting factors and terminology to describe equivalent and effective doses have changed [I1], and newer calculations of dose rates from immersion have been published [E7]. The net changes in the calculated numbers appear to be small.

(a) Atmospheric nuclear testing

63. Although the potential pathways of cloud shine and immersion were considered in the first report of the

Committee, the UNSCEAR 1958 Report [U13], the doses from these pathways for radionuclides released from explosions of nuclear weapons have not been evaluated. The conclusion was reached that, except at the immediate site of the explosion, external irradiation from airborne material is negligible in comparison with external irradiation from fission products deposited on the ground. As much of the material from nuclear explosions was injected into the stratosphere or high troposphere, most of the short-lived radionuclides potentially responsible for the majority of dose from cloud shine or immersion would have decayed before reaching the earth's surface.

(b) The Chernobyl accident

64. Doses from "external irradiation during cloud passage" were calculated for the releases of radionuclides from the Chernobyl accident [U4]. Although exposure rates could in theory be measured directly, in practice it is generally impossible to distinguish this smaller component from radiation arising from material deposited on the ground. Doses can, however, easily be calculated from measured air concentrations or inferred from measured deposition densities.

65. The cloud-gamma dose for radionuclide i is evaluated from the formula

$$E_i = C^*_{a,i}\, d_{c,i}(1-F_0) + C^*_{a,i}\, d_{c,i}\, F_0\, F_s \qquad (15)$$

where E is the effective dose (Sv) from external radiation during cloud passage; C_a^* is the integrated concentration in outdoor air (Bq d m^{-3}); d_c is the effective dose coefficient per unit integrated air concentration (Sv per Bq d m^{-3}); F_0 is the indoor occupancy factor (the fractional time spent indoors); and F_s is the building shielding factor (the ratio of indoor to outdoor dose rate).

66. The first term in equation (15) is the component received while the individual is outdoors, and the second term is the component received indoors. At the time of the Chernobyl assessment, values from Kocher [K7] were used; these values were for $H_E + 0.01H_{skin}$ rather than E. The values used then and the newer recommended values of $H_E + 0.01H_{skin}$ from Eckerman and Ryman [E7] are listed in Table 9.

67. For the Chernobyl assessment, an indoor occupancy factor of 0.8 and a building shielding factor of 0.2 were used for all countries. The values of these factors had been used previously by the Committee [U6, U7]. It was noted, however, that measurements as well as calculations of the shielding factor afforded by buildings showed a large variation, depending on the type of building [C8, M6, S6, U4].

68. To make the above calculation, it is necessary to know the integrated concentration in air of the many short-lived radionuclides. In some countries, complete data were available. In others, data for only one or a few radionuclides were available. In the latter case concentrations of other radionuclides were inferred from ratios measured in nearby countries. In some cases, no measured air concentrations were available, so the integrated air concentration of ^{137}Cs was inferred from its ground-deposition density and a nominal quotient of ground deposition to integrated air concentration of 1,000 m d^{-1} [U4]; the integrated air concentrations of other radionuclides were then inferred from the ratios to ^{137}Cs measured at other locations.

(c) Nuclear installations

69. During the operation of nuclear reactors, several fission noble gases are released, as is the activation radionuclide ^{41}Ar. Among the more prominent fission noble gases are ^{133}Xe from pressurized water reactors and ^{85}Kr, ^{87}Kr, ^{88}Kr, ^{133}Xe, ^{135m}Xe, ^{135}Xe and ^{138}Xe from boiling water reactors [U6]. Much of the dose from these (and other) radionuclides is delivered by the pathway of cloud shine and immersion. Later reports [U3, U4] of the Committee refer to the models developed in the UNSCEAR 1982 Report [U6]. Thus, while the absolute amounts and the relative mixture of radionuclides have changed, the dose-assessment methods have not. As most of the fission-product noble gases and the activation gas are short-lived, attention has been focused on exposures to nearby residents.

70. When the radionuclide is uniformly distributed in the atmosphere or the photon energy is sufficiently low that this is a reasonable approximation over the volume of a plume, then the simplest calculational method is the semi-infinite cloud model. This method assumes that the radiation from the cloud is in electronic equilibrium, so that the energy absorbed by a given volume element equals that emitted by the same element. For a point at ground level, only half the space contributes to the dose, so that the energy absorbed is divided by two. The absorbed dose rate in air is then given by

$$\dot{D}_a = 0.5\frac{k}{\rho_a}\, C_a \sum_{i=1}^{n} F_i E_i \qquad (16)$$

where $\dot{D}_a$ is the absorbed dose rate (Gy h^{-1}); C_a is the average activity concentration of the radionuclide in the cloud (Bq m^{-3}); ρ_a is the mass density of air (kg m^{-3}); F_i is the fraction of photons of initial energy E_i (MeV) emitted per disintegration; and k is a conversion coefficient from energy deposition per unit mass and unit time to absorbed dose rate equal to 5.76 10^{-10} Gy h^{-1} (MeV $kg^{-1})^{-1}$. A modified version of this model, where F_i and E_i pertain to beta emissions, is used for beta irradiation of the skin.

71. If the distribution of the activity concentration in the plume is sufficiently non-uniform to invalidate the above approach, then a finite cloud model must be used. Such a condition arises near the source, when persons are not in

the cloud but receive dose from an overhead plume. In this model, the cloud is simulated by a number of small-volume sources, and integration is performed over these sources. The calculation proceeds by finding the photon flux density, summing over all the decay energies for the radionuclide of interest and then converting to absorbed dose. The basic expression for the photon fluence due to the fraction F_i of photons of energy E_i emitted per disintegration is [N2]

$$\phi_i = \int_V \frac{X_V F_i B_{en}(E_i, \mu_i x) e^{-\mu_i x}}{4 \pi x^2} dV \qquad (17)$$

where φ_i is the photon fluence; X_V is the concentration of the atoms of each radionuclide in volume element dV; μ_i is the linear attenuation coefficient, x is the distance from the volume element dV; and $B_{en}(E_i, \mu_i x)$ is the energy absorption build-up factor at a distance x for a radiation of initial energy E_i, having an attenuation coefficient μ_i. This integral is evaluated numerically.

72. In general terms, the Committee has considered 0.7 Sv Gy^{-1} to be the most appropriate average value of the quotient of effective dose rate to absorbed dose rate in air for males and females for environmental exposures to gamma rays. However, when the absorbed dose in air is the result of a calculation such as is described in this Section, then there are sufficient data on the photon energy spectrum to use more precise conversions. These conversion coefficients have been derived for infants, children, and adults by Saito et al. [S5, S11], based on a semi-infinite cloud model. These age-dependent results have not been used by the Committee, but the energy-dependent variations for the adult have been incorporated into the radionuclide-specific results [E7].

73. Based on the types of calculations indicated above, the Committee has estimated values of the collective effective doses from immersion exposure per unit release of fission noble gases and the activation gas ^{41}Ar [U6]. These calculations are updated in Table 10. On the assumption of a semi-infinite cloud and uniform concentrations over the mean paths of gamma rays in air, the effective dose rates to the adult per unit concentration of the radionuclide in air, d_i, have been calculated [E7]. The collective dose over the local and regional areas is evaluated as follows:

$$S_i = \int C_{a,i}(x)\, d_i\, N\, 2\pi x\, dx \qquad (18)$$

where d_i is the dose factor for radionuclide i, N is the number of inhabitants per unit area, and x is the downwind distance. The concentration of radionuclide i at distance x, $C_{a,i}(x)$, can be determined from equations (7–10). For short-lived radionuclides, radioactive decay during the dispersal time must be taken into account. In this case the concentration is

$$C_{a,i}(x) = C_{a,j}\, e^{-\lambda x/u} \qquad (19)$$

where $C_{a,j}$ was defined in equation (7), λ is the radioactive decay constant (s^{-1}), and u is the wind speed (m s^{-1}) for a given stability class. Since noble gases do not deposit, the wet and dry depletion factors D_w and D_d in equation (7) are set equal to 1 in these calculations.

74. Analytical evaluation of the integral, equation (18), with the expression of equations (7–10) is not possible, so a numerical integration is required. The results are given in Table 10. The radionuclide releases apply to the model site with the meteorological conditions given in Table 4. The release height was 30 m and the population densities were taken to be 400 inhabitants km^{-2} in the local area (1–50 km) and 20 inhabitants km^{-2} in the regional area (50–2,000 km). A similar method could be used to obtain the immersion dose from radon released from mill tailings, but the result is of much less significance than that due to inhalation.

75. The composition of noble gas releases from reactors is variable, depending on the reactor type and discharge delay features. If the composition is not known specifically, representative compositions may be assumed, such as used previously by the Committee and as listed in Table 11. For PWRs, the long-lived noble gas ^{133}Xe predominates with secondary release of ^{135}Xe. For BWRs, the composition includes several short-lived components. For GCRs, the noble gas release is assumed to comprise wholly ^{41}Ar. The dose factors derived in Table 11 to be applied in the general case to noble gas releases are 0.11 man Sv PBq^{-1} for PWRs, 0.43 man Sv PBq^{-1} for BWRs, and from Table 10 (^{41}Ar) 0.90 man Sv PBq^{-1} for GCRs. Because of changes in the parameters and calculational procedure, these values are slightly different from those previously derived [J1, U6].

76. For discharges from fuel reprocessing plants, the only radionuclide of interest in terms of cloud dose is ^{85}Kr [U6]. The Committee assessed the dose resulting from discharges of ^{85}Kr from the Windscale plant (Sellafield) between 1975 and 1979 using the methodology provided for the European Community [N2]. The average annual release of ^{85}Kr was 35 PBq and the resulting local and regional collective absorbed dose commitments were as follows: gonads, 0.058; breast, 0.078; red bone marrow, 0.095; lungs, 0.074; thyroid, 0.065; bone lining cells, 0.095; liver, 0.074; skin, 19; and remainder tissues, 0.078 man Gy. The collective effective dose equivalent commitment was estimated to be 0.074 man Sv from the cloud gamma irradiation using the conversion coefficients of Poston and Snyder [P3]. A further contribution from the beta irradiation to the skin is 0.19 man Sv, for which a skin-weighting factor of 0.01 is applied. Thus, the normalized collective effective dose commitment, $H_E + 0.01\, H_{skin}$, for this site is 0.0075 man Sv PBq^{-1}.

(d) Globally dispersed ^{85}Kr

77. The model used to calculate the global collective dose commitment from ^{85}Kr released at fuel reprocessing plants is given in the UNSCEAR 1982 Report [U6]. A two-compartment model similar to that proposed by Kelly et al. [K3] is used in which the released krypton is assumed to be instantaneously dispersed throughout the troposphere of the northern hemisphere, which is assumed to have a height of 10 km and a mass of $1.9\ 10^{18}$ kg (1 m^3 of air corresponds to 1.2 kg). Exchanges take place between the troposphere of the two hemispheres with a half-time of about two years. Within a few years the ^{85}Kr becomes uniformly dispersed, and the sole removal mechanism is radioactive decay.

78. The whole-body absorbed dose commitment per unit time integral of air concentration of ^{85}Kr was estimated to be $4.3\ 10^{-9}$ Gy (Bq a $kg^{-1})^{-1}$ [N2], and the dose commitment to the skin from the beta irradiation was $5.4\ 10^{-7}$ Gy (Bq a $kg^{-1})^{-1}$. These values were restated in the UNSCEAR 1988 Report [U4] to correspond to a collective effective dose equivalent commitment from ^{85}Kr of 0.17 man Sv PBq^{-1}, assuming a world population of $4\ 10^9$. This value was then scaled to a value of 0.2 man Sv PBq^{-1} for the world population of $4.6\ 10^9$ during the 1985–1989 period. Newer calculations [E7] indicate a value for effective dose equivalent, H_E, of $4.51\ 10^{-9}$ Sv (Bq a $kg^{-1})^{-1}$ and for skin, H_{skin}, of $5.00\ 10^{-7}$ Sv (Bq a $kg^{-1})^{-1}$, or 7.92 nSv (Bq a $m^{-3})^{-1}$ ($H_E + 0.01\ H_{skin}$). With this slight change and for a world population of $6\ 10^9$, the normalized effective dose commitment becomes 0.22 man Sv PBq^{-1}.

D. RADIONUCLIDES DEPOSITED ON SOIL

1. Exposure processes

79. Radionuclides released to the atmosphere undergo decay in transit or are deposited on the earth's surface by wet or dry deposition within relatively short periods. There follows a generally longer period in which the radionuclides on the terrestrial surface will eventually decay and produce external radiation exposure and dose to the population living in the areas. Radionuclides are initially deposited on the upper surface of the soil, but they quickly weather into the first centimetre of soil, especially if they are deposited via rainfall. This weathering effect and also the fact that the soil surface is not a smooth plane (soil roughness) reduce the radiation field at the generally used reference height of 1 m above the soil surface. Other mechanisms, such as plowing and countermeasures, can reduce the exposure rate, but such processes have not been considered in assessments of the Committee.

80. Following the deposition of radioactive material from the Chernobyl accident, several groups observed that the measured external gamma exposure rate decreased more rapidly over urban surfaces than over grass surfaces [J2, K6, S7]. Although varied, these results were consistent with the loss of half of the material with a half-time of seven days and the other half being firmly fixed on urban surfaces. This urban runoff effect was reflected in the Chernobyl assessment in the UNSCEAR 1988 Report [U4] by applying these coefficients to that portion of a country's population considered to be urban. Such an effect was not considered in the Committee's assessment of dose from nuclear weapons fallout.

2. Methods for estimating exposures

81. The Committee has traditionally used two approaches to estimate the external doses that result from the deposition of radionuclides on soil surfaces: direct measurements and calculations based on radionuclide deposition densities, which are the same procedures as used to evaluate exposures from naturally occurring radionuclides. As the calculational approach is more easily applied and as it is not always possible to measure very low dose rates, it is results of this approach that are more generally available.

(a) Atmospheric nuclear testing

82. The evaluation of radiation doses from fallout of radionuclides onto the earth's surface following the testing of nuclear weapons in the atmosphere was one of the earliest problems to be addressed by the Committee and one that has been regularly considered. The general method of assessing radiation doses from fallout from nuclear tests is indicated in Figure I. Within this model the external effective dose commitment, E_c, for a specific radionuclide released in an atmospheric test is

$$E_c = P_{01} P_{12} P_{25} A_0 = P_{25} F \qquad (20)$$

where A_0 is the amount released, P_{01} is the integrated concentration of a radionuclide in air at a specified location divided by the amount released, P_{12} is the quotient of the deposition density and the integrated air concentration, and P_{25} is the quotient of the effective dose commitment and the deposition density. The second part of the equation represents a more direct method of evaluation, namely beginning with the measured deposition density F (also equal to $P_{01}P_{12}A_0$) and multiplying this by the transfer coefficient P_{25}.

83. The P_{25} transfer coefficients for external irradiation have been calculated by multiplying the dose-rate conversion coefficients for radionuclides deposited on the ground, derived from Beck [B9], by the mean lifetime of the radionuclide and by an average factor accounting for air–to–tissue dose conversion, indoor occupancy in buildings (80% assumed) with a shielding factor of 0.2. The latter factor is 0.7 Sv Gy^{-1} (effective dose rate in the body per unit absorbed dose rate in air) times 0.36 (0.2 outdoor occupancy plus 0.8 indoor occupancy times 0.2

building shielding). For short-lived radionuclides (all except ^{137}Cs for fallout from nuclear testing) the dose-rate conversion coefficient applying to a plane source has been used. For ^{137}Cs, the dose-rate conversion coefficient applying to an exponential concentration profile in the ground of mean depth 3 cm is used. The indoor occupancy, as well as the shielding factor, can vary a great deal among different populations and is a source of uncertainty in the calculations of external dose. Also, the different behaviour of radionuclides deposited in urban and rural environments has not been taken into account for estimates of dose from nuclear weapons fallout. This difference was, however, considered for the assessment of doses from the Chernobyl accident (see below).

84. The P_{25} transfer coefficients that are used to estimate external doses from deposited radionuclides from fallout from nuclear testing are presented in Table 12. Transfer coefficients for many other radionuclides can be derived from the basic data of Beck [B9]. In earlier assessments the Committee assumed a plane source to be appropriate for short-lived radionuclides, however, to account for ground roughness, it is more realistic to assume an exponentially distributed source with a relaxation depth of 0.1, 1, and 3 cm for radionuclides of half-lives <30 days, 30–100 days, and >100 days, respectively. This change reduces the doses by about 15%–50%, but it does not have a significant impact on the calculated total dose from nuclear weapons fallout, which is dominated by the dose from ^{137}Cs.

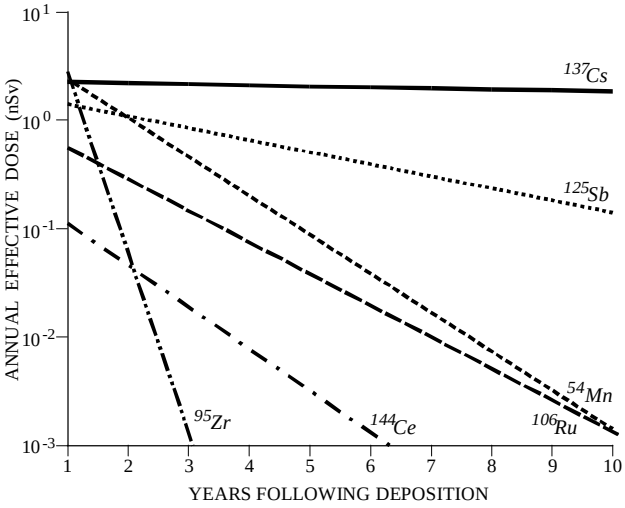

Figure III. External exposure following unit deposition (1 Bq m^{-2}) of radionuclides.

85. An indication of annual contributions to doses from external exposure following a single deposition event may be of interest, although the analysis is simple, as it depends only on the radioactive decay of the radionuclides. The time course of contributions to dose from unit deposition density of the radionuclides is illustrated in Figure III, and the annual average doses are listed in Table 13. Several short-lived radionuclides (^{131}I, ^{140}Ba, ^{141}Ce and ^{103}Ru) make no contributions to external exposure beyond the first year following deposition. The values in Table 13 have been calculated from decayed monthly deposition density, averaged over the year and multiplied by the absorbed dose rate in air per unit deposition density (Table 12, column 3) and by the shielding/occupancy factor of 0.36 and the conversion factor 0.7 Sv Gy^{-1}. The sum of the annual contributions to dose is equal to the dose commitment.

(b) The Chernobyl accident

86. The methods used to calculate external doses caused by the Chernobyl accident were basically those applied to estimate the external doses from radionuclides produced in atmospheric nuclear testing, although several modifications were introduced to account for the shorter term of the release, urban-rural differences, and an improved assessment of the movement of radionuclides into soil. The results of calculations of doses from the Chernobyl accident were presented in the UNSCEAR 1988 Report [U4]. During the first month after deposition, a number of short-lived emitters, including ^{132}Te, ^{132}I, ^{131}I, ^{140}Ba, ^{140}La, and ^{136}Cs, were important components of the total external gamma exposure rate (or dose rate in air). For several months, ^{103}Ru and ^{106}Ru made contributions, but since then only ^{134}Cs and ^{137}Cs have been of significance. Exposure from ^{137}Cs remains significant for several years and must be projected into the future.

(i) First month

87. The outdoor exposure X_1 (C kg^{-1}) during the first month was assessed by four different methods, with the choice dependent on the data available. If continuous or daily data were provided, the exposure rates were integrated. If incomplete data were provided, an attempt was made to fit a power function of the form at^b to the data, where t is time (days) and a and b are constants to be determined. X_1 is then the integral of this function from arrival day 1 to day 30.

88. If measurements of external gamma exposure rate were not available, two approaches were used. If data on the ground deposition of the radionuclides were provided, the exposure rate from each radionuclide was computed using the coefficients published by Beck [B9] for a relaxation depth of 1 mm to account for surface roughness. In several cases only data on the deposition of ^{137}Cs were available, and X_1 was evaluated on the basis of the relationship of the exposure to ^{137}Cs deposition density as measured at a specific location, e.g. Neuherberg, Germany [G2].

89. The effective dose during the first month, E_{e1}, (Sv) was calculated from X_1 by:

$$E_{e1} = AX_1(1 - F_0) + AX_1F_0F_s \qquad (21)$$

where A is the conversion coefficient (23.6 Sv per C kg^{-1}, i.e. 33.7 Gy per C kg^{-1} × 0.7 Sv Gy^{-1}), F_0 is the indoor occupancy factor, and F_s is the building shielding factor. The last two values were taken as 0.8 and 0.2, respectively.

(ii) One month to one year

90. The calculation of external gamma dose beyond one month was based on the measured total deposition of ^{134}Cs and ^{137}Cs and, although less important, ^{103}Ru, ^{106}Ru, and ^{131}I. The conversion coefficients for long-term deposition to dose rate depend on the penetration of these radionuclides into soil. Change with time is accounted for by using coefficients appropriate for a relaxation depth of 1 cm during the first year and 3 cm thereafter. Also, the effect of more rapid removal of radionuclides from urban surfaces was considered.

91. The equation for the calculation of external gamma effective dose, E_{e2}, (Sv) for the time period between one month and one year for radionuclide i is as follows:

$$E_{e2,i} = \left[\frac{F_i}{\lambda_i}\right]\left[d_{e2,i}(e^{-\lambda_i m/12} - e^{-\lambda_i m})\right] \left[1 - F_0(1 - F_s)\right]\left[1 - F_p(1 - F_u)\right] \quad (22)$$

where F_i is the deposition density (Bq m^{-2}); $d_{e2,i}$ is the deposition density to effective dose conversion coefficient during the period between one month and one year (relaxation depth of 1 cm) (Sv per Bq m^{-2}); λ_i is the radioactivity decay constant (a^{-1}); m is a constant equal to one year; F_p is the urban fraction of a country's population; F_u is the fraction of the deposition that remains fixed on urban surfaces (assumed to be equal to 0.5); and F_0 and F_s are as defined previously. Effective dose equivalent conversion coefficients are listed in Table 14.

(iii) Periods beyond one year

92. External effective dose, $E_{e3,i}$, (Sv) for periods beyond one year were evaluated according to the equation

$$E_{e3,i} = \left[\frac{F_i}{\lambda_i}\right]\left[d_{e3,i}\, e^{-\lambda_i m}\right] \left[1 - F_0(1 - F_s)\right]\left[1 - F_p(1 - F_u)\right] \quad (23)$$

where $d_{e3,i}$ is the deposition density to effective conversion coefficient for periods greater than one year. This coefficient is based on a relaxation depth of 3 cm. Values of this coefficient are also listed in Table 14.

(c) Nuclear installations

93. Releases from nuclear installations of radionuclides that contribute to external exposures are, in general, too low to be measured in air or deposition at distances beyond the installation site and point of release. As was discussed in Section I.B.3, long-term average dispersion of radionuclides in air may be estimated using a formulation that combines a dilution factor at 1 km and a power function of distance from the release point. With use of an effective deposition velocity that accounts for both wet and dry deposition, the deposition densities of radionuclides may be estimated. This method is appropriate for routine continuous and near-surface releases from sources such as nuclear installations. The local area of exposure is taken to be 1–50 km surrounding the point of release, and the regional area extends to 2,000 km.

94. In the dispersion estimation method, equation (11), an average dilution factor is assigned at 1 km, namely $5\ 10^{-7}$ Bq m^{-3} per Bq s^{-1} released, and further dispersion reduces the radionuclide concentration in air in inverse proportionality to the 1.4 power of the distance. The air concentration may be related to the deposition density by multiplying by the effective deposition velocity. The general formula for application of the transfer factor method is

$$S_i = 5\ 10^{-7}\, v_g\, P_{25} \left[N_1 \int_1^{50} x^{-1.4}\, 2\pi x\, dx + N_2 \int_{50}^{2000} x^{-1.4}\, 2\pi x\, dx\right] \quad (24)$$

where S_i is the collective effective dose per unit release of radionuclide i (man Sv Bq^{-1}); v_g is the effective deposition velocity (m s^{-1}); P_{25} is the transfer factor from deposition density to dose (Sv per Bq m^{-2}); N_1 is the population density in the local area (inhabitants km^{-2}); N_2 is the population density in the regional area (inhabitants km^{-2}); and x is the distance from the point of release (km). The parameter $x^{-1.4}$ should actually be expressed as $(x/1\ km)^{-1.4}$ to rectify the units. The quantity in brackets has the unit number of persons. The population densities applied are those assumed for model reactor site: $N_1 = 400$ inhabitants km^{-2} and $N_2 = 20$ inhabitants km^{-2}. The value of the effective deposition velocity is taken to be 0.002 m s^{-1} for annual average deposition, which is the value for dry deposition alone. In reality, more material is deposited under wet conditions than under dry, and an effective deposition velocity for point sources that includes both wet and dry contributions would range from 0.005 to 0.013 m s^{-1}, depending on downwind distance. However, use of a larger value in conjunction with the power law expression for the air concentration (equation 11) results in a greater estimated activity amount deposited in the local and regional areas than was released to the atmosphere. The most probable explanation for this is that equation 11 overestimates air concentrations, for the reason given in paragraph 41. Although larger (more negative) values of the exponent in the power function could be selected to offset a higher effective deposition velocity, the values of 1.4 and 0.002 m s^{-1} preserve the mass balance to distances of 2,000 km and ensure that doses from airborne and deposited activity are not underestimated. Therefore, these values are used on local and regional scales for purposes of estimating average deposition.

95. Estimates of normalized collective effective doses from external exposure from radionuclides released as particles in airborne effluents from reactors are listed in Table 15. The transfer coefficients P_{25} from deposition density to effective dose were derived from the basic data of Beck [B9]. The collective effective doses per unit release were then estimated using equation (24). Other fission and activation products could be added to Table 15 by applying this method.

96. A representative composition of radionuclides in particulates released in airborne effluents is not easily established because of the large number and varying amounts of radionuclides that may be involved. An earlier approach of the Committee [U6] was to assume equal activity distribution across 18 radionuclides that were commonly reported to be present. A slight variation would be to recognize two groups of radionuclides, one of more dominant contributors to the total activity release and a secondary group. Consistent with reported data [U5, U6] is to assume 90% of the activity of release present as ^{54}Mn, ^{58}Co, ^{60}Co, ^{89}Sr , ^{134}Cs, ^{137}Cs, and ^{140}Ba (including ^{140}La). The radionuclides of the second group, contributing 10% of the activity total, are ^{51}Cr, ^{59}Fe, ^{65}Zn, ^{90}Sr, ^{90}Y, ^{95}Zr (including ^{95}Nb), ^{124}Sb, ^{136}Cs, ^{141}Ce, and ^{144}Ce. Assuming equal contributions to the activity release within each group, weighted average values of the local and regional collective effective doses per unit total (representative) release of particulates are obtained. These results are included in Table 15. Adjusted weightings could be made in specific circumstances, if the exact composition of the release is known.

III. INHALATION EXPOSURE

97. There are two main processes that contribute to internal exposure, the general term used to describe exposures that involve the intake of radionuclides into the body as opposed to external exposure, which is considered in Chapter II above. The two processes are inhalation of contaminated air and ingestion of contaminated foodstuffs. For inhalation, if the time dependent concentration of a radionuclide in air is known, it is a straightforward matter to calculate committed dose by multiplying by a breathing rate and by a dose-conversion coefficient. The ingestion pathway involves additional steps of transfer to plants and animals, from which are derived the foods consumed by humans. For convenience, inhalation exposures are considered in this Chapter and ingestion exposure in the following Chapter.

98. Many of the Committee's past calculations of inhalation doses were performed using a nominal breathing rate of 20 $m^3\ d^{-1}$, or 7,300 $m^3\ a^{-1}$. This generally reflects the concern of the Committee with the collective dose, which is substantially determined by the intake of the adults in the population. For calculating inhalation doses from the Chernobyl accident, inhalation rates of 22 $m^3\ d^{-1}$ for adults and 3.8 $m^3\ d^{-1}$ for infants were used [U4]. The latter values are the same as those used for naturally occurring radionuclides and are derived from the same source [I7].

99. The Committee has generally used the dose coefficients published by ICRP for its evaluations. Initially such values were available only for adult workers, but starting in 1989 age-dependent values have been made available for members of the general public. The latest compilation of values for both ingestion and inhalation is provided in [I5]. The breathing rates now used by ICRP [I4] are indicated in Table 16. An indication is also given in Table 16 of the fraction of the population in each of the six age categories and the age-weighted average breathing rate. The age-weighted value corresponds to 19 $m^3\ d^{-1}$. Considering the uncertainty of the age distribution of the population and the differences between countries, a rounded value of the nominal breathing rate of 20 $m^3\ d^{-1}$ would seem to be appropriate for use in most applications. In assessments of the Committee, the population groups specified as infants, children, and adults are assumed to correspond with the ICRP age categories of 1–2 years, 8–12 years, and >17 years, respectively.

A. NATURAL RADIONUCLIDES

100. Naturally occurring radionuclides are present in the atmosphere owing to their production by cosmic ray interactions, the emanation of gases from soil or building materials and the resuspension of soil particles from the ground surface. The main cosmogenic radionuclides, ^{3}H and ^{14}C, are fairly uniformly dispersed in the atmosphere. Inhalation exposures from these radionuclides are, however, almost completely negligible compared with the ingestion exposures.

101. Soil-derived radionuclides are present in air in variable amounts, depending on local soil, wind, and moisture conditions. In earlier assessments by UNSCEAR [U6, U7], a dust loading of 50 μg m^{-3} was assumed and applied to typical concentrations of natural radionuclides in soil. Some portion of the solid matter in air may not come from the soil, however, but from organic matter, building dusts, smoke, and fly ash from coal burning.

102. A very important contribution to inhalation exposure is made by radon and its decay products. The gas emanates from soil and can enter and attain high concentrations in indoor spaces. Because this exposure component dominates that from all other pathways, it is important that the dosimetry for radon be well established.

103. The ICRP has not provided values of the doses per unit intake for ^{222}Rn and ^{220}Rn and their decay products from application of the respiratory tract model [I4, I5], and the dosimetry for these mixtures is very complex. Because lung cancer has been observed and studied extensively in miners exposed to ^{222}Rn, the ICRP [I13] has adopted a conversion convention for radon exposures that is based on equality of detriments from epidemiological determinations. The detriment per unit effective dose for members of the public is $7.3\ 10^{-5}$ per mSv, and the detriment (to miners) per unit exposure to ^{222}Rn progeny is $8.0\ 10^{-5}$ per (mJ h m^{-3}). Thus, an exposure to ^{222}Rn progeny of 1 mJ h m^{-3} is equivalent to an effective dose of 1.10 mSv. As 1 mJ h m^{-3} is equal to $1.80\ 10^{5}$ Bq h m^{-3} of ^{222}Rn in equilibrium with its short-lived progeny, a dose coefficient of 6.1 nSv per (Bq h m^{-3}) can be derived and applied to equivalent equilibrium concentrations (the activity concentration of radon, in equilibrium with its short-lived progeny, which would have the same potential alpha energy concentration as the existing non-equilibrium mixture). The dosimetric evaluations give dose coefficients in the range 6–15 nSv (Bq h m^{-3})$^{-1}$. The value previously used by the Committee in earlier evaluations [U3, U4], 9 nSv (Bq h m^{-3})$^{-1}$, is within this range and would seem to be still appropriate for use in dose evaluations. An epidemiologically based conversion convention is not available for ^{220}Rn. However, by analogy with the risk determined for ^{222}Rn and by comparing the dose coefficients for ^{220}Rn and ^{222}Rn calculated on a dosimetric basis [I18], a dose-conversion convention of 40 nSv per (Bq h m^{-3}) equilibrium equivalent concentration of ^{220}Rn can be derived; this value is intended to include the dose to organs other than lung due to the transfer of ^{212}Pb from the lung. The half-life of ^{212}Pb is sufficiently long (10.64 h) for this effect to be significant, whereas none of the short-lived progeny of ^{222}Rn is sufficiently long-lived to merit similar consideration.

B. RADIONUCLIDES RELEASED TO THE ATMOSPHERE

104. In its various assessments, the Committee has used the best available estimates of dose per unit intake of radionuclides by inhalation; whenever possible, the values provided by ICRP have been used. The ICRP values have been updated [I4, I5] based on a revision to the ICRP model of the respiratory tract [I6], and age-dependent values for the general public are now provided. The values for radionuclides used by the Committee in its assessments are given in Table 17.

1. Exposure processes

105. Inhalation of radionuclides in air can result from a short-term or continuous release processes. Inhalation is rarely the primary pathway of exposure if radionuclides are released to the atmosphere, but there are some notable exceptions. The importance of radon and its decay products was mentioned in the preceding Section. Another exception involves radionuclides of extremely low biological availability. Such radionuclides pass readily through the gut following ingestion intake, but they can be deposited in the lungs following inhalation intake and be retained for long times. The most notable example of such a radionuclide is ^{239,240}Pu.

2. Methods for estimating exposures

(a) Atmospheric nuclear testing

106. According to the general model developed by the Committee to describe environmental transport processes, the equation for committed effective dose, E_c, (Sv) via inhalation is

$$E_c = P_{01} P_{14} P_{45} A_0 = P_{245} F \qquad (25)$$

where $P_{01}A_0$ is the integrated air concentration (Bq a m^{-3}), P_{14} is the breathing rate (m^3 a^{-1}) and P_{45} is the dose-conversion coefficient (Sv Bq^{-1}) for inhalation. To determine the integrated air concentration, measurements must be made for the entire time that radionuclides remain in air. Since this is not always achieved in practice, the second part of the equation is the more common approach, in which the integrated air concentration is estimated from the deposition density, F. In this case, the transfer coefficient for the inhalation pathway is determined as $P_{245} = P_{14}P_{45}/P_{12}$.

107. The average value of P_{12}, which is also the effective deposition velocity, varies with the precipitation rate at different locations and also with the chemical and physical nature of the radionuclide considered. The average value of P_{12} for particulate material deposited following atmospheric nuclear testing has been estimated to be 1.76 cm s^{-1}, or $5.56\ 10^{5}$ m a^{-1} [B2]. Although this value is based on observations in New York City over several years, measurements in the United Kingdom [C7] and Sweden [B10, D5] are in reasonable agreement after normalization to the same annual precipitation. Furthermore, since the annual rainfall in New York City is fairly close to the population-weighted average for the whole world, the New York value is considered adequate for global average calculations.

108. Values of the transfer coefficient, P_{245}, for the inhalation pathway are listed in Table 18. These update the listing in the UNSCEAR 1993 Report (Table 8, page 127 [U3]). The values are for the adult with a breathing rate of

7,300 m^3 a^{-1} and P_{45} values from Table 17. These transfer coefficients are applicable to the release and deposition conditions of radionuclides in fallout from nuclear tests.

(b) The Chernobyl accident

109. For the Chernobyl accident assessment, a somewhat modified approach was used to account for a filtration effect that reduces the concentrations of radionuclides in indoor air [U4]. The calculation of the inhalation committed effective dose, $E_{h,i}$ (Sv) for radionuclide i was as follows:

$$E_{h,i} = C^*_{a,i} \; B \; d_{h,i}(1 - F_0) + C^*_{a,i} \; B \; d_{h,i} \; F_0 \; F_r \qquad (26)$$

where $C^*_{a,i}$ is the integrated activity concentration of radionuclide i in outdoor air, B is the breathing rate, $d_{h,i}$ is the committed dose per unit intake from inhalation, F_0 is the indoor occupancy factor and F_r is the ratio of indoor to outdoor air concentration. The latter parameter was assigned a value of 0.3 for all countries [C9, R2, U4].

110. If the integrated concentration in air is known, then the calculation is very simple as indicated above. Furthermore, if an average concentration over a one-year period is known, then the calculation is also quite straightforward. It is, however, rather rare that measurements of integrated activity in air are available following accidental releases, especially over a short period of time. In that case the integrated concentration in air is usually estimated on the basis of the deposition density for a particular radionuclide and the effective deposition velocity, as mentioned above. The deposition density divided by the deposition velocity gives the integrated concentration in air.

111. If the relative amounts of the radionuclides released at the time are known and if these releases are concurrent, then the measurement of the deposition density for only one radionuclide in the mixture can be considered sufficient to define the deposition densities of all radionuclides at the time of deposition, if the deposition velocities of the radioelements do not differ significantly. In fact, measurements of the deposition density of a long-lived radionuclide can be made many years after the deposition occurred and used to define the original deposition densities of all radionuclides, provided that the soil is undisturbed and the sampling is deep enough to encompass all of the original deposition.

112. Other methods can be used to define the deposition densities and the integrated air concentrations of radionuclides. Although subject to more error and in need of more sophisticated interpretation, measurements of the external gamma-dose rate in air, of concentrations of radionuclides in foodstuffs, and even of radionuclides in people can be used to estimate the original deposition densities and integrated air concentrations.

(c) Nuclear installations

113. Estimates of inhalation exposure from releases of radionuclides from nuclear installations may be made using the dispersion model presented in Section I.B.3 and the transfer coefficients P_{245}. The results of this calculation are listed in Table 19. These estimates apply to longer-term releases, as the meteorological conditions for the representative site have been averaged over an annual period. The deposition velocity appropriate for near-surface releases of 0.002 m s^{-1} has been used. This is determined mainly by dry deposition, since precipitation can be expected to occur only during a small fraction of the time of plume passage.

114. As discussed above with regard to external exposure (paragraph 96), a representative composition of radionuclides in particulates released in airborne effluents from reactors may be assumed. A weighted average of the collective dose from inhalation exposure per unit release of particulates may then be derived for general application. The values pertaining to the local and regional areas are included in Table 19. The transuranium radionuclides are not normally reported in routine releases from reactors, however for reference purposes, the values are included in Table 19.

IV. INGESTION EXPOSURE

115. Ingestion exposure occurs when radionuclides in the environment enter food chains. This component and that of external exposure are usually the significant and continuing sources of exposure following releases of radionuclides to the environment. Radionuclides released to the atmosphere may deposit onto both terrestrial and aquatic surfaces, for which different calculational methods are required. The terrestrial and aquatic food pathways are considered in separate Sections of this Chapter.

116. Ingestion exposures have been evaluated by UNSCEAR for natural radionuclides present in the environment and for several cases of radionuclide release to the environment, including atmospheric testing, releases from nuclear fuel cycle installations and the Chernobyl accident. For the most part, annual average values have been considered with the aim of evaluating committed exposures. This is adequate for longer-term or continuous releases. Short-term releases at particular times, such as

was the case for the Chernobyl accident, require taking into account some seasonal variations.

A. NATURAL RADIONUCLIDES

117. In the general case, doses from the ingestion of natural radionuclides in foods and drinking water have been estimated from measured concentrations of the radionuclides in body tissues or organs. For ^{40}K, metabolic balance maintains body levels irrespective of intake amounts. For uranium- and thorium- series radionuclides, however, this is not the case, and the concentrations in foods, water and total diet have been useful for determining geographic variations in the body burdens.

118. Beginning with the UNSCEAR 1993 Report [U3], representative dietary intakes of natural radionuclides were compiled; these could be used with age-dependent estimates of dose per unit intake to extend the limited data on tissue concentrations and to obtain more broadly based dose estimates.

119. Estimates of dose per unit intake of radionuclides are provided by the ICRP [I5]. These are the committed effective doses to age 70 years, based on recent metabolic data and models. The values used in UNSCEAR assessments are summarized in Table 20. The age categories are infants (1–2 years), children (>7 years to 12 years), and adult (>17 years). Values for age categories from 0 to 1 year, >2 years to 7 years, and >12 years to 17 years are also provided by ICRP [I5].

B. RADIONUCLIDES RELEASED TO THE TERRESTRIAL ENVIRONMENT

120. An extensive database of deposition and diet measurements from the years when there was atmospheric testing has allowed empirical relationships to be derived to evaluate transfer coefficients for radionuclides released in this practice. Empirical models describing the time course of annual transfers from deposition to diet and from diet to the body have been the basis of the Committee's evaluations of doses from ^{90}Sr and ^{137}Cs, and this method was also applied to transuranic radionuclides. Fewer data have been available from which to derive ingestion pathway transfer coefficients for ^{131}I, ^{140}Ba, ^{89}Sr and ^{55}Fe.

1. Transfer processes

121. Plants are the primary recipients of radioactive contamination to the food chain following atmospheric releases of radionuclides. Vegetation may be subject to direct and indirect contamination. The direct contamination of terrestrial vegetation refers to the deposition of radioactive materials from the atmosphere onto the above-ground parts of plants. Indirect contamination refers to the sorption of radionuclides from the soil by the root system of plants. Secondary recipients of food chain contamination are animals that consume plants or other animals. Both plant and animal products enter the diet of humans.

(a) Direct deposition on plants

122. Direct deposition on plants may play an important role in the contamination of plant products for some radionuclides, including those characterized by low root uptake and short-lived radionuclides, especially ^{131}I, that can transfer relatively rapidly through the food chain. The direct contamination of plants may be of two types: primary, which involves direct transfer from the source via the atmosphere to the plants, and secondary, by which activity already deposited on the ground may be resuspended, e.g. by the wind, and thus transferred to the plants. The resuspension process is not usually a substantial factor, except for radionuclides with very small uptake through the roots. Primary direct deposition involves three processes: deposition, interception and retention. Direct contamination of the plants depends on the development stage of the plants at the time of contamination. This, in turn, depends on the season of the year when the contamination occurs.

123. Radionuclides in the atmosphere may be deposited as either dry or as wet deposition. Dry deposition occurs continuously, while wet deposition occurs when rain or some other form of precipitation intervenes. Dry deposition is usually described by applying the deposition velocity, v_g = F/C [C1], where F is the fallout rate of the depositing radionuclide to a unit area of land (Bq m^{-2} s^{-1}), and C is the concentration in ground-level air over the area of land considered (Bq m^{-3}). The unit of v_g is thus m s^{-1}. The deposition velocity varies with the aerodynamic diameter of the particles deposited. Particles with a diameter between 0.1 and 1 μm have a deposition velocity of about 0.02 cm s^{-1}; those between 1 and 10 μm have values ranging from 0.02 to about 5 cm s^{-1} [H1]. This magnitude also varies with the type of surface and with the chemical and physical characteristics of the radioelements involved.

124. Wet deposition occurs during precipitation. The wash-out ratio, W, is defined as the ratio of the radionuclide concentrations in precipitation (Bq l^{-1}) and in ground-level air (Bq m^{-3}) [E3]. Experience from global fallout studies has shown that around 90% of the total deposition of ^{90}Sr and ^{137}Cs occurs as wet deposition. In an accident, most of the deposition usually takes place within a few days. The Chernobyl accident demonstrated that high rainfall during the cloud passage results in deposition rates an order of magnitude higher than those observed for dry conditions [E5].

125. Interception is the fractional deposition of radionuclides on the plant surfaces. It depends on both the physical characteristics of the deposit and the growth form of the plants. The subsequent fate of the deposit, i.e. the

retention, is influenced by these factors and by the rate at which the material is removed by precipitation and other processes, called weathering or field loss.

126. The fraction of material intercepted by the crop canopy was studied by Chamberlain [C2], who derived an empirical parameter dependent on the physico-chemical properties of the deposit, the manner of deposition, the morphology of the crop and the meteorological conditions. The quotient of the fraction retained and the dry weight biomass usually falls within the range 0.2–4 $m^2\ kg^{-1}$ [C3]. The normalized specific activity is defined as the concentration in the crop (Bq kg^{-1} dry weight) divided by the deposition density rate (Bq $d^{-1}\ m^{-2}$) [C2]. The normalized specific activity is thus a rate factor with the unit $m^2\ kg^{-1}$ d. Values between 20 and 40 $m^2\ kg^{-1}$ d have been observed for ^{137}Cs and ^{90}Sr for herbage in good growing conditions [E4]. Chamberlain found that winter grass had normalized specific-activity values 2–3 times higher than summer grass.

127. The weathering or field loss is expressed by $M/M_0 = e^{-t/\tau}$, where M_0 and M are the quantities retained on the crop initially and after time t and τ is an empirical constant. During the growing season, τ is about two weeks; in the winter period, it increases to about eight weeks. When there is rain, the field half-time may be short.

128. Resuspension of radionuclides on the soil surface may result in secondary direct contamination of the crops. The resuspension factor RF is defined as the radionuclide concentration in air (Bq m^{-3}) divided by the ground contamination (Bq m^{-2}). The resuspension factor thus has the unit m^{-1}. The resuspension factor measured at locations in Denmark for 100–3,000 days after the Chernobyl accident decreased according to a power function of the time, t, in days ($RF = 9.3\ 10^{-6}\ t^{-1.17}$) [A3].

129. It appears that resuspended ^{137}Cs is less available to the plant than primarily deposited amounts [A1], i.e. the transfer factor for primary direct contamination is higher than that for secondary direct contamination. There may be two reasons for the lower availability of resuspended particles compared with directly deposited fallout. First, a higher field loss can be expected for resuspended particles than for global fallout. Secondly, ^{137}Cs adheres to minerals, especially clay, allowing the radiocaesium to be less available for absorption by the crops and thus for translocation to the grain.

130. A special case of secondary direct contamination of crops is rain splash, which may occur during heavy showers, when the recoil from rain drops carries contaminated soil to the surface of the vegetation. Secondary contamination is expected to be less efficient with respect to translocation to the plants than the initial, direct contamination route.

131. Seasonal variation in direct contamination is of particular importance for cereals. This feature was first studied by Middleton [M2]. It appears that the two important factors influencing contamination of grain are the initial retention and the translocation from the vegetative part of the seeds. Initial retention is largely independent of the radionuclide, whereas translocation depends strongly on the radioelement and its solubility.

132. Time of year was observed to affect the transfer factor of ^{137}Cs to grain at the time of the Chernobyl accident [U4]. Transfer factors were higher in southern Europe, where the crops were more developed when the deposition from Chernobyl occurred than in northern Europe, where the growth of crops had not yet begun. Seasonality also affected total diet intakes.

(b) Root uptake

133. In the first period after a radioactive contamination event, direct deposition on plant surfaces is the dominant pathway, but in the long term, the contamination of the human diet will depend on absorption through the roots of plants. The extent to which plant roots absorb radionuclides from the soil depends not only on their physiology but also on processes in the soil.

134. The uptake of radionuclides by plants from the soil is normally described by the transfer factor B_v, the ratio of radionuclide concentrations in vegetation and soil (Bq kg^{-1} dry weight plant to Bq kg^{-1} dry weight soil). Observed values of B_v vary widely, mainly as a result of different soil and vegetation types and environmental conditions. In addition, management practices such as ploughing, liming, fertilization and irrigation greatly affect uptake. Variability can also result if uptake into the whole plant is compared with uptake into parts of the plant, such as grain. The transfer factor B_v is not constant in time. Decreases occur as radionuclides in soil become less available to plants through changes in physical or chemical forms or in moving below the rooting zone. In some cases, the rate of uptake increases in time, when physical weathering or transformation of the chemical form takes place or when the radionuclide reaches an optimum depth for root uptake. Databases for root uptake transfer parameters have been published [I9, N7].

135. The main soil characteristics affecting the transfer of radionuclides from soil to plants through root uptake are: clay and organic matter content, pH and cation exchange capacity. These soil characteristics interact causing variability in the transfer in different circumstances, so that generalizations are not always valid. A high clay content in the soil provides binding for caesium and reduces root uptake. A high organic matter content often enhances the root uptake of caesium but may also have the opposite effect; an excess of potassium dilutes caesium ions, which decreases uptake, but may also cause the desorption of fixed caesium, which increases uptake.

136. The root uptake of ^{137}Cs usually decreases with time, in the beginning quite rapidly, later more slowly. The

decrease is seen particularly in clay soils and is due to the fixation of caesium by clay minerals such as illite and vermiculite. In organic soils the decrease is mainly due to redistribution of caesium within and transport out of the rooting zone. Strontium is less firmly fixed to the soil matrix and is thus more available for root uptake than caesium. The higher mobility of ^{90}Sr also means that this radionuclide migrates faster than ^{137}Cs through the soil column. Nevertheless, root uptake of ^{90}Sr generally remains significantly greater than that of ^{137}Cs over periods of several years, and for terrestrially produced foods a generally increasing ^{90}Sr/^{137}Cs ratio will occur with time after deposition.

137. Under special circumstances the root uptake, especially of ^{90}Sr, may increase with time after contamination. This has been observed, for example, in the near zone around the Chernobyl reactor where some of the ^{90}Sr was imbedded in uranium fuel particles. Weathering throughout the years has dissolved these fuel particles, making the ^{90}Sr available for delayed root uptake by plants.

(c) Animal pathways

138. Several important pathways for the transfer of radionuclides to the diet of humans involve animal food chains, including milk and eggs from living animals and meat or flesh from animals and fish. Depending on the radionuclide and the metabolism in the organism, the concentrations may be enhanced or reduced compared with the earlier steps of the food chain. Some parts of the animal are not consumed, e.g. bones, shells, skin and feathers, and this prevents the transfer from animal products of bone-seeking radionuclides such as ^{90}Sr and plutonium. Bone tissue might, however, re-enter the food chain as bonemeal in various fodder products, and it might also appear in fertilizers.

139. The main animal pathway to humans of the radiologically important radionuclides such as ^{90}Sr, ^{131}I and ^{137}Cs is milk consumption. All three radionuclides are readily transferred from animal fodder to the milk. Other radionuclides such as the transuranic elements are absent or secreted to only a very small extent in milk. Caesium is transferred with its chemical congener potassium to the soft tissues of animals, particularly muscle. Strontium is preferentially transferred to bone, like its congener calcium.

140. Fish and shellfish receive radionuclides both directly from the water and from their food. Some radionuclides that are of no concern in the terrestrial animal food chains may be concentrated in aquatic animals. This is the case, for example, for plutonium, which is concentrated in crustaceans, and for polonium in fish and seafood. A substantial part of the marine fish catch is used for making fishmeal, which is used as fodder for pigs and poultry and for fish produced in fish farms. In this way, marine pathways may interact with terrestrial and freshwater animal food chains.

(d) Losses in food preparation

141. Knowledge of the effects of processing and culinary preparation on radionuclide contents in foods is needed when assessing the radiation dose to humans from the ingestion of contaminated foodstuffs. Appropriate allowances might be made for the reductions brought about by food processing to ensure that doses are not systematically overestimated [N1]. However, the Committee has not specifically considered this for its calculations. In some cases, losses via food processing are considered implicitly, if the assessment is based upon nuclide content in people.

142. Food-processing retention factors, i.e. the fractional amount of the radionuclide remaining in the food after processing, are quite variable, depending on the food and the processing procedure. Drying foods increases the concentrations in the dried products, typically by a factor of 5 compared with the fresh foods. Boiling meat considerably reduces the radionuclide content. It should, however, be kept in mind that some of the water used for the boiling may be consumed as soup or sauce. In dairy products, radionuclides are retained less in cream, thus affecting the levels in various milk products. Radionuclide contents in vegetables and fruits are also significantly affected by washing, peeling, and cooking. In particular, the reduction of ^{137}Cs by various treatments is significant. If crops have been contaminated only by direct deposition, the effect of washing and peeling will be even higher, because the contamination in that case is confined to the outer parts of the crops. Some translocation may eventually take place.

143. The process of milling cereal grains apportions the radionuclide content of the whole grain to significantly lower radionuclide concentrations in the flour and correspondingly higher concentrations in the bran. The intakes of ^{137}Cs and particularly ^{90}Sr are thus higher for consumers of wholemeal bread than for consumers of white bread. The concentrations of ^{90}Sr and ^{137}Cs in white bread are 20% and 40% of the concentrations in the wheat, respectively. In rye bread, the percentage is 75% of that in the grain for both radionuclides. There is essentially no transfer of ^{90}Sr or ^{137}Cs to alcohol from grain or potatoes nor to sugar made from beets.

144. Conversion of foods, e.g. milk to cheese, may also change the radionuclide concentrations. The concentration of ^{90}Sr in cheese is thus typically 5–10 times higher than in milk, while the concentration of ^{137}Cs in normal cheese is only about 70% of that in milk. Butter contains essentially none of the ^{90}Sr and ^{137}Cs present in the milk.

145. Assessments by UNSCEAR have not specifically accounted for losses in food preparation. Rather, it has been assumed that dietary intake estimates reflect actual amounts in prepared and consumed foods. When it appears that this is not the case, the intake estimates will need to be adjusted.

(e) Behaviour of tritium and carbon-14

146. The radionuclides tritium and ^{14}C require special consideration because of their high mobility in the environment and the fundamental nature of hydrogen and carbon cycles in the biosphere.

147. Carbon is highly mobile and is distributed throughout the environment. A small fractionation effect reduces environmental concentrations of ^{14}C by about 5% relative to stable ^{12}C, but this difference is usually disregarded in the models. Carbon-14 released into the environment via the atmosphere enters the carbon cycle and becomes dispersed in the atmosphere, terrestrial biosphere, and more gradually into the ocean, ocean sediment and sedimentary rocks. Much of the carbon in plants has a short residence time, although carbon is held longer in woody plant parts and is released only on decomposition. Turnover time of carbon in humans is generally of the order of a few days or weeks. The most important form of carbon from the point of view of dose is CO_2, since this is the form in which carbon becomes bound in plants and ingestion contributes 99% of the dose from ^{14}C. The remaining fraction of dose comes from inhalation of ^{14}C in air.

148. Tritium released to the atmosphere occurs in two forms: tritiated hydrogen (HT) and tritiated water vapour (HTO). HTO is subject to the same wet and dry deposition processes as other nuclides, but it can also diffuse into the soil pore space and the leaf stomates [B7, G1]. If the HTO gradient is reversed, however, (for example, if a wind shift blows the plume away), tritium will rapidly be lost from the soil and plants to the atmosphere by evaporation and transpiration, generating a secondary airborne HTO plume. HT can diffuse into the soil and be converted to HTO by an enzyme-mediated reaction [D2, T2]. Tritium not returned to the atmosphere by evaporation moves through the soil primarily by the mass flow of liquid water.

149. Like other radionuclides, tritium enters plants via root uptake. Under steady conditions, the concentration in the plant lies between the concentrations in the soil and the air, with a magnitude that depends on atmospheric humidity and the air/leaf temperature difference [M4]. Some of the tritium that enters plants can be incorporated into organic compounds to form organically bound tritium (OBT) [D3]. Tritium bonded to carbon forms non-exchangeable OBT, which has a much longer retention time in plants and animals than HTO and so can contribute significantly to the total dose. Organically bound tritium makes up only a small percentage of the total tritium activity in most plants, but up to 90% in grains, which have a high organic content.

150. Tritium is taken into the bodies of animals (including humans) by the normal mechanisms, and HTO equilibrates with body fluids within minutes. For the most part, the retention time of tritium in the body is about 10 days, although for the organically bound form it increases to about 40 days [T3].

2. Food and water consumption

151. The consumption of foods and water by individuals varies widely around the world, depending on climate, food availability and cultural dietary preferences. Locally produced or gathered foods are now usually greatly supplemented by foods imported from other regions or countries. Moreover, it is difficult to obtain accurate estimates of food consumption: there are considerable individual variations, and many foods are of a seasonal nature. Average rates in countries may be indicated by food balance analysis, taking into account local production, imports, and exports [F1]. These will be overestimates, however, if losses from wastage or preparation are not taken into account.

152. When UNSCEAR has needed dietary intake information, it has used values reported from a few countries. For example, the analysis of fallout ^{90}Sr transfer to humans was based on measurements in Argentina, Denmark, and New York City. For lack of more extensive data, these results were averaged and assumed to be generally applicable. Milk consumption has been reported for many other locations. For general assessment purposes, the Committee has used an average dietary intake of 500 kg a^{-1}.

153. For the analysis of exposures following the Chernobyl accident, the Committee compiled consumption data for all countries reporting first-year measurements. These values were as assessed by scientists of the various countries or, secondarily, derived from food balance considerations. The consumption rates, as given in the UNSCEAR 1988 Report [U4], are listed in Table 21. This listing is relatively extensive, allowing regionally relevant estimates to be derived. Some variations within geographic regions are fairly wide. Some of these differences might be explained by local habits. Other differences may result from inconsistencies in the definitions of the food categories, especially for leafy and other vegetables. Population-weighted average values from this listing [U4] are given in Table 21. These may be taken to be reasonable representative for very broad geographic regions. Changing dietary habits, however, require such food consumption data to be periodically updated.

154. Although many regional differences in consumption can be noted, the data seem to separate only very broadly into western and eastern countries. The western diet contains greater amounts of dairy products and meat. These foods are replaced by grain products, vegetables, and fish in Asian countries. The average value of consumption for the world would not apply to any individual and could only be used in some generic dose assessments. The consumption rates of children are less well known. In the UNSCEAR 1993 Report [U3], milk consumption was assumed to be 120 kg a^{-1} for infants and 110 kg a^{-1} for children. Other foods were assumed to be consumed at the rate of two thirds (children) or one third (infants) of the

adult values [C5]. This gives consistent and reasonable values to be used in dose assessments (see Table 13 of Annex B, "*Exposures from natural radiation sources*").

155. Drinking water intake has been estimated for reference individuals. For both water and beverages, the estimates are 500 l a^{-1} for adults, 350 l a^{-1} for children and 150 l a^{-1} for infants [I7]. Since the water balance is affected by ambient temperatures, regional estimates of these quantities should be established, if possible.

156. The consumption of foods from semi-natural and natural ecosystems, such as mushrooms and game, varies widely and is, in general, poorly known. Although these foods may comprise only a few percent (5–10 kg a^{-1}) of an individual's total annual dietary intake, such intake could be important for some radionuclides in certain times and places such as the arctic food chain (lichen-caribou/ reindeer-human) for both natural and fallout radionuclides and for consumers of game and forest mushrooms and berries for ^{137}Cs following the Chernobyl accident. Usually only a very small portion of a country's population will be significantly affected, so collective dose estimates are little modified. For further analyses of these situations, better data on the consumption of these foods are needed.

3. Methods for estimating exposures

(a) Atmospheric nuclear testing

157. To make reliable assessments of doses through the ingestion pathway of radionuclides released in atmospheric nuclear testing, extensive empirical data were compiled on the concentrations of the relevant radionuclides in different types of food and the diets of different population groups. The data were analysed in previous reports of the Committee, especially for ^{90}Sr and ^{137}Cs, which together with ^{14}C, are the main contributors to the ingestion dose commitments from this practice [U6, U7]. To evaluate the transfer coefficients, regression analyses were applied to models relating measured radionuclide concentrations in diet to the annual deposition density rates and the measured concentrations in relevant organs. Information on diet and deposition levels of other radionuclides are incomplete, so the P_{23} coefficients estimated for such radionuclides are less reliable than those available for ^{90}Sr and ^{137}Cs.

158. The empirical model used to relate the deposition density of a radionuclide, specifically ^{90}Sr or ^{137}Cs, to the integrated concentration in components of the diet or in total diet is the following

$$C_i = b_1 \dot{F}_i + b_2 \dot{F}_{i-1} + b_3 \sum_{n=1}^{\infty} e^{-\lambda n} \dot{F}_{i-n} \qquad (27)$$

where C_i is the concentration of the radionuclide in a food component or in the total diet in the year i due to the deposition density rate in the year i, F_i, in the previous year, F_{i-1}, and in all previous years, reduced by exponential decay. The exponential decay with decay constant λ reflects both radioactive decay and environmental loss of the radionuclide. The coefficients b_i and the parameter λ are determined by regression analysis of measured deposition and diet data.

159. The transfer coefficient from deposition to diet is given by

$$P_{23} = \int_0^{\infty} C(t)\,dt \;/\; \int_0^{\infty} \dot{F}(t)\,dt \quad \text{or} \quad \sum_{i=1}^{\infty} C_i / \sum_{i-1}^{\infty} \dot{F}_i \qquad (28)$$

From the above model, the transfer coefficient can be expressed as

$$P_{23} = b_1 + b_2 + b_3 e^{-\lambda m}/(1 - e^{-\lambda m}) \qquad (29)$$

where b_i are the transfer components per unit annual deposition: b_1 is the transfer in the first year, primarily from direct deposition; b_2 is the transfer in the second year from lagged use of stored foods and uptake from the surface deposit; and b_3 is the transfer via root uptake from the accumulated deposit. The units of P_{23} and b_i are Bq a kg^{-1} per Bq m^{-2}. In the exponential term, the unit for λ is a^{-1} and m is a constant equal to one year. The values of the parameters used are given in Table 22.

160. Results of regression fitting of this fallout model to monitoring data were presented in previous UNSCEAR Reports [U6, U7, U8]. Relatively minor adjustments in parameter values were needed in the fits to extended monitoring data, indicating, in particular, that the projections of long-term transfers are confirmed.

161. Adequate representations of transfers to the total diet or to separate components of the diet are obtained for relatively uniform deposition during the year, as occurred for fallout from atmospheric weapons testing. For deposition occurring within a much shorter time period, such as following the Chernobyl accident, the transfer is dependent on the particular agricultural conditions at the time of deposition and on short-term restrictions on certain foods in the diet that may have been imposed.

162. If P_{23} is multiplied by the individual annual consumption of food (kg a^{-1}), the transfer coefficient P_{24}, which relates to the intake of the radionuclide, is obtained. The transfer coefficients P_{45} then relate the intake amount to the dose (Sv Bq^{-1}). This is a committed dose that accounts for longer-term retention of the radionuclide in the body.

163. The Committee's earlier evaluations of transfer coefficients related the integrated concentration of the radionuclide in the body to the dose [U6]. For ^{90}Sr, the empirical relationship was as follows:

$$C_{b,i} = c\,C_{d,i} + g \sum_{m=0}^{\infty} e^{-\lambda_b m} C_{d,i-m} \qquad (30)$$

where $C_{b,i}$ and $C_{d,i}$ are the concentrations of ^{90}Sr in bone and diet in the year i and the parameters c and g may be related to short- and longer-term components of ^{90}Sr retention in bone. The exponential term accounts for radioactive decay and removal from the body. Average values derived for the parameters are listed in Table 22. This formulation is useful for determining the annual components of dose from a specific deposition occurrence.

164. The results of transfer coefficient evaluations for a number of radionuclides are listed in Table 23. For ^{90}Sr and ^{137}Cs, the values are the same as those previously derived [U3]. It should be stressed that the transfer coefficients P_{23}, P_{234} and P_{2345} are all calculated for an even distribution of the deposition throughout the year, as was the case for global fallout from atmospheric nuclear testing. If the deposition occurs during the winter season, the transfer coefficients are lower, and for a summer deposition they are higher than the value for the even distribution.

165. The transuranic radionuclides considered by the Committee in dose evaluation from atmospheric testing were ^{238}Pu, ^{239}Pu, ^{240}Pu and ^{241}Pu together with its decay product ^{241}Am. The empirical model described above, equation (27), has been used to relate the deposition amount to the integrated levels in diet. The lag term, however, was not included ($b_2 = 0$).

166. As the number of measurements of the annual ingestion intake, I_{ig}, of plutonium radionuclides were very few and covered only 11 years, the determination of λ is very uncertain; large variations in the value of λ result in only small variations in the value of I_{ig}. Taking λ to be very small, Bennett [B2] found the average solutions for b_1 and b_3 to be $3.3\ 10^{-2}$ Bq per Bq m^{-2} and $3.5\ 10^{-4}$ Bq per Bq m^{-2}, respectively, for $^{239,240}Pu$. The estimation of P_{234} depends on the real value of λ. It could be as low as $5\ 10^{-2}$ Bq per Bq m^{-2} if the availability of plutonium decreases with a mean residence time of 50 years ($\lambda = 0.02\ a^{-1}$) and as high as about 10 Bq per Bq m^{-2} for ^{239}Pu and 3 Bq per Bq m^{-2} for ^{240}Pu, if the availability of plutonium decreased only as a result of radioactive decay ($\lambda = 3\ 10^{-5}\ a^{-1}$ and $1\ 10^{-4}\ a^{-1}$). Aarkrog [A4] estimated the transfer of $^{239,240}Pu$ to bread, an important component of diet, to be $2\ 10^{-2}$ Bq per Bq m^{-2}. Until additional information becomes available, the geometric mean of the extremes for transfer to total diet can be assumed for the transfer coefficient P_{234}, namely, 0.7 Bq per Bq m^{-2}. This result corresponds to a mean residence time of $^{239,240}Pu$ in soil of about 100 years, the value that was also adopted in Annex C of the UNSCEAR 1982 Report [U6] for the mean residence time in soil of long-lived natural radionuclides released from industrial plants.

167. For ^{238}Pu, the above estimate of P_{234} using the 50–year residence time in soil is appropriate, considering the similar radioactive half-life of this isotope. Given the short half-life of ^{241}Pu (14.4 a), the value of P_{234} is dominated by the rate effect and is taken to be equal to $4\ 10^{-2}$ Bq per Bq m^{-2}. In the case of ^{241}Am, the formulation is complicated by the need to take the decay of ^{241}Pu into account. Using the equivalent of equation (29) and taking λ_s to be very small and b_1 to have the same value as that obtained for $^{239,240}Pu$, Bennett [B11] estimated b_3 to be equal to $8\ 10^{-4}$ Bq per Bq m^{-2}. This value is very uncertain, as only one measurement of the annual dietary intake of ^{241}Am has been reported, but it points to the possibility that americium contained in the soil may be slightly more available to plants than plutonium. The value of P_{234} can be roughly assessed to range from $6\ 10^{-2}$ Bq per Bq m^{-2} for a residence time of ^{241}Am in soil of 50 years to 0.7 Bq per Bq m^{-2} if the availability of ^{241}Am decreases only by radioactive decay. The geometric mean of this range is 0.2 Bq per Bq m^{-2}.

168. The estimated values of the transfer coefficients for the transuranic radionuclides are listed in Table 23. These estimates are about 20 times higher than those used previously by the Committee because of the higher values of the dose factors that have since been recommended by ICRP. The total dose from plutonium will, however, not be influenced by this change because the dominating pathway for plutonium is inhalation, and here the dose factors are reduced by a factor of 4 (for class Y = type S). The transfer coefficients for ^{241}Am are also listed in Table 23.

169. Curium is chemically very similar to americium, and it may be assumed that the transfer coefficients for the various curium isotopes can be calculated as for ^{241}Am, taking the half-lives of the curium isotopes into consideration. Curium-244, which has a half-life of 18.1 years, can thus be assumed to have a P_{234} coefficient equal to 0.04 Bq per Bq m^{-2} and with the dose factor $1.2\ 10^{-7}$ Sv Bq^{-1}, P_{2345} is estimated to be 5 nSv per Bq m^{-2}.

170. Less complete data are available for deriving transfer coefficients for ^{131}I, ^{140}Ba and ^{55}Fe. Radioiodine can be transferred rather quickly via the pasture–cow–milk chain to humans. Hence, although ^{131}I is a short-lived radionuclide (half-life: 8 days), it may contribute significantly to the dose in the first weeks after a release. P_{234} for ^{131}I was calculated to be 0.07 Bq per Bq m^{-2} from a P_{23} coefficient for milk of 0.63 mBq a l^{-1} per Bq m^{-2} and an average milk consumption rate of 0.3 l d^{-1}. The dose factor for ^{131}I ingestion by adults is $2.2\ 10^{-8}$ Sv Bq^{-1}, so P_{25} becomes 1.5 nSv per Bq m^{-2} for ^{131}I. For the age group 0–1 year, daily milk consumption is 0.9 l and the dose factor is $1.8\ 10^{-7}$ Sv Bq^{-1}, so in this case P_{25} becomes 37 nSv per Bq m^{-2}. A weighted average of P_{25} for all age groups for ingestion is 4.3 nSv per Bq m^{-2} for ^{131}I. Similar considerations were applied for ^{140}Ba. The estimates are included in Table 23.

171. The transfer coefficient P_{234} for ^{55}Fe was estimated in the UNSCEAR 1993 Report [U3] to be 10 Bq per Bq m^{-2}. For adults, P_{45} is 3.3 10^{-10} Sv Bq^{-1} for ^{55}Fe, and P_{2345} for the ingestion of ^{55}Fe becomes 3 nSv per Bq m^{-2}. It should be noted that P_{234} for ^{55}Fe also includes a contribution from consumption of fish, which are known to concentrate ^{55}Fe from seawater [I10]. Hence P_{234} for ^{55}Fe in the terrestrial environment is overestimated. Values of 6 Bq per Bq m^{-2} for P_{234} and 2 nSv per Bq m^{-2} for P_{2345}, assumed applicable for terrestrial pathways, have been inserted in Table 23.

172. Although not previously considered in exposure evaluations from ingestion, some limited data may be used to derive rough estimates of transfer coefficients for ^{54}Mn and ^{144}Ce. From measurements of ^{54}Mn in grain in 1962–1966 in localized areas in the northern hemisphere [A5], the transfer coefficient from deposition density to concentration in grain was estimated to be 0.025 Bq a kg^{-1} per Bq m^{-2}. Assuming an annual consumption of grain products of 80 kg and that all ^{54}Mn in the diet comes from grain products, P_{234} becomes 2 Bq per Bq m^{-2}. For adults, P_{45} is 7.1 10^{-10} Sv Bq^{-1} for ^{54}Mn, and P_{2345} for ingestion of ^{54}Mn becomes 1.4 nSv per Bq m^{-2}.

173. Cerium is relatively unavailable to plants. Assuming that a first-term component similar to that of plutonium applies and that there are no other terms because of the short half-life of ^{144}Ce, the transfer coefficient P_{234} would have the value 0.1 Bq per Bq m^{-2}. The further transfer coefficients have been added in Table 23.

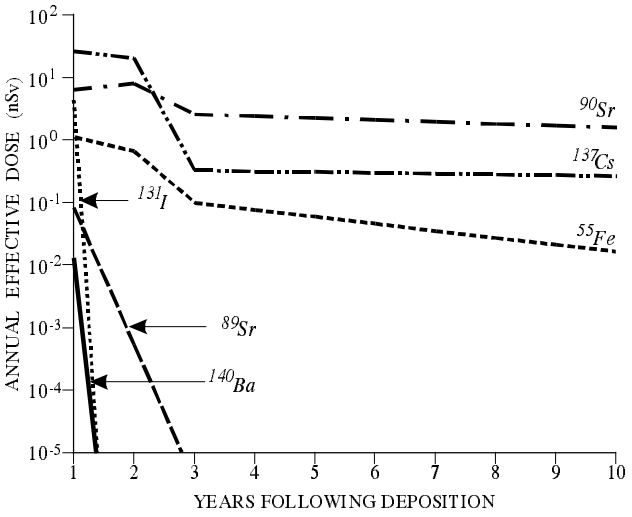

Figure IV. Ingestion exposure following unit deposition (1 Bq m^{-2}) of radionuclides.

174. The above analyses have been made to derive dose commitments per unit deposition density, but annual values of contributions to dose from a single deposition event may be useful and can also be provided. These values are given in Table 24. Because of the short half-lives of ^{131}I and ^{140}Ba, the dose is delivered within a few weeks of deposition and for ^{89}Sr, within a few months. All of the dose is delivered within one year of the deposition. There is no model for the transfer of ^{55}Fe to diet, but an approximation may be made on the pattern of transfer of longer-lived radionuclides to diet, namely significant transfer within the first two years following deposition and residual transfer over the remaining mean life of the radionuclide (see footnote to Table 24). The empirical models for ^{90}Sr and ^{137}Cs provide the time course of transfer to dose for annual periods following deposition of these radionuclides. The annual contributions to dose from ingestion for a period of 10 years following deposition are illustrated in Figure IV. The contributions beyond 10 years, which are significant only for ^{90}Sr and ^{137}Cs, are given in Table 24. The total of all annual contributions is equal to the dose commitment.

(b) Nuclear installations

175. Radionuclides released to the atmosphere from nuclear installations may contribute to exposures from ingestion in the local and regional areas surrounding the site. The concentrations of the radionuclides in the environment and the doses are too low to be measured, but they can be estimated with calculational methods.

176. The dispersion estimation method described in Section I.B.3 and applied to the external exposure pathway is also applicable to ingestion exposure, substituting in equation (24) the P_{25} transfer coefficients for ingestion intake. A more specific designation of the ingestion transfer coefficients is P_{2345}. The values in Table 23 are applicable also to the case of routine continuous or long-term averaged releases of radionuclides from nuclear installations. Several additional radionuclides not normally included in analysis of weapons fallout but present in releases from nuclear installations such as ^{59}Fe, ^{58}Co, ^{60}Co and ^{134}Cs have been added to Table 23.

177. The estimates of local and regional collective dose from ingestion per unit release of radionuclides from nuclear installations are presented in Table 25. The results should be adjusted if it is known that some portion of the diet is derived from non-local foods. Also the representative population density may not apply to specific sites. The representative values of population densities for various steps of the fuel cycle are given in Table 26.

178. Specific values given in Table 25 of the collective dose per unit release are needed in exposure evaluations for releases from separate fuel cycle installations. For the general category of particulates released from reactors, a representative composition may be assumed (see paragraph 96). The weighted average collective doses from ingestion per unit release of particulates are included in Table 25. Of course, the specific radionuclide weightings should be adjusted, if the exact composition of the release is known.

179. An alternative method, the specific-activity approach, is used to estimate doses from tritium and ^{14}C. In this approach the specific activity of ^{14}C, for example, in ingested food and water (activity per gram carbon) is assumed to be the same as the activity per gram carbon in

air at the point of interest. This is a good approximation for situations where rapid exchange occurs, such as between atmosphere and terrestrial biota, and the specific-activity model provides a good estimate of ^{14}C doses for chronic releases from nuclear facilities. However, it is necessary to know the carbon content of plants and animals to apply this approach rigorously. The specific activity in air is reflected in humans after about one year.

180. The specific-activity model for tritium is expressed in terms of the tritium to hydrogen atom ratio. For aqueous compartments (air moisture, plant water, soil water and so on), the constancy of this ratio is equivalent to assuming that the HTO concentration in Bq l^{-1} is constant. However, a strict specific-activity approach overestimates doses for tritium, since it assumes a level of equilibrium between tritium in the environment and in the atmosphere that is rarely achieved. Concentrations in precipitation, and therefore in soil, are lower than those in air, because the airborne plume is not always present when precipitation occurs. Concentrations in plants will be lower than those in air by an amount that depends on the transpiration rate. Concentrations in drinking water tend to be much less than air concentrations because of the large dilution that occurs in most drinking water sources. Concentrations in animals and humans reflect the concentrations in the food products and drinking water they ingest.

181. The general formula for the specific-activity approach to evaluate the collective dose is as follows:

$$S_i = \frac{5\,10^{-7}\ s\ m^{-3}}{3.15\ 10^{7}\ s\ a^{-1}} \frac{I_{g,n}}{C_{a,n}}\ d_{g,i} \left[N_1 \int_1^{50} x^{-1.4}\, 2\pi x\, dx + N_2 \int_{50}^{2000} x^{-1.4}\, 2\pi x\, dx \right] \quad (31)$$

where $I_{g,n}$ is the ingestion intake rate of the stable form of element n (kg a^{-1}); $C_{a,n}$ is the concentration of the stable form of element n in air (kg m^{-3}); $d_{g,i}$ is the effective dose per unit intake by ingestion of radionuclide i (Sv Bq^{-1}); and N_1 and N_2 are the population densities in the local and regional areas. For tritium, the exponent in the power function of distance should take the value 1.2 rather than 1.4 because of less local retention of deposited tritium.

(i) Tritium

182. Application of equation (31) for tritium requires estimates of the intake rates of both water-bound and organically bound hydrogen in foods and drinking water. To account for the fact that tritium concentrations in the various foodstuffs ingested are lower than the concentrations in moisture in air, $I_{g,n}$ is determined as follows:

$$I_{g,n} = \sum_i f_i U_i \quad (32)$$

where U_i is the intake rate of hydrogen from ingestion of food type i and f_i is the ratio of tritium concentration in food type i to the concentration in moisture in air.

183. Representative intake rates of plant foods, animal foods, and drinking water may be assumed to be 370, 170, and 500 kg a^{-1}, respectively. With typical water content of plant foods of 85% and of animal foods of 78% [D6] and the hydrogen content of water being 11.1%, the intake rates, U_i, of water-bound hydrogen are 35 kg a^{-1} in plant foods (370 × 0.85 × 0.111), 15 kg a^{-1} in animal foods (170 × 0.78 × 0.111), and 56 kg a^{-1} in drinking water (500 × 0.111).

184. In the organic matter of foods, the hydrogen content is, on average, 5.8% in plant foods and 8.4% in animal foods [D6]. The intake rates, U_i, of organically bound hydrogen are thus 3.2 kg a^{-1} in plant foods (370 × 0.15 × 0.058) and 3.1 kg a^{-1} in animal foods (170 × 0.22 × 0.084).

185. The value of f_i for plant foods is about 0.8 or less [D1, H3, H4, M4]. For drinking water the value of f_i is variable, depending on local conditions. At sites on large water bodies, where tritium enters only from the atmosphere, f_i tends to be less than 0.1 [L2, N6]. Larger values of f_i, even approaching 1.0, might apply to small water bodies, but the low volume or flow rate of such sources would limit the suitability of the site to supply drinking water. Larger values of f_i could also apply to sites downstream of liquid discharges of tritium [N6] or if groundwater had been contaminated. Both of these cases, however, do not pertain to atmospheric releases. For the present calculations, f_i for drinking water is assumed to be 0.1. The value for specific sites should be based on local conditions. For animals, it may be assumed that 40% of water intake is derived from drinking water [R1]. The value of f_i for animal foods is thus estimated to be 0.5 for combined intakes of drinking water and plants (0.4 × 0.1 + 0.6 × 0.8).

186. It will be assumed that the concentration of organically bound tritium (Bq l^{-1} water equivalent) is the same as the concentration of water-bound tritium in both plants and animals so that the same values of f_i apply to the aqueous and organic phases. The value of $I_{g,n}$ is then estimated to be 40 kg a^{-1} in water-bound form (0.8 × 35 + 0.5 × 15 + 0.1 × 56) and 4 kg a^{-1} in organically bound form (0.8 × 3.2 + 0.5 × 3.1).

187. The annual average content of water vapour in air is assumed to be 8.1 g m^{-3} [U6], implying that C_a for hydrogen is $9\,10^{-4}$ kg m^{-3}. Population densities surrounding the point of release are given above. The dose per unit intake, d_g, was previously taken to be $2.2\ 10^{-11}$ Sv Bq^{-1} for water-bound tritium [U4, U6], but the value now recommended is $1.8\ 10^{-11}$ Sv Bq^{-1} [I5]. The dose coefficient for organically bound tritium is $4.2\ 10^{-11}$ Sv Bq^{-1} [I5]. Applying these parameters in equation (31), recalling that for tritium the exponent in the power function is 1.2, and summing the water- and organically bound doses, the result is 2.1 man Sv PBq^{-1} (local plus regional exposure) (Table 25). The dilution factor and the dose per unit intake are lower than in previous

assessments by the Committee, and the allowance has been made for reduced environmental concentrations relative to moisture in air. These reductions are partially offset by the use of a smaller exponent for the decrease in air concentration with downwind distance and the separate consideration of organically bound tritium. The net result is a dose lower by about a factor of about 4 than the previously derived value of 9 man Sv PBq^{-1} [U4, U6]. Organically bound tritium contributes about 20% of the dose but would contribute more for diets high in grain or rice, which have high organic fractions.

(ii) Carbon-14

188. The dose from local and regional exposure to ^{14}C released to the atmosphere represents only a small proportion of the total dose commitment. The main significance of ^{14}C stems from its global dispersion and entry into the carbon cycle, leading to long-term exposure (see Section V.B). The local and regional collective dose commitment was previously assessed by the Committee using the specific-activity approach. The Committee assumed in its 1982 Report [U6] that the release of ^{14}C is in the form of CO_2 and the concentration of carbon in the atmosphere, C_a, is 0.16 g m^{-3}. A more recent, revised value is 0.18 g m^{-3} [T1]. The intake rate of carbon is 300 g d^{-1} by men and 210 g d^{-1} by women, averaging 93 kg a^{-1} intake by ingestion, I_g. The dose per unit intake of ^{14}C by ingestion is 5.8 10^{-10} Sv Bq^{-1} [I5]; the value formerly used was 5.6 10^{-10} Sv Bq^{-1} [U6]. It is assumed that, unlike tritium, all components of the diet attain the specific-activity level of air at the location of interest downwind from the source. Substituting these parameters into equation (31), the result is 270 man Sv PBq^{-1} (local plus regional exposure) (Table 25).

189. For both tritium and ^{14}C, the approximations of the specific-activity method are recognized. The assumption for ^{14}C that all intake attains the specific activity at the point of calculation is not realistic. For tritium, the concentrations in the environment, although allowed to differ from the concentration in air, are probably overestimates. For both radionuclides, the time distribution in the delivery of the dose must be ignored. The approach thus probably leads to overestimates of the doses. Nevertheless, the method has the advantage of being a simple approach that can be easily adjusted for alternative parameters that might more accurately reflect actual local conditions.

190. Significant doses from a short-term ^{14}C release will be received only in the year of the release. Carbon-14 doses arise only from ingestion and once the food crop of the year of release is consumed, there are no significant pathways for further exposure. Small amounts of ^{14}C deposited in the soil during the release may be re-emitted and taken up by plants, but concentrations would be very low and doses imparted by eating the plants would be insignificant compared with those received in the year of release.

C. RADIONUCLIDES RELEASED TO THE AQUATIC ENVIRONMENT

191. Radioactive contamination of the aquatic environment may result in ingestion doses by three pathways: drinking of freshwater from both surface and ground sources, consumption of biota living in the water, typically fish, and consumption of terrestrial foods that have been contaminated by the use of freshwater for irrigation, by the application of sediments as soil conditioners, or by the application of aquatic plants as fertilizer. Water consumed by animals may also form a pathway for the transfer of radionuclides to the human diet. Shoreline deposits of contaminated sediments can contribute to external exposures.

1. Transfer processes

192. Radioactive material released to the aquatic environment is transported and dispersed by advective and turbulent processes occurring in the water body. Interactions between radionuclides and suspended matter and sediments may remove radionuclides from the solution. It is convenient to consider separate categories of water bodies for modelling the behaviour of radioactive material: lakes, rivers, groundwater, coastal seas, and oceans.

(a) Lakes

193. Contaminants in lakes may occur in solution in the water phase or in the sediments. Most radionuclides occur in both phases, and the distribution factor K_d describes their partition between water and sediments. Lakes receive water from rivers, soil run-off and rainfall and lose water by outflows and evaporization. The mean residence time of the water in a lake depends on this in- and outflux of water to and from the lake. The mean residence time of the radionuclide in the water phase of a lake depends furthermore on the K_d for the radionuclide and its radioactive decay. The water chemistry of the lake (pH, mineral and organic matter content, and redox) influence K_d. These factors also influence the uptake of radionuclides in biota. Lakes that are low in nutrients usually show higher concentration factors from water to biota than nutrient-rich lakes.

(b) Rivers

194. Rivers may be considered as lakes with a high in- and outflux of water. Thus, the mean residence time of radionuclides in water in a river is usually significantly shorter than that in a lake for a similar volume of water, so lower concentrations are usually found in rivers than in lakes for the same input of radionuclides to the two systems. The amounts of water carried by a river may vary considerably throughout the year. In the spring, when the snow melts, the river may cover an area several times that covered in the dry season of the year. The flood land along a river may retain radionuclides carried by the river water,

and this contamination may be released to the river again in subsequent years. Accordingly, it is more complicated to model the behaviour of radionuclides in river systems than in lakes. Sediments in the river bed may, during flooding conditions, also be transported to new locations in the river system and eventually be carried to the sea. Sediments may also be disturbed by dredging and other activities.

(c) Groundwater

195. Lakes and rivers contain 0.3% and 0.003%, respectively, of the total freshwater inventory of the world [U14]. Ice sheets and glaciers contain 75% and groundwater the remaining part, i.e. about one fourth of all freshwater is present as groundwater. Groundwater is, in general, well protected against atmospheric radioactive contamination, because adsorption, chemical precipitation and ion exchange prevent or delay the migration of many radionuclides, such as ^{90}Sr, ^{137}Cs, and ^{239,240}Pu. But some radionuclides, especially those of a noncationic form, e.g. tritium, ^{99}Tc, and ^{129}I are not completely retained by the soil. Tritium in the form of HTO is particularly mobile and is readily measurable in young groundwaters (less than 30 years). Groundwater may be contaminated in connection with underground waste disposal. This has been seen, for instance, at the Hanford site in the United States, where liquid waste has been discharged to the ground, contaminating the groundwater, first of all with tritium. Underground nuclear explosions at, for example, the Nevada test site, contaminated groundwater with tritium [M9]. The contamination of groundwater by long-lived radionuclides may be of interest in connection with the permanent disposal of high-level waste in underground depositories.

(d) Marine waters

196. The total volume of the water in the ocean is $1.37\ 10^{21}$ l [K1], which is four orders of magnitude more water than found in rivers and lakes together. However, most of the water in the ocean belongs to the deep ocean which is not used by man for food production. Fish and other marine foods are mainly produced in the coastal seas, which have a mean depth of about 50 m and a volume of $1.37\ 10^{18}$ l, or 0.1% of the total water volume of the ocean.

197. Some coastal seas are much like closed systems, and the residence time of the water in such systems is relatively long. Other coastal waters have a more direct connection to the open ocean, and the mean residence time there is shorter. In the present context, the North Sea has been taken as a typical coastal sea, and the mean residence time of the water of the North Sea has been taken as representative of all coastal seas.

198. Unlike freshwater systems, where the composition of the water shows great variation, marine waters generally have the same mineral composition. The increase in salinity in the transition from river to sea causes a desorption of radionuclides from sediments. The decreased fixation in estuaries is partly counterbalanced by a lower uptake by biota.

2. Methods for estimating exposures

199. Dose assessments for radionuclides released to the aquatic environment require, in general, information on the activity of each radionuclide released, the volume of the receiving water into which the radioactive material is diluted, the concentration levels reached in fish and shellfish, the factors regarding removal to sediments and exchange rates of water bodies, and the number of individuals who use the water for drinking purposes and who consume fish.

200. The local and regional collective dose commitments from radionuclides in liquid effluents can be estimated using the expression

$$S_i^c = \frac{A_i}{V(\tau + \lambda_i)} \sum_k N_k f_{k,i} I_k d_i \qquad (33)$$

where A_i (Bq) is the activity of radionuclide i released to water; V (liters) is the volume of the receiving water; τ (a^{-1}) is the reciprocal of the mean residence time of a radionuclide in the receiving water assuming no decay (removal to sediments is incorporated implicitly in this value); λ_i (a^{-1}) is the radioactive decay constant for radionuclide i; N_k is the number of individuals for pathway k; $f_{k,i}$ (Bq kg^{-1} per Bq l^{-1}) is the concentration factor for an item in pathway k for radionuclide i; I_k (kg a^{-1}) is the individual consumption rate of pathway item k; d_i (Sv Bq^{-1}) is the effective dose per unit activity ingested.

201. The quantity $A/V(\tau + \lambda)$ (Bq a l^{-1}) is the integral concentration in water for release of an activity A (Bq) or, alternatively, the equilibrium concentration in water, $C_{w,i}$ (Bq l^{-1}), for a constant continuing release rate (Bq a^{-1}). The equilibrium concentration in fish or shellfish is $C_{w,i}f_{k,i}$ (Bq kg^{-1}), where $f_{k,i}$ is the appropriate freshwater or salt-water concentration factor.

202. For radionuclide releases to small volumes of water, the concentrations in water or fish may be high, but the population that can be served with drinking water or by fish production will be limited. For releases to larger water volumes, the concentrations will be less, but the populations involved will be correspondingly larger. In fact, the N_k/V relationship could be taken, in a crude approximation, as relatively constant, the inverse of which indicates the water use with regard to the specific pathway, k, of each individual in the population.

203. For the drinking water pathway, a value for the quotient V/N_k of $2.2\ 10^7$ l man^{-1} is assumed for estimating the collective dose commitments from generalized liquid releases. This value is assumed to be a global average and

is obtained from an estimated global total of $1.3\ 10^{17}$ l of freshwater in lakes ($1\ 10^{17}$ l) and rivers (annual flow $0.3\ 10^{17}$ l) [U7], serving a world population of $6\ 10^{9}$.

204. Average fish plus seafood consumption per individual is about 8 kg a^{-1}, ranging from 4 to 6 kg a^{-1} in the Near East and Africa to 10–14 kg a^{-1} in the Far East and Europe [I7]. It may be assumed that the annual consumption is 6 kg a^{-1} ocean fish, 1 kg a^{-1} freshwater fish and 1 kg a^{-1} shellfish. Total freshwater fish consumption by the world population is thus $6\ 10^{9}$ kg a^{-1}, which, when a correction is made for an edible weight of 50%, agrees with the estimated annual global harvest of 10^{10} kg landed weight [F2]. Dividing by the global freshwater volume given in the above paragraph, the result is $4.6\ 10^{-8}$ man kg a^{-1} l^{-1}, which will be assumed to be the factor $N_k I_k/V$ needed for estimating collective doses from freshwater fish consumption.

205. The annual global ocean fish and shellfish harvest is 10^{11} kg landed weight [F2], which is consistent with the ocean fish and shellfish consumption by the world population, $42\ 10^{9}$ kg a^{-1}. The catch mostly takes place within the continental shelf over an area of $27.5\ 10^{6}$ km^2 and with a mean depth of approximately 50 m [K1]. The volume of these waters is thus $1.4\ 10^{18}$ l. The factor $N_k I_k/V$ required for the salt-water fish and shellfish pathway is, therefore, $3\ 10^{-8}$ man kg a^{-1} l^{-1}. This is about 35 times higher than the factor used in the UNSCEAR 1977 Report [U7]. The mean residence time of the water over the continental shelf is assumed to be the same as that observed for the North Sea, i.e. approximately 3 years for ^{90}Sr and ^{137}Cs [N2] and 3.5 years for $^{239,240}Pu$ (first pass). Experience from Chernobyl has shown the turnover time of ^{137}Cs in freshwater systems to be 0.3 a^{-1}, i.e. similar to the turnover observed in coastal waters. This turnover rate is less by a factor of 3 than the value of 1 a^{-1} used in the UNSCEAR 1977 Report [U7]. For ^{90}Sr, the turnover rate in freshwater systems is 0.2 a^{-1}, somewhat less than for ^{137}Cs, owing to a lower sedimentation rate.

206. The specific-activity concepts for tritium and ^{14}C discussed above apply in aqueous systems as well as in terrestrial systems. HTO released to a water body is transported in the same way as other radionuclides but with the additional process of evaporation, which can have a large influence on HTO concentrations in some systems [H5]; neglecting this evaporation will result in overestimates of the tritium concentration. For an atmospheric release, concentrations in water bodies are usually much less than in air because of the large amounts of water available for dilution. Uptake of HTO by aquatic organisms is very quick: concentrations in tissue become equal to water concentrations within minutes or hours. Aquatic plants form organically bound tritium through photosynthesis. Fish and invertebrates also produce small amounts of organically bound tritium from the HTO in their bodies and can directly incorporate organically bound tritium taken up through ingestion.

207. The calculations made here of local and regional collective doses from tritium and ^{14}C in liquid effluents are based on equation (33) rather than on the specific-activity model. Because tritium concentrations in water and aquatic organisms are essentially the same, $f_{k,i} = 1$ for tritium. On the other hand, $f_{k,i}$ for ^{14}C is very high, since the carbon content of the organisms is much greater than the carbon content of water.

208. The parameters used and the estimates of collective dose per unit release of radionuclides to the aquatic environment are given in Tables 27 and 28. The estimates are the local and regional components of collective dose. Many radionuclides have been included that might have to be considered in specific circumstances. For releases of all radionuclides other than tritium in liquid effluents from reactors, it is useful to specify a representative composition, as was done for particulates in airborne effluents. The release composition can vary widely depending on the reactor type, the fuel integrity and the waste management practices. A representative composition is given in Table 29, which is derived from previously reported data [U3, U4]. Although these referred mainly to PWRs and BWRs, the composition can be taken to be reasonably applicable to all reactor types. For analysis of worldwide releases from reactors, the Committee has used an average of the results for releases to freshwater and to salt water. In this case, the estimated collective dose per unit release of the representative composition of radionuclides in liquid effluents is 330 man Sv PBq^{-1}. More appropriate selections and weightings of values can be made in applications to actual circumstances of releases from specific sources.

209. For many radionuclides, sediment removal considerations and radioactive half-lives limit the contributions to global collective doses. Only a few radionuclides achieve widespread, global dispersion, and these are considered in the following Chapter.

V. GLOBALLY DISPERSED RADIONUCLIDES

A. TRITIUM

210. Estimates of doses from globally dispersed tritium are required for three sources: natural occurrence, atmospheric nuclear testing, and nuclear power production. The most direct estimates of dose are obtained from measurements of the environmental concentrations of tritium, which have been made at a number of locations worldwide and from which individual doses from natural tritium and tritium produced in atmospheric testing may be inferred. Collective doses can be determined from an assumed variation of dose with latitude and the known population distribution. Doses from globally dispersed tritium arising from nuclear power production cannot be derived in this way, since the concentrations are undetectable beyond a few kilometres from the release point. Instead, the doses are estimated from model calculations.

211. The Committee based its estimate of the annual effective dose from natural tritium on measurements of the uniform levels of tritium in surface waters (and in the human body) prior to input from man-made sources. The estimated effective dose to individuals is 10 nSv a^{-1} [U7]. With reference to the total annual production of natural tritium of 72 PBq a^{-1} (see Annex B, "*Exposures from natural radiation doses*") and the present world population of 6 10^9, the collective dose per unit release is 6 10^9 × 10 nSv a^{-1} ÷ 72 PBq a^{-1} = 0.8 man Sv PBq^{-1}. Considering the population of each hemisphere (89% north, 11% south), the collective doses per hemispheric input are 1.5 man Sv PBq^{-1} for the northern hemisphere and 0.2 man Sv PBq^{-1} for the southern hemisphere.

212. The doses from tritium produced in atmospheric testing were estimated initially from measurements of the concentrations in surface waters [B3]. The estimated dose commitments were 20 μSv in the northern hemisphere and 2 μSv in the southern hemisphere [U7]. Based on estimated inputs of tritium into the atmosphere from the practice of 1.9 10^{20} Bq to the northern hemisphere and 0.5 10^{20} Bq to the southern hemisphere [U6] and applying the natural tritium dose/production rate ratio, the estimates of dose commitment were adjusted to 51 μSv and 14 μSv in the northern and southern hemispheres, respectively [U6]. These last values were derived from and correspond to the dose coefficients given at the end of the previous paragraph.

213. The models used to estimate the global doses from tritium similate the world hydrological cycle. Calculations are thereby made of the specific activity of tritium in the various global water pools. Most tritium is released to the atmosphere as HTO, and tritium gas (HT, T_2) is transformed in the soil to HTO. Tritium, therefore, follows the local and global water cycles. The hydrological models are invariably formulated in terms of compartments, in which the tritium is assumed to be instantaneously and uniformly mixed. Transfers between compartments are quantified using rate constants that are based on the known rates of water movement due to processes such as precipitation, evapotranspiration and run-off. Tritium concentrations in foodstuffs are assumed to equal concentrations in air moisture, soil water or surface water, depending on the model. The concentration of tritium in humans is calculated from an average of the concentrations in the sources of water ingested, weighted by the relative amount that each source contributes to intake. Several models of this kind exist, differing primarily in the number and size of compartments considered. The compartment approach is sufficient to calculate mean tritium concentrations over long times and large spaces. As well as providing estimates of doses from nuclear power production, the models can be used to confirm the doses from natural production and atmospheric testing deduced from observations.

214. The simplest model for estimating global tritium doses consists of single compartments representing the circulating waters of the hemispheres (to an ocean depth of 75 m). The model of Kelly et al. [K3], as implemented by NRPB and the Commissariat à l'Energie Atomique (CEA) [N2], used this basic approach and allowed for slow exchanges between the hemispheres and the deep oceans. For a release to the atmosphere or to surface waters, the collective dose per unit release was determined to be 0.028 man Sv PBq^{-1} relevant to a world population of 4 10^9. The Committee used this result in the UNSCEAR 1982 Report [U6] and adjusted it in the UNSCEAR 1988 Report [U5] to 0.032 man Sv PBq^{-1} for a population of 4.6 10^9. These results are probably underestimates of doses, because the tritium is mixed in large compartments that include the world's oceans and is diluted more than it would be in the terrestrial environment normally accessible by humans.

215. Improved estimates of the global dose from tritium are obtained using more realistic models developed by the NCRP [N3], Bergman et al. [B4] and Killough and Kocher [K2]. The seven compartments in the NCRP model represent atmospheric water, surface soil water, surface streams and freshwater lakes, groundwater, saline lakes and inland seas, the ocean surface and the deep ocean (Figure V). Water volumes and mean residence times of water in each compartment were estimated, together with fractional transfer rates for movement among the compartments. The volumes and transfer rates for the hemispheres and the world are listed in Table 30. The intake of tritium by man was calculated from the predicted environmental concentrations and the amount of water taken in through drinking and food ingestion. Eighty percent of drinking water was assumed to come from surface streams and freshwater lakes and 20% from deep

groundwater. Tritium concentration in plant water was assumed equal to $0.7\ C_a + 0.3\ C_s$, where C_a and C_s are the concentrations in air moisture and soil water, respectively. Although the NCRP model is not divided into latitude bands, it can be used to estimate doses from releases to different parts of the atmosphere by adjusting the size of the compartments to hemispheric or latitudinal water volumes.

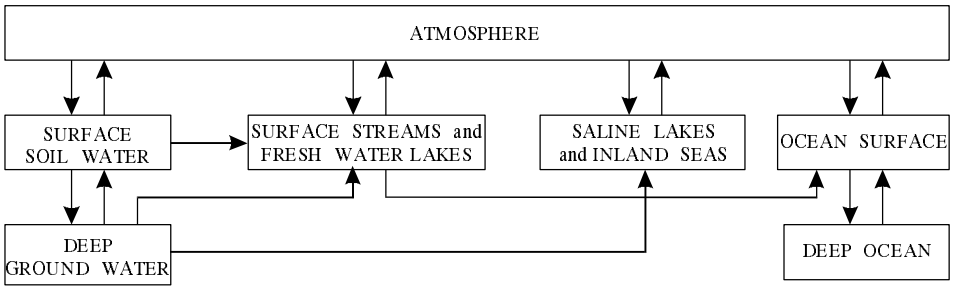

Figure V. Seven-compartment model of the hydrological cycle for global circulation of tritium [N3].

216. The model developed by Bergman et al. [B4] improved on the NCRP model by dividing all compartments into two latitude zones in each of the northern and southern hemispheres. It included a separate reservoir for organically bound tritium in terrestrial biota and was able to account for HT releases. The Killough and Kocher model [K2] separates the atmosphere into stratosphere and troposphere and further subdivided all atmosphere and ocean compartments, allowing the model to account for latitudinal inhomogeneities. Killough and Kocher noted that, without the stratospheric compartments, HTO entering the northern atmosphere is removed too rapidly to permit significant interhemispheric transport, and estimates of doses from atmospheric nuclear testing are unreliable. The use of a diffusive ocean module improved the ability of the model to estimate concentrations in the surface waters of the ocean.

217. The estimates of global collective doses from atmospheric tritium releases obtained with several models are shown in Table 31, together with the estimates based on natural tritium production. The estimates of the model calculations are those available in published reports, since with the exception of the NCRP model, the codes are too poorly documented to be run independently. The results are not easily compared since different source distributions are used. In general, however, there is a relatively good level of agreement. The dose estimates for releases to the global atmosphere are within a factor of 2, regardless of whether latitudinal zonation is considered or not. This level of agreement is maintained for releases to the northern hemisphere, but differences by a factor of 10 arise for releases to the southern hemisphere. The estimate of Bergman et al. [B4] for release to the global stratosphere (0.76 man Sv PBq^{-1}) agrees well with the global dose from natural production (0.8 man Sv PBq^{-1}). The NCRP results tend to be lower than those of the other models. Killough and Kocher [K4] found that the NCRP model underestimates observed freshwater concentrations of fallout tritium by about an order of magnitude and overestimates ocean concentrations by a factor of 3. Use of the NCRP model therefore likely leads to underestimates of the global collective doses from releases of tritium.

218. The global collective dose from near-surface atmospheric releases from nuclear installations is best obtained from model estimates of releases to the 30°–50° band of the troposphere in the northern hemisphere. The Killough and Kocher estimate of 2.3 man Sv PBq^{-1} is the most reliable in this regard. The NCRP model result is 0.7 man Sv PBq^{-1} for this case, but as noted above, this is probably an underestimate. The northern hemispheric estimate from natural production, 1.5 man Sv PBq^{-1}, may also reflect doses due to releases from nuclear installations, although it, too, may be an underestimate because the release is not confined to the latitude band in which the greatest population density is found.

219. Estimates of the global collective dose arising from releases to the ocean from nuclear installations are available from both the NCRP and Bergman et al. models. Both obtain estimates of doses that are about one tenth lower than those resulting from atmospheric releases. Taking the atmospheric result to be 2.3 man Sv PBq^{-1}, the dose from releases to the ocean becomes 0.2 man Sv PBq^{-1}.

220. Estimating the global distribution of tritium released from nuclear installations is a difficult task, and calculated doses contain an element of uncertainty. Based on a comparison of model estimates with observations [K2] and on the level of agreement among the estimates of the more reliable models, the true value of the global collective dose is believed to lie within a factor of 3 of the values given above. Much of the uncertainty is due to the large size of the compartments used in the models. The average concentrations assumed throughout these compartments are incompatible with the rapid changes in concentration that occur in the environment surrounding local sources and the non-uniform population density that actually exists.

221. The seven-compartment NCRP model may be used to demonstrate the spatial and temporal variations in the estimated tritium doses (Tables 32 and 33).These results should be considered illustrative only, since the NCRP model does not include latitudinal zonation and it tends to underestimate doses. However, it is well documented, transparent and accessible, and its estimates are probably realistic in terms of trends if not of magnitudes. Such results from use of other models are unavailable to the Committee.

222. Results of the seven-compartment model [N3] for releases to different parts of the atmospheric compartment are presented in Table 32. The slight difference between the northern and southern hemisphere reflects the fact that more of the global land surface (67%) is in the northern hemisphere and more of the global ocean surface (57%) is in the southern hemisphere.

223. The time course of the delivery of dose from tritium released to the atmosphere is indicated in Table 33. In this example, the seven-compartment model [N3] is applied to the 30°–50° latitude band of the northern hemisphere. The distribution of tritium within the seven compartments is indicated, with the decreasing total reflecting radioactive decay. The concentrations of tritium within the compartments may be determined by dividing by the water volumes: $1.7\ 10^{12}$ m^3 in the atmosphere, $1.4\ 10^{13}$ m^3 soil water, $5.6\ 10^{13}$ m^3 in freshwater, $9.9\ 10^{13}$ m^3 in saline water, $1.8\ 10^{15}$ m^3 in groundwater, $2.7\ 10^{15}$ m^3 in the ocean surface, and $1.3\ 10^{17}$ m^3 in the deep ocean. The concentration in humans is determined from the concentrations in the environment, weighted for fractional daily intake: 0.99 l from the atmosphere, 0.77 l from soil water (foods), 1.22 l from drinking water (80% from fresh water and 20% from groundwater) and 0.02 l from the ocean surface (seafood) for a total daily water intake of 3 l. The effective dose is largely received within the first few years of release, since much of the tritium is by then transferred to the oceans, from which less than 1% of the water intake by humans is derived.

224. From the above discussion it would appear that some consolidation of the results of tritium modelling would be useful in order to be somewhat more certain about the best estimates of global doses. On the whole, however, dose estimates can be selected that should be adequate for the general purposes. In summary, the estimates of the global collective doses per unit release of tritium from various sources are 0.8 man Sv PBq^{-1} for natural production, 1.5 and 0.2 man Sv PBq^{-1} for northern and southern hemisphere releases from atmospheric testing, and 2 and 0.2 man Sv PBq^{-1} for airborne and liquid discharges from nuclear installations.

B. CARBON-14

225. After its release, carbon is distributed among the various reservoirs of the global carbon cycle: the atmosphere, the terrestrial biosphere, the hydrosphere, and the lithosphere. The fluxes of radiocarbon and stable carbon between the different reservoirs are governed by the same exchange processes. Isotopic fractionation is negligibly small, within the other uncertainties involved. The total carbon content in the atmosphere is about $7.5\ 10^{17}$ g, of which the overwhelming bulk is present as CO_2. Exchange of carbon with the terrestrial biosphere and the hydrosphere is estimated to be $2\ 10^{17}$ g a^{-1}, with more than half going to the biosphere. The largest reservoir is the lithosphere ($7.2\ 10^{22}$ g), but the exchange rates between this and other compartments are extremely low.

226. Because of the long half-life of ^{14}C, its consequences must be evaluated through the collective effective dose commitment, which is complete about 50,000 years after the release. About 70% of the collective effective dose commitment will have been delivered by 10,000 years. Most models assume that the global population grows until the middle of the next century and then stabilizes at 10^{10} people.

227. As with tritium, the most direct estimates of global ^{14}C dose are obtained from environmental measurements. A natural production rate of 1 PBq a^{-1} leads to an individual effective dose rate of 12 μSv a^{-1}. This implies a collective effective dose commitment of 120,000 man Sv PBq^{-1} if it is assumed that the equilibrium population of the world of 10^{10} is achieved within a short time compared with the mean environmental lifetime of ^{14}C.

228. Recent interest in climate change has led to the development of many models to study the global circulation of stable carbon. For the most part, these models cannot be used to calculate global ^{14}C doses without major modifications. The models discussed below are those developed specifically to assess the doses from man-made sources of radiocarbon. As was the case for tritium, they are all compartment models of varying complexity. The assumption of instantaneous mixing in compartments is invalid in the short term for ^{14}C but is sufficiently accurate for long-term dose assessment. The models predict activities per gram carbon in each environmental compartment over time. Once mixing is achieved, the specific-activity model may be used to estimate collective dose commitments from ^{14}C. It is assumed that the specific activity of ^{14}C in the carbon ingested by humans is the same as that in the most relevant compartments for food intake (ground vegetation for terrestrial foods and relevant surface ocean compartments for marine foods).

229. The long time required to deliver the dose means that details of the source location and distribution are not as important for ^{14}C as they were for tritium. For all doses derived from model calculations, the release was assumed to be to a single compartment representing the global atmosphere, and the results apply equally to ^{14}C releases from natural production, atmospheric testing and nuclear power production.

230. The models for global carbon dose consider radiocarbon only in the form of $^{14}CO_2$, as this is the only form in which ^{14}C can enter the food chain. Thus, $^{14}CO_2$ is

the only direct contributor to ingestion dose, which makes up 99% of the total ^{14}C dose. Assuming that all radiocarbon is released as $^{14}CO_2$ will overestimate doses if hydrocarbons are also present in the emissions. However, the hydrocarbons will be oxidized to $^{14}CO_2$ within a few years [E6], and this can be taken into consideration.

231. The ability to make reasonable time-dependent estimates of regional and global ^{14}C fluxes and doses from arbitrary release locations over thousands of years requires a fairly sophisticated model. It should include the atmosphere, biosphere with multiple compartments, soil, oceans with multiple layers (a well-mixed upper layer, unstirred dense thermocline, and deep water), and, possibly, ocean sediments. Input fluxes should include both ^{14}C and ^{12}C, so that the specific activity of the radiocarbon can be calculated. Recent models incorporate the influx of ^{12}C from the burning of fossil fuels.

232. The Committee has used a variety of methods to estimate global ^{14}C doses for releases from nuclear installations. The estimates in the UNSCEAR 1977 Report [U7] were calculated using a model with compartments for terrestrial biosphere, atmosphere and short-term biosphere combined; surface ocean, thermocline layer in the ocean (a diffusive layer), and deep ocean. The parameters were adjusted to fit measurements of excess ^{14}C in the atmosphere and surface ocean from atmospheric testing. The incomplete (to 10^4 years) whole-body collective dose commitment was found to be 120,000 man Sv PBq^{-1} for a future world population of 10^{10} people. In the UNSCEAR 1982 and 1988 Reports [U4, U6], the NRPB/CEA [N2] model was used to estimate an incomplete collective effective dose commitment of 67,000 man Sv PBq^{-1} as an average for both atmospheric and aquatic releases for a population of 10^{10}, which was assumed constant during the integration period. A model developed by Emanuel et al. [E2] was used in the UNSCEAR 1993 Report [U3] that produced estimates of the incomplete collective dose commitment of 85,000 man Sv PBq^{-1} for a projected world population of 10^{10} people.

233. Global ^{14}C modelling has been further advanced by the work of Titley et al. [T1], and this model is recommended for use in ^{14}C dose assessments. It contains 23 compartments (Figure VI): atmosphere, ocean sediments, Antarctic Ocean (four layers), Atlantic Ocean (four layers), Pacific Ocean, including the Indian Ocean (three layers), Arctic Ocean (two layers), woody tree parts, non-woody tree parts, ground vegetation, decomposers, soil, and a compartment representing input from fossil fuel burning. The terrestrial portion of the model was adapted from Emanuel et al. [E1] with minor modifications to allow the transfer of soil via rivers to the ocean surface compartments. Exchanges between the atmosphere and the terrestrial biosphere are based on estimates of the photosynthetic uptake of carbon by plants and its release to the atmosphere by plants, animals, and soil by respiration [C4, E1].

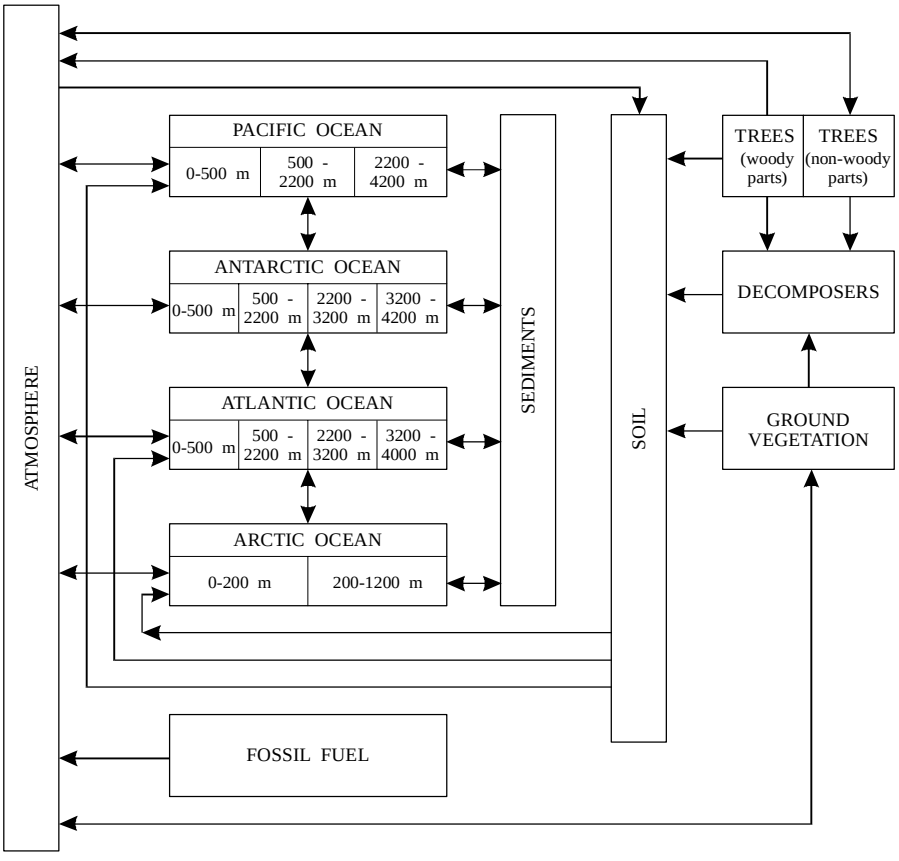

Figure VI. Compartment model for global circulation of carbon-14 [T1].

234. The ocean model in Titley et al. [T1] takes into account temperature changes, surface areas and varying amounts of ice cover in winter. Photosynthesis in the surface ocean layers and subsequent transfer of carbon down the water column was included and found to be important. In contrast, net sedimentation from water to the seabed was found to be a relatively insignificant process. The parameter values for the oceanic compartments were derived from several references [B6, M1, S3]. Exchanges between the oceanic and atmospheric compartments were based on estimates of the dissolution of CO_2 at the ocean/atmosphere interface using data from Mobbs et al. [M1] and Siegenthaler [S3]. Carbon dioxide is very soluble, and exchange with the atmosphere is rapid in open aerated water. The model was tested and validated against stable carbon distributions and ^{14}C specific activities arising from natural sources and atmospheric nuclear testing [T1].

235. The model of Titley et al. [T1] provides an estimate of the complete collective effective dose commitment per unit release: 109,000 man Sv PBq^{-1}. This is similar to the estimate of Emanuel et al. (108,000–114,000 man Sv PBq^{-1}), to the estimate derived from natural ^{14}C production (120,000 man Sv PBq^{-1}) and to previous UNSCEAR estimates. Indeed, the ^{14}C dose estimates of the many models in the literature are all very consistent. Killough and Rohwer [K5] found that the predictions of six models ranged over a factor of only 1.5. A similar range was found by Titley et al. [T1] in their comparison of four other models. Finally, McCartney et al. [M5] found less than a 15% difference in the results of three models. Killough and Rohwer [K5] attribute the consistency to the long half-life of ^{14}C relative to its rate of environmental transport, which makes the estimated dose commitments insensitive to the detailed structure of the models or to the values of the parameters used in them.

236. The collective dose coefficient of 109,000 man Sv PBq^{-1} was calculated with the assumption that the release is to the atmosphere, that the future world population stabilizes at 10^{10} people, and that the global inventory of stable carbon does not increase from its present value. Based on the values provided by the various models, there is a high probability that a range of 100,000–140,000 man Sv PBq^{-1} will encompass the actual collective dose under these conditions. Assuming fossil fuels continue to be burned at the present rate of $5\ 10^{15}$ g carbon per year until supplies are exhausted, the best estimate of the collective dose (from predictions of the Titley model) is 92,000 man Sv PBq^{-1}, with a range of 80,000–130,000 man Sv PBq^{-1}. Doses following a release to soils or surface oceans are about the same as those for an atmospheric release, but doses from release to deep oceans would be about 20% lower.

237. The time course of collective dose for a release of ^{14}C to the atmosphere or to the ocean surface is shown in Table 34. The equilibrium specific activities assuming fixed, stable carbon inventories match those of natural ^{14}C production, which is of the order of 1 PBq a^{-1}. Estimates of dose are given for a variable inventory of stable carbon caused by the burning of fossil fuels. About 9% of the complete dose commitment from a single release is delivered within 100 years, 23% within 1,000 years and 75% within 10,000 years.

C. IODINE-129

238. Because of its very long half-life ($1.57\ 10^7$ a), ^{129}I may become widely distributed in the global environment much like stable iodine, ^{127}I, over a long time. Whether released into the atmosphere or into the aquatic environment, ^{129}I will eventually reach the oceans in a time period presumably shorter than its half-life. Iodine is released from the ocean into the atmosphere as organic iodine (mostly as methyl iodide) [L1] as a consequence of microbial activity. The emitted organic iodine is decomposed by sunlight into inorganic iodine compounds. Both the organic and inorganic forms enter the terrestrial environment by the processes of wet and dry deposition [W1]. The deposition velocity of inorganic iodine onto vegetation is about two orders of magnitude higher than that of organic forms [N4]. The global iodine cycle and the dynamic behaviour of iodine in the environment is being further studied to improve the estimates of doses from ^{129}I releases.

239. Doses to humans from ^{129}I are delivered principally by its incorporation into the body by ingestion or inhalation. Iodine accumulates primarily in the thyroid, but the low specific activity of ^{129}I (6.55 MBq g^{-1}) limits the activity of the radionuclide that can be present in the gland [T1]. Artificially produced ^{129}I is released into the environment from nuclear installations, and small amounts were also released in atmospheric nuclear testing.

240. The behaviour of iodine in the terrestrial environment is influenced by many factors, e.g. soil type, microbial activity, and chemical form. It is known that stable iodine accumulates in soil; iodine concentrations in soil are 10–1,000 times higher than those of the parent rocks. The levels of ^{129}I in soils collected from the vicinity of nuclear reprocessing plants are markedly higher than the levels in other places [B5, M7, R6]. Vertical distribution of ^{129}I in soil showed that most of the radionuclide is retained in the surface layer (<10 cm). These observations indicate that the transfer of ^{129}I from the terrestrial environment to the ocean would occur only relatively slowly.

241. A model of the global iodine cycle was developed by Kocher [K8]. The environmental compartments assumed in the model are the atmosphere, hydrosphere, lithosphere, and terrestrial biosphere. It is estimated that the mean residence time of iodine in surface soil is of the order of 10,000 years and that the mixing of iodine throughout the ocean would require 1,000 years or more. Therefore, the

most important parameter for determining dose rates and cumulative doses following the release of ^{129}I is the 10,000-year mean residence time of iodine in the surface soil region. It thus appears that for a realistic long-term population dose assessment, a progression from local to regional to global-scale models would be required [K8]. When the released ^{129}I reaches equilibrium with stable iodine, the specific-activity method could be used in the assessment.

242. In the specific-activity approach, the activity concentration of ^{129}I per unit mass of ^{127}I is assumed to be the same in sea water and in the human thyroid. Assuming that the concentration of stable iodine per unit mass of thyroid is 80, 180, 300, and 600 μg g^{-1} at ages 6 months, 4 years, and 14 years and for adults, respectively, and using the age distribution given previously, a specific activity of 1 Bq per gram of stable iodine in the thyroid would lead to an age-weighted annual thyroid dose of 1.5 10^{-7} Gy. Since the sea contains 3.8 10^{16} g stable iodine (water mass of 6 10^{23} g and iodine concentration in water of 0.064 μg g^{-1}), a release of 1 PBq ^{129}I results in a long-term specific activity of 0.026 Bq g^{-1}. The collective thyroid dose commitment arising from the discharges of ^{129}I would be about 9 10^{8} man Gy PBq^{-1}, assuming a world population of 10^{10} and no sink for iodine in the environment.

243. The compartment model for the global circulation of iodine is shown in Figure VII. This represents a revision [T1] of the model described by Kocher [K8] and modified by Smith and White [S4]. The inventories of stable iodine in the model compartments and the fluxes between them were determined from environmental measurements and from the requirement for mass balance. Iodine-129 released into any compartment is assumed to be transported with stable iodine, and so the specific activity of ^{129}I can be determined for each compartment. Intake of ^{129}I occurs by inhalation and by the ingestion of water and terrestrial and marine foods.

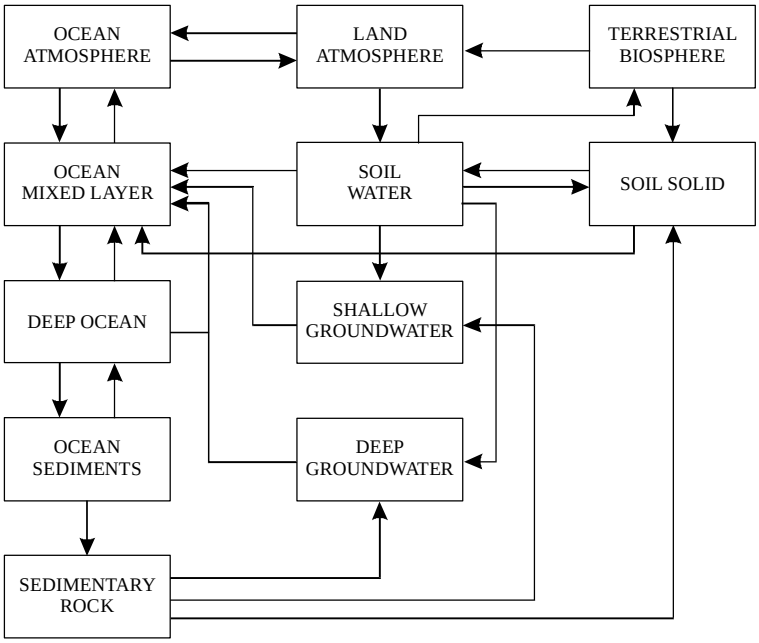

Figure VII. Compartment model for global circulation of iodine-129 [T1].

244. Important fluxes added to the global iodine model are from the sedimentary rock compartment to the two subsurface groundwater compartments and to the solid soil compartment, and from the solid soil compartment to the ocean mixed layer. This model includes the transfer of iodine from soil to the oceans and its subsequent movement back to soil from sedimentary rock. Titley et al. [T1] estimated the residence time of iodine in deep ocean waters to be 350 years and the flux of iodine from the deep ocean to the ocean mixed layer to be 2.3 10^{14} g a^{-1}. The amount of iodine transferred annually from the sedimentary rock compartment back to the solid soil compartment is estimated to be 1.8 10^{11} g a^{-1} [T1]. The mean residence times of iodine in the major compartments used in the model are 0.1 years in the ocean atmosphere, 0.09 years in the land atmosphere, 5.9 years in the ocean mixed layer, 19 years in the terrestrial biosphere, 3.6 10^{5} years in ocean sediments, 970 years in shallow subsurface region, and 38,000 years in deep subsurface region [T1].

245. The doses to individuals and collective doses following a release of ^{129}I can be calculated using the estimated time-dependent concentrations in the various compartments and either a pathway or a specific-activity analysis [K9]. The pathway analysis procedure involves identification of a number of exposure pathways; transfer coefficients are then used to estimate the movement of the radionuclide from the various compartments to humans. This approach requires considerable judgement because of the possible variations in the transfer coefficients and in the assumed intake rates, but the results are then quite realistic. The specific-activity approach is a means of

bypassing all the uncertainties and difficulties associated with the pathway analysis.

246. The pathway analysis method was adopted in the ^{129}I model under consideration. Five exposure pathways were assumed as follows: inhalation by humans from the land atmosphere, the daily intake rate of iodine being 0.29 μg d^{-1}; deposition from the land atmosphere onto food crops ingested directly by humans or by dairy and beef cattle and subsequently ingested by humans (6.6 μg d^{-1}); ingestion of land surface water directly by humans or by cattle (5.3 μg d^{-1}); ingestion of marine fish and shellfish from the ocean mixed layer (11 μg d^{-1}); root uptake from the surface soil region or from the soil water region into crops consumed by humans or by cattle subsequently ingested by humans. The intake of iodine through root uptake considers the concentration of iodine in the terrestrial biosphere and the ingestion of vegetables, cereals, all other foods, meat and milk. The daily intake of iodine through root uptake of iodine using average world consumption rates is 200 μg d^{-1}, and the total daily uptake of iodine is 220 μg d^{-1} [T1]. The calculation of effective dose utilized the following values: equivalent dose in the thyroid per unit intake 1.3 μSv Bq^{-1} (inhalation) and 2.1 μSv Bq^{-1} (ingestion) and tissue weighting factor 0.05.

247. A comparison of collective effective dose to the world population arising from a release of 1 TBq of ^{129}I during one year to the five different compartments calculated using this model [T1] is given in Table 35. At 10^8 years the collective effective dose for release to the land atmosphere (727 man Sv) and to solid soil (828 man Sv) are higher than the collective effective dose for release to the ocean compartments, 530, 469, and 469 man Sv for release to the ocean atmosphere, the ocean mixed layer and the deep ocean, respectively. The trend in collective effective doses from 50 years indicates higher amounts of iodine in the land atmosphere with negligible amounts in the deep ocean, but by 10^8 years the amounts in the deep ocean will have increased, while the amounts in the land atmosphere will have decreased. Thus the transfer to the deep ocean is much faster than the reverse process.

248. The long residence times of iodine in the solid soil compartment and the deep ocean compartment and the fact that a larger fraction of iodine in the ocean mixed layer compartments is transported downwards rather than to the atmosphere imply that it takes much longer for ^{129}I to reach the soil water compartment, from which most of the iodine intake by humans is derived. Collective effective doses estimated assuming that ^{129}I is discharged into the land atmosphere compartment are generally higher because of the direct connection between this compartment and the soil water compartment. The long residence time in the sedimentary rock compartment implies that iodine entering the sedimentary rock compartment is trapped there for a time of the same order as the half-life of ^{129}I before being cycled back to the soil.

CONCLUSIONS

249. In this Annex, the procedures used by the Committee for calculating doses from radionuclides in the environment are reviewed and updated. The radionuclides considered are those present either because they occur naturally or they have been released by anthropogenic practices. Although the calculational procedures are well established from extensive measurement and modelling experience, the increasing knowledge of transfer processes and radionuclide behaviour and better judgement of representative conditions allow the relevant parameters to be adjusted and the dose estimates to be improved.

250. For the Committee's purposes of estimating average doses under general conditions of release or presence of radionuclides in the environment, relatively simple calculational methods are sufficient. More detailed, time-dependent or otherwise complex methods have not been considered. For releases to the atmosphere or to the aquatic environment, such as those that occur from nuclear installations, average annual doses per unit release are estimated for populations in the local and regional areas. For longer-lived radionuclides that become widely dispersed, the average global doses are also evaluated. The main pathways of external irradiation, inhalation, and ingestion are considered.

251. The Committee has selected representative parameters to reflect the various conditions of release, environmental transport and behaviour, and the personal habits of intake and metabolism of the various radionuclides. These should provide reasonably accurate estimates of dose in many applications. Alternative selections of the parameters may lead to wide variations in the dose estimates. Therefore, the methods presented in this Annex should be used with caution. In particular, it is recommended that site-specific data should be used as appropriate and when available.

Table 1
Radiation weighting factors
[I1]

Type of radiation	*Energy range*	*Radiation weighting factor w_R*
Photons, electrons, muons	All energies	1
Neutrons Protons	<10 keV, >20 MeV >2 MeV	5 5
Neutrons	10-100 keV, >2-20 MeV	10
Neutrons Alpha particles, fission fragments, heavy nuclei	>0.1-2 MeV All energies	20 20

Table 2
Tissue weighting factors
[I1, I11]

Tissue or organ	*Weighting factor w_T*	
	1977	*1990*
Gonads	0.25	0.20
Breast	0.15	0.05
Colon		0.12
Red bone marrow	0.12	0.12
Lungs	0.12	0.12
Stomach		0.12
Urinary bladder		0.05
Liver		0.05
Oesophagus		0.05
Thyroid	0.03	0.05
Bone surface	0.03	0.01
Skin		0.01
Remainder	0.30 [a]	0.05 [b c]

a The value 0.06 is applied to the average dose among each of the five remaining organs or tissues receiving the highest dose, excluding the skin, lens of the eye, and the extremities.

b The remainder is composed of the following tissues and organs: adrenals, brain, extrathoracic region of the respiratory tract, small intestine, kidney, muscle, pancreas, spleen, thymus, and uterus.

c The value 0.05 is applied to the average dose to the remainder tissue group. However, when the most exposed remainder tissue or organ receives the highest committed equivalent dose of all organs, a weighting factor of 0.025 is applied to that organ and a weighting factor of 0.025 is applied to the average dose in the rest of the remainder.

Table 3
Values of the parameters used to evaluate vertical dispersion in the Gaussian plume model

Stability class	*Stability-dependent parameters*			
	a	*b*	*c*	*d*
A: extremely unstable	0.112	1.060	$5.38\ 10^{-4}$	0.815
B: moderately unstable	0.130	0.950	$6.52\ 10^{-4}$	0.750
C: slightly unstable	0.112	0.920	$9.05\ 10^{-4}$	0.718
D: neutral	0.098	0.889	$1.35\ 10^{-3}$	0.688
E: slightly stable	0.0609	0.895	$1.96\ 10^{-3}$	0.684
F: moderately stable	0.0638	0.783	$1.36\ 10^{-3}$	0.672
Roughness length (m)	*Roughness-dependent parameters*			
	p	*q*	*r*	*s*
0.01: Lawns, water bodies	1.56	0.048	$6.25\ 10^{-4}$	0.45
0.04: Plowed land	2.02	0.0269	$7.76\ 10^{-4}$	0.37
0.1: Open grassland	2.72	0	0	0
0.4: Rural areas, small villages	5.16	-0.098	18.6	-0.225
1.0: Forest, cities	7.37	-0.0957	$4.29\ 10^{3}$	-0.60
4.0: Cities with tall buildings	11.7	-0.128	$4.59\ 10^{4}$	-0.78

Table 4
Representative values of meteorological and release parameters

Parameter	*Units*	*Value*					
Effective release height (H)	m	30					
Direction frequency (f_i)	Dimensionless	0.083					
Sector width ($\Delta\theta$)	Radians	0.524					
Roughness length (z_0)	m	0.4					
		Stability class					
		A	*B*	*C*	*D*	*E*	*F*
Frequency of occurrence (f_i)	Dimensionless	0.05	0.10	0.20	0.30	0.20	0.15
Wind speed (u_i)	m s^{-1}	2	3	4	5	3	2
Inversion height (h_i)	m	2 000	1 500	1 200	800	400	200

Table 5
Dilution factors for the representative source and long-term average conditions

Downwind distance (km)	*Dilution factor (Bq m^{-3} per Bq s^{-1})*
0.5	$9.7\ 10^{-7}$
1	$5.3\ 10^{-7}$
2	$2.5\ 10^{-7}$
5	$7.1\ 10^{-8}$
10	$2.5\ 10^{-8}$
20	$8.7\ 10^{-9}$
50	$2.2\ 10^{-9}$
100	$7.6\ 10^{-10}$
200	$2.7\ 10^{-10}$
500	$6.7\ 10^{-11}$
1 000	$2.4\ 10^{-11}$
2 000	$8.2\ 10^{-12}$

Table 6
Analysis of variability in results of the Gaussian plume model

Varied parameter	*Value of varied parameter*	*Dilution factor at 1 km*	*Exponent of power function*
Wind speed (u_i)	Twice the representative value	$2.6\ 10^{-7}$	1.39
	Half the representative value	$1.1\ 10^{-6}$	1.55
Mixed layer height (h_i)	Twice the representative value	$5.3\ 10^{-7}$	1.55
	Half the representative value	$5.3\ 10^{-7}$	1.37
Frequency of stability class (f_i)	0.02, 0.05, 0.15, 0.55, 0.15,0.08 for classes A, B, C, D, E, F (high proportion of neutral classes)	$5.0\ 10^{-7}$	1.44
	0.02, 0.05, 0.15, 0.25, 0.3, 0.23 for classes A, B, C, D, E, F (high proportion of stable classes)	$5.8\ 10^{-7}$	1.49
	0.15, 0.25, 0.3, 0.2, 0.05, 0.05 for classes A, B, C, D, E, F (High proportion of unstable classes)	$4.4\ 10^{-7}$	1.41
Surface roughness (z_0)	0.1 m	$5.1\ 10^{-7}$	1.46
	1.0 m	$5.3\ 10^{-7}$	1.44
Effective release height (H)	0 m (ground-level release)	$1.2\ 10^{-6}$	1.70
	60 m	$2.2\ 10^{-7}$	1.36
Dry deposition velocity (v_d)	0.01 m s^{-1}	$5.2\ 10^{-7}$	1.74
	$4\ 10^{-4}$ m s^{-1}	$5.3\ 10^{-7}$	1.31
	0 m s^{-1}	$5.3\ 10^{-7}$	1.21
Wash-out coefficient (Λ)	$3\ 10^{-4}$ s^{-1}	$5.3\ 10^{-7}$	1.46
	$3\ 10^{-5}$ s^{-1}	$5.3\ 10^{-7}$	1.46
v_d and Λ	0	$5.3\ 10^{-7}$	1.21

Table 7
Outdoor effective dose rate to the adult per unit concentration in soil for the significant naturally occurring radionuclides

Radionuclide	*Effective dose rate per unit concentration (nSv h^{-1} per Bq kg^{-1})*		
	[B8] [a]	*[S10, S11]* [b]	*[E7]* [b]
^{40}K	0.029	0.030	0.033
^{232}Th series	0.46	0.42	0.51
^{238}U series	0.30	0.31	0.35

a Calculated as $E = X \times 0.0087$ Gy $R^{-1} \times 0.7$ Sv Gy^{-1}.
b $H_E + 0.01\ H_{skin}$.

Table 8
Conversion coefficients from air kerma to effective dose for terrestrial gamma rays
[S11]

Radionuclide	*Effective dose per unit air kerma (Sv Gy^{-1})*		
	Infants	*Children*	*Adults*
^{40}K	0.926	0.803	0.709
^{232}Th	0.907	0.798	0.695
^{238}U	0.899	0.766	0.672
Average	0.91	0.79	0.69

Table 9
Effective dose factors for cloud immersion

Radionuclide	*Effective dose per unit time-integrated concentration in air (nSv per Bq d m^{-3})*	
	[K7] [a]	*[E7]* [a]
^{89}Sr	0.033	0.039
^{90}Sr [b]	0.062	0.079
^{95}Zr	2.9	3.1
^{95}Nb	3.0	3.3
^{99}Mo [b]	1.1	1.1
^{103}Ru [b]	1.8	2.0
^{106}Ru [b]	0.87	0.99
^{110m}Ag [b]	11	12
^{115}Cd [b]	1.4	1.6
^{125}Sb [b]	1.6	1.8
^{127}Sb [b]	2.6	3.0
^{129m}Te [b]	0.29	0.32
^{131m}Te [b]	6.0	6.5
^{131}I	1.4	1.6
^{132}Te [b]	9.8	11
^{133}I	2.3	2.6
^{134}Cs	6.0	6.6
^{136}Cs	8.5	9.3
^{137}Cs [b]	2.2	2.4
^{140}Ba	0.72	0.76
^{140}La	9.3	10
^{141}Ce	0.29	0.31
^{143}Ce	1.0	1.1
^{144}Ce [b]	0.27	0.31
^{239}Np	0.64	0.68

a Calculated as $H_E + 0.01\ H_{skin}$.
b Decay products included.

Table 10
Collective effective doses from immersion exposure to noble gases released from reactors

Radionuclide	*Half-life*	*Effective dose rate per unit concentration* [a] *[E7] (nSv per Bq a m^{-3})*	*Collective dose per unit release* [b] *(man Sv PBq^{-1})*		
			Local	*Regional*	*Total*
^{41}Ar	1.827 h	2 080	0.90	0.005	0.90
^{85m}Kr	4.48 h	243	0.15	0.004	0.15
^{85}Kr	10.72 a	7.92	0.007	0.007	0.014
^{87}Kr	76.3 m	1 340	0.47	0.001	0.47
^{88}Kr	2.84 h	3 260	1.73	0.021	1.75
^{131m}Xe	11.9 d	13.8	0.012	0.009	0.021
^{133m}Xe	2.188 d	46.5	0.039	0.013	0.052
^{133}Xe	5.245 d	50.8	0.043	0.025	0.068
^{135m}Xe	15.29 m	653	0.062	- [c]	0.062
^{135}Xe	9.09 h	385	0.28	0.016	0.30
^{138}Xe	14.17 m	1 850	0.16	- [c]	0.16

a Evaluated as $H_E + 0.01\ H_{skin}$.
b Release from model reactor site; population density 400 km $^{-2}$ in local area (1-50 km) and 20 km $^{-2}$ in the regional area (50-2,000 km).
c Negligible result.

Table 11
Collective effective dose from immersion exposure for representative composition of noble gases released from reactors

Radio-nuclide	*Fractional release [U4]*		*Weighted collective dose per unit release [a] (man Sv PBq^{-1})*					
			PWRs			*BWRs*		
	PWRs	*BWRs*	*Local*	*Regional*	*Total*	*Local*	*Regional*	*Total*
^{41}Ar	0.005	0.03	0.005	0.00002	0.005	0.026	0.0001	0.027
^{85m}Kr	0.004	0.06	0.001	0.00002	0.0006	0.009	0.0002	0.010
^{85}Kr	0.016	0.01	0.0001	0.0001	0.0002	0.00009	0.00009	0.0002
^{87}Kr	0.009	0.08	0.004	0.00001	0.004	0.039	0.0001	0.039
^{88}Kr	0.004	0.15	0.007	0.00008	0.007	0.25	0.003	0.26
^{131m}Xe	0.006	0.03	0.00008	0.00006	0.0001	0.0004	0.0003	0.0007
^{133m}Xe	0.006	0	0.0002	0.00008	0.0003	0	0	0
^{133}Xe	0.81	0.20	0.035	0.020	0.055	0.009	0.005	0.014
^{135m}Xe	0.002	0.06	0.0001	-	0.0001	0.004	-	0.004
^{135}Xe	0.14	0.17	0.039	0.002	0.041	0.049	0.003	0.052
^{138}Xe	0.003	0.20	0.0005	-	0.0005	0.032	-	0.032
Total	1.0	1.0	0.09	0.02	0.11	0.42	0.01	0.43

a Collective dose per unit release (values from Table 10) multiplied by the release fraction. The results apply for the model reactor site.

Table 12
Transfer coefficients P_{25} from deposition to external exposure from radionuclides produced in atmospheric nuclear testing

Radionuclide	*Half-life*	*Absorbed dose rate in air per unit deposition density [a] (nGy a^{-1} per Bq m^{-2})*	*Effective dose commitment per unit deposition density [b] (nSv per Bq m^{-2})*
^{54}Mn	312.3 d	12.9	4.02
^{95}Zr [c]	64.02 d	45.0	2.87
^{103}Ru	39.26 d	10.8	0.42
^{106}Ru	373.6 d	3.21	1.19
^{125}Sb	2.76 a	6.52	6.54
^{131}I	8.02 d	13.0	0.10
^{137}Cs	30.07 a	8.89	97.2
^{140}Ba [c]	12.75 d	73.5	0.93
^{141}Ce	32.5 d	1.49	0.048
^{144}Ce [c]	284.9 d	0.693	0.20

a Ref. [B9]; converted with 0.869 rad per R and 0.01 Gy per rad. Assumes relaxation lengths of 0.1, 1, and 3 cm for radionuclides of half-lives <30 d, 30-100 d, and >100 d, respectively.

b Derived from absorbed dose rate in air times 0.7 Sv Gy^{-1} times 0.36 (occupancy/shielding factor) times mean-life (in years) of radionuclide (1.44 × half-life).

c Includes decay product.

Table 13
Annual components of dose from external exposure to radionuclides following a single deposition event

Year following deposition	*Annual effective dose per unit deposition density (nSv per Bq m^{-2})*									
	^{131}I	*^{140}Ba*	*^{141}Ce*	*^{103}Ru*	*^{95}Zr*	*^{144}Ce*	*^{54}Mn*	*^{106}Ru*	*^{125}Sb*	*^{137}Cs*
1	0.10	0.93	0.048	0. 42	2.82	0.12	2.23	0.59	1.45	2.21
2				0.001	0.054	0.05	0.99	0.30	1.13	2.16
3					0.001	0.020	0.44	0.15	0.88	2.11
4						0.008	0.20	0.08	0.68	2.07
5						0.003	0.09	0.039	0.53	2.02
6						0.001	0.039	0.020	0.41	1.97
7						0.0006	0.017	0.010	0.32	1.93
8						0.0002	0.008	0.005	0.25	1.88
9						0.0001	0.003	0.003	0.19	1.84
10							0.002	0.001	0.15	1.80
Total										
1-10	0.10	0.93	0.048	0.42	2.87	0.20	4.02	1.19	6.01	20.0
11-20							0.001	0.001	0.49	15.9
21-50									0.04	30.6
51-100										21.0
101-∞										9.69
Commitment										
1-∞	0.10	0.93	0.048	0.42	2.87	0.20	4.02	1.19	6.54	97.2

Table 14
Effective dose equivalent factors for external irradiation outdoors from deposited radionuclides
[B9, U4]

Radionuclide	*Effective dose equivalent per unit deposition density (nSv per Bq m^{-2})*	
	30 days to 1 year [a]	*After 1 year [b]*
^{103}Ru	0.691	0.00128
^{106}Ru	2.09	1.65
^{131}I	0.015	0.0
^{134}Cs	18.6	36.2
^{137}Cs	8.04	264

a Assumes relaxation length in soil of 1 cm.
b Assumes relaxation length in soil of 3 cm.

Table 15
Estimates of collective dose from external exposure per unit release of radionuclides from nuclear installations

Radionuclide	*Transfer coefficient P_{25} (nSv per Bq m^{-2})*	*Collective effective dose per unit release [a] (man Sv PBq^{-1})*		
		Local	*Regional*	*Total*
^{51}Cr	0.021	0.6	0.3	0.9
^{54}Mn	4.0	120	54	170
^{59}Fe	1.1	33	15	48
^{58}Co	1.1	32	14	46
^{60}Co	71	2 100	940	3 040
^{65}Zn	2.1	63	28	92
^{95}Zr [b]	2.9	85	38	120
^{103}Ru	0.42	13	5.7	18
^{106}Ru	1.2	35	16	51
^{124}Sb	2.3	69	31	100
^{131}I	0.10	3.1	1.4	4.5
^{134}Cs	18	540	240	780
^{136}Cs	0.92	27	12	40
^{137}Cs	97	2 890	1 300	4 190
^{140}Ba [b]	0.93	28	12	40
^{141}Ce	0.048	1.4	0.65	2.1
^{144}Ce [b]	0.20	5.8	2.6	8.5
^{241}Am	44	1 310	590	1 890
Particulates [c]		740	340	1 080

a Estimated from dispersion relationship: $5\ 10^{-7}\ x^{-1.4}$, where x is the distance from the release point; deposition velocity = 0.002 m s^{-1}; and population density = 400 km^{-2} in local area (1-50 km) and 20 km^{-2} in the regional area (50-2,000 km). Reduction due to urban runoff (factor of 0.75) also assumed.
b Includes decay product.
c Weighted average for assumed representative composition: 13% each of ^{54}Mn, ^{58}Co, ^{60}Co, ^{89}Sr, ^{134}Cs, ^{137}Cs, and ^{140}Ba; 0.9% each of ^{51}Cr, ^{59}Fe, ^{65}Zn, ^{90}Sr, ^{90}Y, ^{95}Zr, ^{124}Sb, ^{136}Cs, ^{141}Ce, and ^{144}Ce.

Table 16
Age-weighted breathing rate for the world population

Age group	*Breathing rate [a] (m^3 d^{-1})*	*Fraction of population [b]*	*Weighted rate (m^3 a^{-1})*
0-12 months	2.86	0.02	21
1-2 years	5.16	0.04	75
3-7 years	8.72	0.10	320
8-12 years	15.3	0.10	560
13-17 years	20.1	0.09	660
Adults (>17 years)	22.2	0.65	5 300
Sum		1.0	6 900

a Ref. [I4].
b Estimated from [U15].

Table 17
Committed effective doses per unit intake by inhalation of radionuclides
[I4, I5]

Radio-nuclide	*Absorption type* [a]	*Effective dose per unit intake (nSv Bq^{-1})*			*Radio-nuclide*	*Absorption type* [a]	*Effective dose per unit intake (nSv Bq^{-1})*		
		Infants [b]	*Children* [c]	*Adults* [d]			*Infants* [b]	*Children* [c]	*Adults* [d]
^{54}Mn	M	6.2	2.4	1.5	^{238}U series:				
^{55}Fe	M	1.4	0.62	0.38	^{238}U	M	9 400	4 000	2 900
^{89}Sr	M	24	9.1	6.1	^{234}U	M	11 000	4 800	3 500
^{90}Sr	M	110	51	36	^{230}Th	S	35 000	16 000	14 000
^{91}Y	M	30	11	7.1	^{226}Ra	M	11 000	4 900	3 500
^{95}Zr	M	16	6.8	4.8	^{210}Pb	M	3 700	1 500	1 100
^{95}Nb	M	5.2	2.2	1.5	^{210}Po	M	11 000	4 600	3 300
^{99}Mo	M	4.4	1.5	0.89					
^{103}Ru	M	8.4	3.5	2.4	^{232}Th series:				
^{106}Ru	M	110	41	28	^{232}Th	S	50 000	26 000	25 000
^{110m}Ag	M	28	12	7.6	^{228}Ra	M	10 000	4 600	2 600
^{115}Cd	M	4.8	1.7	0.98	^{228}Th	S	130 000	55 000	40 000
^{125}Sb	M	16	6.8	4.8					
^{127}Sb	M	7.3	2.7	1.7	^{235}U series:				
^{129m}Te	M	26	9.8	6.6	^{235}U	M	10 000	4 300	3 100
^{131m}Te	M	5.8	1.9	0.94	^{231}Pa	S	69 000	39 000	34 000
^{132}Te	M	13	4.0	2.0	^{227}Ac	M	550 000	260 000	220 000
^{131}I	F	72	19	7.4					
^{133}I	F	18	3.8	1.5	^{239}Np	M	4.2	1.4	0.93
^{134}Cs	F	7.3	5.3	6.6	^{238}Pu	M	74 000	44 000	46 000
^{136}Cs	F	5.2	2.0	1.2	^{239}Pu	M	77 000	48 000	50 000
^{137}Cs	F	5.4	3.7	4.6	^{240}Pu	M	77 000	48 000	50 000
^{140}Ba	M	20	7.6	5.1	^{241}Pu	M	970	830	900
^{140}La	M	6.3	2.0	1.1	^{241}Am	M	69 000	40 000	42 000
^{141}Ce	M	11	4.6	3.2					
^{143}Ce	M	3.9	1.3	0.75					
^{144}Ce	M	160	55	36					

a Absorption rates in body fluids are fast (F), moderate (M), and slow (S).
b From 1 year to 2 years.
c More than 7 years to 12 years.
d More than 17 years.

Table 18
Transfer coefficients for the inhalation pathway applicable to the deposition of radionuclides produced in atmospheric nuclear testing

Radionuclide	*Effective dose per unit intake* [a] P_{45} *(nSv Bq^{-1})*	*Effective dose per unit deposition density* [b] P_{245} *(nSv per Bq m^{-2})*
^{54}Mn	1.5	0.020
^{55}Fe	0.38	0.0050
^{89}Sr	6.1	0.080
^{90}Sr	36	0.47
^{91}Y	7.1	0.093
^{95}Zr	4.8	0.063
^{95}Nb	1.5	0.020
^{103}Ru	2.4	0.032
^{106}Ru	28	0.37
^{125}Sb	4.8	0.063
^{131}I	7.4	0.097
^{137}Cs	4.6	0.061
^{140}Ba	5.1	0.067
^{141}Ce	3.2	0.042
^{144}Ce	36	0.47
^{238}Pu	46 000	610
^{239}Pu	50 000	660
^{240}Pu	50 000	660
^{241}Pu	900	12
^{241}Am	42 000	550

a Absorption assumed to be Type F (fast) for ^{131}I and ^{137}Cs and Type M (moderate) for all other radionuclides.

b Equal to $P_{14}P_{45}/P_{12}$, where $P_{14} = 20$ m^3 d^{-1} (adult breathing rate) and $P_{12} = 0.0176$ m s^{-1} (the deposition velocity applicable to fallout from atmospheric testing).

Table 19
Estimates of collective dose from inhalation exposure per unit release of radionuclides from nuclear installations

Radionuclide	*Transfer coefficient P_{245} (nSv per Bq m^{-2})*	*Collective effective dose per unit release [a] (man Sv PBq^{-1})*		
		Local	*Regional*	*Total*
^{51}Cr	0.0037	0.15	0.07	0.21
^{54}Mn	0.17	6.9	3.1	10
^{55}Fe	0.044	1.7	0.8	2.5
^{59}Fe	0.43	17	7.6	25
^{58}Co	0.19	7.3	3.3	11
^{60}Co	1.2	46	21	66
^{65}Zn	0.19	7.3	3.3	11
^{89}Sr	0.71	28	13	41
^{90}Sr	4.2	165	74	240
^{90}Y	0.16	6.4	2.9	9.3
^{91}Y	0.82	33	15	47
^{95}Zr	0.56	22	9.9	32
^{103}Ru	0.28	11	5.0	16
^{106}Ru	3.2	130	58	190
^{124}Sb	0.74	29	13	43
^{131}I	0.86	34	15	49
^{134}Cs	0.76	30	14	44
^{136}Cs	0.14	5.5	2.5	8.0
^{137}Cs	0.53	21	9.5	31
^{140}Ba	0.13	5.0	2.3	7.3
^{141}Ce	0.37	15	6.6	21
^{144}Ce	4.2	165	74	240
^{238}Pu	5 320	211 000	95 000	306 000
^{239}Pu	5 790	229 000	103 000	332 000
^{240}Pu	5 790	229 000	103 000	332 000
^{241}Pu	100	4 130	1 860	5 990
^{241}Am	4 860	193 000	86 700	279 000
Particulates [b]		23	10	33

a Estimated from dispersion relationship: $5\ 10^{-7}\ x^{-1.4}$, where x is the distance from the release point; deposition velocity = 0.002 m s^{-1}; and population density = 400 km^{-2} in local area (1-50 km) and 20 km^{-2} in the regional area (50-2,000 km).

b Weighted average for assumed representative composition: 13% each of ^{54}Mn, ^{58}Co, ^{60}Co, ^{89}Sr, ^{134}Cs, ^{137}Cs, and ^{140}Ba; 0.9% each of ^{51}Cr, ^{59}Fe, ^{65}Zn, ^{90}Sr, ^{90}Y, ^{95}Zr, ^{124}Sb, ^{136}Cs, ^{141}Ce, and ^{144}Ce.

Table 20
Committed effective doses per unit intake by ingestion of natural radionuclides
[I5]

Radionuclide	*Fractional absorption*	*Effective dose per unit intake (nSv Bq^{-1})*		
		Infants [a]	*Children* [b]	*Adults* [c]
^{3}H (water)	1.0	0.048	0.023	0.018
^{3}H (organic)	1.0	0.12	0.057	0.042
^{7}Be	0.005	0.13	0.053	0.028
^{14}C	1.0	1.6	0.80	0.58
^{22}Na	1.0	15	5.5	3.2
^{40}K	1.0	42	13	6.2
^{238}U series:				
^{238}U	0.02	120	68	45
^{234}U	0.02	130	74	49
^{230}Th	0.0005	410	240	210
^{226}Ra	0.2	960	800	280
^{222}Rn [d]		23	5.9	3.5
^{210}Pb	0.2	3 600	1 900	690
^{210}Po	0.5	8 800	2 600	1 200
^{232}Th series:				
^{232}Th	0.0005	450	290	230
^{228}Ra	0.2	5 700	3 900	690
^{228}Th	0.0005	370	150	72
^{235}U series:				
^{235}U	0.02	130	71	47
^{231}Pa	0.0005	1 300	920	710
^{227}Ac	0.0005	3 100	1 500	1 100

a From 1 year to 2 years.
b More than 7 years to 12 years.
c More than 17 years.
d Ref. [N5].

Table 21
Food consumption rates by individuals [a]

Country / region	*Population*	*Consumption rate (kg a^{-1})*					
		Milk	*Grain*	*Leafy vegetables*	*Fruit/ vegetables*	*Meat*	*Total*
North Europe							
Denmark	5.11	173	80	18	150	66	487
Finland	4.87	263	73	6 [b]	169	71	582
Norway	4.16	202	65	37	120	76	500
Sweden	8.35	222	77	36	121	56	512
Central Europe							
Austria	7.56	145	66	71	136	99	517
Czechoslovakia	15.48	134	132	25	107	86	484
Germany	77.66	109	84	28	145	63	429
Hungary	10.62	185	110	25	160	80	560
Poland	37.46	160	180	20	132	67	559
Romania	22.73	150	190	40	240	86	706
Switzerland	6.49	180	99	29	230	110	648
West Europe							
Belgium	9.86	180	65	55	150	40	490
France	53.6	130	84	84	132	73	503
Ireland	3.54	163	68	40	69	50	390
Luxembourg	0.37	110	95	33	150	88	476
Netherlands	14.49	145	65	65	135	70	480
United Kingdom	55.87	163	68	40	100	71	442
South Europe							
Bulgaria	8.89	123	179	20	76	64	462
Greece	9.83	80	100	30	250	60	520
Italy	56.91	90	110	50	150	60	460
Portugal	9.94	45	125	113	105	42	430
Spain	37.3	104	88	124	132	62	510
Yugoslavia	22.49	146	146	55	128	55	530
USSR	279	332 [b]	133	37	118	63	683
West Asia							
Cyprus	0.64	83	94	87	315 [b]	83	662
Israel	3.87	120	130	140	190	60	640
Syrian Arab Rep.	8.98	70	190	30	340 [b]	22	652
Turkey	52	125	200	100	150	40	615
East Asia							
China	1046.4	5 [b]	229	29	173	30	466
India	750.9	39	183	28	89	5 [b]	344
Japan	121.0	50	193	30	180	120	573
North America							
Canada	25.4	181	93	21	301 [b]	130	726
United States	238.7	174	91	25	260	146	696
Average values [c]							
Countries of East and West Asia		25	210	30	140	25	430
Countries of Europe, USSR, and North America		200	110	40	165	85	600
World		85	170	35	150	50	490

Table 21, continued

Country / region	*Population*	*Consumption rate (kg a^{-1})*					
		Milk	*Grain*	*Leafy vegetables*	*Fruit/ vegetables*	*Meat*	*Total*
Representative values [d]							
Countries of East and West Asia		90	210	30	140	60	530
Countries of Europe, USSR and North America		150	110	40	170	85	555
World		120	170	35	150	70	545

a Population and consumption rates valid for 1986 [U4].
b Unusually high or low values.
c Average values are population-weighted results.
d Rounded, generic values (unusually high and low values excluded).

Table 22
Parameters of empirical models for transfer of ^{90}Sr and ^{137}Cs from deposition to diet to dose [a]

Pathway	*Transfer parameter*	*^{90}Sr*	*^{137}Cs*
Deposition to diet	b_1 (Bq a kg^{-1} per Bq m^{-2})	0.001	0.0038
	b_2 (Bq a kg^{-1} per Bq m^{-2})	0.001	0.0029
	b_3 (Bq a kg^{-1} per Bq m^{-2})	0.00011	0.000052
	λ' (a^{-1})	0.06	0.03
	P_{23} (Bq a kg^{-1} per Bq m^{-2})	0.0038	0.0084
Diet to body	c (Bq a kg^{-1} per Bq a kg^{-1})	17.5	
	g (Bq a kg^{-1} per Bq a kg^{-1})	3.7	
	λ_b (a^{-1})	0.13	
	P_{34} (Bq a kg^{-1} per Bq a kg^{-1})	48	2.6
Body to dose	P_{45} (nSv per Bq a kg^{-1})	290	2 500
Diet to intake [b]	P_{34} (Bq per Bq a kg^{-1})	500	500
Diet to dose	P_{45} (nSv per Bq)	28	13
Deposition to dose	P_{2345} (nSv per Bq m^{-2})	53	55

a Annual dose in a specific year is the deposition density of ^{90}Sr or ^{137}Cs in that year times the annual component of P_{23} times the annual component of P_{34} times P_{45} plus the contribution from intake in earlier years, which equals the residual body burden (for ^{90}Sr) reduced by exponential decay and removal ($e^{-\lambda b}$) times P_{45}.
b Assumes consumption intake of food of 500 kg a^{-1}.

Table 23
Transfer coefficients for radionuclides in the ingestion pathway

Radionuclide	*Deposition to diet* P_{23} *(mBq a kg^{-1} per Bq m^{-3})*	*Deposition to intake* [a] P_{234} *(Bq per Bq m^{-2})*	*Intake to dose* [b] P_{45} *(nSv Bq^{-1})*	*Deposition to dose* P_{2345} *(nSv per Bq m^{-2})*
^{51}Cr	7 [c]	0.56	0.038	0.02
^{54}Mn	25 [c]	2	0.71	1.4
^{55}Fe		6	0.33	2.0
^{59}Fe		0.76	1.8	1.4
^{58}Co	26 [c]	2.1	0.74	1.6
^{60}Co	36 [c]	2.9	3.4	9.9
^{65}Zn	45 [c]	3.6	3.9	14
^{89}Sr		0.03	2.6	0.08
^{90}Sr	3.8	1.9	28	53
^{95}Zr	1.3 [c]	0.1	0.95	0.10
^{95}Nb	0.9 [c]	0.07	0.58	0.04
^{124}Sb	13 [c]	1	2.5	2.5
^{131}I	0.6 [d]	0.07	61 [e]	4.3
^{134}Cs	4	2	19	38
^{136}Cs	0.6	0.3	3	0.90
^{137}Cs	8.4	4.2	13	55
^{140}Ba		0.005	2.6	0.013
^{141}Ce	0.9 [c]	0.07	0.71	0.05
^{144}Ce	1.3 [c]	0.1	5.2	0.52
^{238}Pu		0.05	230	12
^{239}Pu		0.7	250	180
^{240}Pu		0.7	250	180
^{241}Pu		0.04	4.8	0.19
^{241}Am		0.2	200	40
^{244}Cm		0.04	120	5

a May be derived from P_{23} by multiplying by total dietary consumption of 500 kg a^{-1}.
b To adults unless otherwise stated.
c To grain. To derive P_{24}, grain consumption of 80 kg a^{-1} has been assumed.
d For milk. To derive P_{24}, milk consumption of 0.3 l d^{-1} has been assumed.
e Population-weighted value.

Table 24
Annual components of dose from ingestion exposure to radionuclides following a single deposition event

Year following deposition	*Annual effective dose per unit deposition density (nSv per Bq m^{-2})*					
	^{131}I	^{140}Ba	^{89}Sr	^{55}Fe [a]	^{90}Sr	^{137}Cs
1	4.2	0.013	0.08	1.00	6.15	24.7
2			0.0005	0.60	7.73	19.2
3				0.089	2.47	0.32
4				0.069	2.30	0.31
5				0.054	2.14	0.30
6				0.042	1.99	0.29
7				0.033	1.86	0.28
8				0.025	1.73	0.27
9				0.020	1.62	0.27
10				0.015	1.51	0.26
Total						
1-10	4.2	0.013	0.08	1.95	29.5	46.2
11-20				0.049	10.7	2.2
21-50				0.004	10.3	3.7
51-100					1.9	2.0
101-∞					0.10	0.49
Commitment						
1-∞	4.2	0.013	0.08	2.0	53	55

a A transfer model does not exist. Using ^{137}Cs as a guide, it is assumed that 50% of commitment arises in first year after deposition, 30% in second year, and remainder at uniform rate over the mean life of ^{55}Fe.

Table 25
Estimates of collective dose from ingestion exposure per unit release of radionuclides from nuclear fuel cycle installations

Radionuclide	*Transfer coefficient P_{2345} (nSv per Bq m^{-2})*	*Collective effective dose per unit release (man Sv PBq^{-1}) [a]*		
		Local	*Regional*	*Total*
^{3}H [b]		1.1	1.0	2.1
^{14}C [b]		190	80	270
^{51}Cr	0.021	0.8	0.4	1.2
^{54}Mn	1.4	56	25	82
^{55}Fe	2.0	79	36	110
^{59}Fe	1.4	54	24	79
^{58}Co	1.6	62	28	89
^{60}Co	9.9	390	180	570
^{65}Zn	14	560	250	810
^{89}Sr	0.078	3.1	1.4	4.5
^{90}Sr	53	2 110	950	3 060
^{95}Zr	0.10	3.8	1.7	5.5
^{124}Sb	2.5	99	45	144
^{131}I	4.3	170	76	250
^{134}Cs	38	1 510	680	2 180
^{136}Cs	0.90	36	16	52
^{137}Cs	55	2 160	970	3 140
^{140}Ba	0.013	0.5	0.2	0.7
^{141}Ce	0.050	2.0	0.9	2.9
^{144}Ce	0.52	21	9.3	30
^{238}Pu	12	460	210	660
^{239}Pu	180	6 930	3 120	10 100
^{240}Pu	180	6 930	3 120	10 100
^{241}Pu	0.19	7.6	3.4	11
^{241}Am	40	1 580	710	2 300
Particulates [c]		570	260	830

a Population density: local (1-50 km): 400 persons km ^{2}; regional (50-200 km) 20 persons km $^{-2}$.
b Doses estimated using specific-activity model.
c Weighted average for assumed representative composition: 13% each of ^{54}Mn, ^{58}Co, ^{60}Co, ^{89}Sr, ^{134}Cs, ^{137}Cs, and ^{140}Ba; 0.9% each of ^{51}Cr, ^{59}Fe, ^{65}Zn, ^{90}Sr, ^{90}Y, ^{95}Zr, ^{124}Sb, ^{136}Cs, ^{141}Ce, and ^{144}Ce.

Table 26
Population densities surrounding nuclear fuel cycle installations

Country / region	*Area*	*Population density surrounding nuclear fuel cycle sites (inhabitants km^{-2})*		
		Uranium mining	*Fuel fabrication*	*Reactors*
World average [a]	Local [b]	3		400
	Regional [c]	25	25	20

a Representative values used in UNSCEAR assessments.
b 0-100 km for mining; 0-50 km for reactors.
c 100-2,000 km for mining and fuel fabrication; 50-2,000 km for reactors.

Table 27
Collective dose per unit release of radionuclides in liquid effluents to fresh water

Radio-nuclide	*Half-life*	*Time integral of unit activity in water* [a] *(Bq a)*	*Drinking water treatment removal factor*	*Concentration factor for fish* [b] *(Bq kg^{-1} fish per Bq l^{-1} water)*	*Dose per unit activity ingested (nSv Bq^{-1})*	*Collective dose per unit activity released (man Sv PBq^{-1})*		
						Drinking water	*Fish*	*Total*
^{3}H	12.26 a	3.90	1	1	0.018	1.3	0.003	1.3
^{14}C	5 730 a	5.00	1	50 000	0.58	54	6 690	6 740
^{24}Na	14.36 h	0.0024	0.5	20	0.43	0.009	0.0009	0.010
^{35}S	87.5 d	0.32	0.5	800	0.13	0.39	1.6	1.9
^{45}Ca	162.2 d	0.57	0.5	20	0.71	3.7	0.37	4.1
^{51}Cr	27.7 d	0.11	0.5	200	0.038	0.038	0.038	0.075
^{54}Mn	312.1 d	0.99	0.5	400	0.71	6.5	13	19
^{55}Fe	2.73 a	2.20	0.5	200	0.33	6.7	6.7	13
^{57}Co	271.8 d	0.88	0.5	300	0.21	1.7	2.6	4.3
^{58}Co	70.8 d	0.27	0.5	300	0.75	1.8	2.8	4.6
^{59}Fe	44.5 d	0.17	0.5	200	1.8	2.8	2.8	5.6
^{60}Co	5.271 a	3.02	0.5	300	3.4	95	140	240
^{65}Zn	244.3 d	0.81	0.5	1 000	3.9	29	150	180
^{89}Sr	50.5 d	0.19	0.5	60	2.6	4.6	1.4	6.0
^{90}Sr	28.78 a	4.46	0.5	60	28	1 150	350	1 500
^{95}Zr	64.02 d	0.24	0.5	300	0.96	2.1	3.2	5.3
^{95}Nb	34.98 d	0.13	0.5	300	0.59	0.73	1.1	1.8
^{97}Zr	16.9 h	0.0028	0.5	300	2.1	0.054	0.081	0.13
^{99}Mo	2.75 d	0.011	0.5	10	0.6	0.060	0.003	0.063
^{103}Ru	39.26 d	0.15	0.5	10	0.73	1.0	0.051	1.1
^{106}Ru	373.6 d	1.14	0.5	10	7.0	74	3.7	77
^{110m}Ag	249.8 d	0.82	0.5	5	2.8	21	0.53	22
^{113}Sn	115.1 d	0.42	0.5	3 000	0.73	2.8	42	45
^{122}Sb	2.73 d	0.011	0.5	100	1.7	0.17	0.084	0.25
^{124}Sb	60.2 d	0.23	0.5	100	2.5	5.2	2.6	7.9
^{125}Sb	2.76 a	2.22	0.5	100	1.1	23	11	34
^{129}I	1.6 10^{7} a	5.00	0.8	40	110	8 120	1 020	9 140
^{131}I	8.02 d	0.032	0.8	40	22	10	1.3	12
^{132}Te	3.2 d	0.013	0.5	400	3.8	0.44	0.89	1.3
^{133}I	20.8 h	0.0034	0.8	40	4.3	0.22	0.027	0.24
^{135}I	6.57 h	0.0011	0.8	40	0.93	0.015	0.002	0.017
^{134}Cs	2.06 a	1.49	0.2	2 000	19	100	2 620	2 720
^{136}Cs	13.16 d	0.051	0.2	2 000	3.0	0.57	14	15
^{137}Cs	30.07 a	2.81	0.2	2 000	13	130	3 370	3 500
^{140}Ba	12.75 d	0.050	0.5	4	2.6	1.2	0.02	1.2
^{141}Ce	32.5 d	0.12	0.1	30	0.71	0.16	0.12	0.28
^{143}Ce	1.38 d	0.0054	0.1	30	1.1	0.011	0.008	0.019
^{144}Ce	284.9 d	0.82	0.1	30	5.2	7.9	5.9	14
^{147}Pm	2.623 a	2.15	0.1	30	0.26	1.0	0.78	1.8
^{239}Pu	24 110 a	3.00	0.1	30	250	1 380	1 040	2 420

a Time integrals of unit activity in freshwater were calculated from the empirically derived values of the mean residence times in water of ^{90}Sr and ^{137}Cs (5 and 3 years, respectively), assuming that those radionuclides with high K_d, i.e. ^{144}Ce and ^{239}Pu, behave as ^{137}Cs and the other radionuclides behave as ^{90}Sr, in both cases correcting for physical delay. The formula is: $A_0 (\tau + \lambda)^{-1}$, where A_0 is unit activity (1 Bq), τ is the reciprocal of the mean residence time, and λ is ln 2 / half-life.

b Ref. [I9].

Table 28
Collective dose per unit release of radionuclides in liquid effluents to salt water

Radio-nuclide	*Half-life*	*Time integral of unit activity in water* [a] *(Bq a)*	*Concentration factor* [b] *(Bq kg^{-1} fish per Bq l^{-1} water)*		*Dose per unit activity ingested (nSv Bq^{-1})*	*Collective dose per unit activity released (man Sv PBq^{-1})*		
			Fish	*Shellfish (crustacea)*		*Fish*	*Shellfish (crustacea)*	*Total*
^{3}H	12.26 a	2.56	1	1	0.018	0.0012	0.0002	0.0014
^{14}C	5 730 a	3.00	20 000	20 000	0.58	890	150	1 040
^{24}Na	14.36 h	0.0024	0.1	0.1	0.43	0.000003	-	0.000003
^{35}S	87.5 d	0.31	2	1	0.13	0.0021	0.0002	0.0022
^{45}Ca	162.2 d	0.53	2	5	0.71	0.019	0.0080	0.027
^{51}Cr	27.7 d	0.11	200	500	0.038	0.021	0.0086	0.029
^{54}Mn	312.1 d	0.87	400	500	0.71	6.4	1.3	7.7
^{55}Fe	2.73 a	1.70	3 000	5 000	0.33	43	12	55
^{57}Co	271.8 d	0.79	1 000	5 000	0.21	4.3	3.6	7.8
^{58}Co	70.8 d	0.26	1 000	5 000	0.75	4.9	4.1	9.1
^{59}Fe	44.5 d	0.17	3 000	5 000	1.8	23	6.4	29
^{60}Co	5.271 a	2.15	1 000	5 000	3.4	190	160	350
^{65}Zn	244.3 d	0.73	1 000	50 000	3.9	73	610	680
^{89}Sr	50.5 d	0.19	2	2	2.6	0.025	0.0042	0.029
^{90}Sr	28.78 a	2.80	2	2	28	4.0	0.67	4.7
^{95}Zr	64.02 d	0.23	20	200	0.96	0.12	0.19	0.31
^{95}Nb	34.98 d	0.13	30	200	0.59	0.060	0.067	0.13
^{97}Zr	16.9 h	0.0028	20	200	2.1	0.0030	0.0050	0.0080
^{99}Mo	2.75 d	0.011	2	10	0.6	0.0003	0.0003	0.0006
^{103}Ru	39.26 d	0.15	2	100	0.73	0.0055	0.046	0.052
^{106}Ru	373.6 d	0.99	2	100	7.0	0.36	3.0	3.3
^{110m}Ag	249.8 d	0.74	500	5 000	2.8	27	45	71
^{113}Sn	115.1 d	0.40	50 000	50 000	0.73	370	62	430
^{122}Sb	2.73 d	0.011	400	400	1.7	0.19	0.031	0.22
^{124}Sb	60.2 d	0.22	400	400	2.5	5.7	0.94	6.6
^{125}Sb	2.76 a	1.71	400	400	1.1	19	3.2	23
^{129}I	1.6 10^{7} a	3	10	10	110	85	14	99
^{131}I	8.02 d	0.031	10	10	22	0.18	0.030	0.21
^{132}Te	3.2 d	0.013	1 000	1 000	3.8	1.2	0.21	1.4
^{133}I	20.8 h	0.0034	10	10	4.3	0.0038	0.0006	0.0044
^{135}I	6.57 h	0.0011	10	10	0.93	0.0003	0.00004	0.0003
^{134}Cs	2.06 a	1.49	100	30	19	73	3.6	77
^{136}Cs	13.16 d	0.051	100	30	3.0	0.39	0.020	0.41
^{137}Cs	30.07 a	2.81	100	30	13	94	4.7	98
^{140}Ba	12.75 d	0.050	10	1	2.6	0.033	0.0006	0.034
^{141}Ce	32.5 d	0.12	50	1 000	0.71	0.11	0.37	0.49
^{143}Ce	1.38 d	0.0054	50	1 000	1.1	0.0077	0.026	0.033
^{144}Ce	284.9 d	0.82	50	1 000	5.2	5.5	18	24
^{147}Pm	2.623 a	1.67	500	1 000	0.26	5.6	1.9	7.5
^{239}Pu	24 110 a	3.50	40	300	250	900	1120	2 020

a Obtained from estimated mean residence times in water of 3 years for ^{90}Sr, ^{137}Cs and other radionuclides and 3.5 years for ^{239}Pu. The time integral of unit activity is $1/(\tau+\lambda)$, where τ is the reciprocal of the mean residence time and λ is ln2 / half-life.

b Ref. [I10].

Table 29
Collective effective dose for representative composition of particulates released from reactors in liquid effluents

Radionuclide	*Fractional release*	*Collective dose per unit release (man Sv PBq^{-1})*		*Contribution to collective dose per unit total release (man Sv PBq^{-1})*	
		Freshwater	*Saltwater*	*Freshwater*	*Saltwater*
^{58}Co	0.20	4.6	9.1	0.92	1.8
^{60}Co	0.20	240	350	47	69
^{51}Cr	0.10	0.075	0.029	0.0075	0.0029
^{131}I	0.10	12	0.21	1.2	0.021
^{137}Cs	0.10	3 500	98	350	9.8
^{24}Na	0.05	0.010	0.000003	0.0005	-
^{54}Mn	0.05	19	7.7	1.0	0.39
^{65}Zn	0.05	170	680	8.7	34
^{134}Cs	0.05	2 720	77	140	3.8
^{133}I	0.02	0.24	0.0044	0.0049	0.00009
^{55}Fe	0.01	13	55	0.13	0.55
^{59}Fe	0.01	5.6	29	0.056	0.29
^{89}Sr	0.01	6.0	0.029	0.060	0.0003
^{95}Nb	0.01	1.8	0.13	0.018	0.0013
^{110m}Ag	0.01	22	71	0.22	0.71
^{125}Sb	0.01	34	23	0.34	0.23
^{135}I	0.01	0.017	0.0003	0.0002	-
^{140}Ba	0.01	1.2	0.034	0.012	0.0003
Total	1.0			550	120
Average				330	

Table 30
Parameters of the seven-compartment model of the world hydrological cycle [N3]

Compartment		*Northern hemisphere*	*Southern hemisphere*	*World*
Volume (10^{12} m^3)				
Atmospheric water		6.33	6.67	13
Soil water [a]		45.13	21.87	67
Freshwater		95	31	126
Saline water		100	4	104
Groundwater [a]		5 624	2 726	8 350
Ocean surface [b]		11 568	15 432	27 000
Deep ocean [b]		553 980	739 020	1 293 000
Transfer rate (10^{12} m^3 a^{-1})				
Atmosphere	– Soil	66.85	32.45	99.3
	– Freshwater	0.452	0.148	0.6
	– Saline water	0.096	0.004	0.1
	– Ocean	137.1	190.3 [c]	320
Soil	– Atmosphere	45.91	22.59	68.5
	– Freshwater	19.80	9.60	29.4
	– Groundwater	17.11	8.29	25.4
Freshwater	– Atmosphere	0.75	0.25	1.0
	– Ocean	20.25	9.75	30.0
Saline water	– Atmosphere	0.48	0.02	0.5
Groundwater	– Soil	15.97	8.03	24.0
	– Freshwater	0.754	0.246	1.0
	– Saline water	0.385	0.015	0.4
Ocean	– Atmosphere	157.4 [c]	200.0	350
	– Deep ocean	685.5	914.5	1 600
Deep ocean	– Ocean	685.5	914.5	1 600

a Land surface area: 67.35% in northern hemisphere, 32.65% in southern hemisphere.
b Ocean surface area: 42.84% in northern hemisphere, 57.16% in southern hemisphere.
c Transfer of 7.4 10^{12} m^3 a^{-1} from ocean surface to atmosphere (northern hemisphere) and atmosphere to ocean surface (southern hemisphere) added to achieve balance.

Table 31
Comparison of model estimates of global collective doses from tritium released to the atmosphere

Model	*Normalized effective dose ($nSv\ PBq^{-1}$)*	*Normalized collective dose [a] ($man\ Sv\ PBq^{-1}$)*
NCRP [N3] Seven-compartment model		
30°-50° northern hemisphere	0.38	0.7
Northern hemisphere	0.13	0.67
Southern hemisphere	0.11	0.07
World	0.06	0.35
Bergmann et al. [B4]		
0°-90° northern troposphere		0.95
0°-90° southern troposphere		0.65
Whole stratosphere		0.76
Killough and Kocher [K2]		
World troposphere		0.94
Northern troposphere		1.4
30°-50° northern troposphere		2.3
Natural tritium production		
Northern hemisphere	0.27	1.5
Southern hemisphere	0.27	0.2
World	0.14	0.8

a World population: $6\ 10^{9}$.

Table 32
Results of model calculation of release of 1 PBq of tritium to the atmosphere [a]

Region	*Integrated concentrations in 70-year period ($Bq\ a\ m^{-3}$)*			
	Release to 30°-50° N latitude	*Release to northern hemisphere*	*Release to southern hemisphere*	*Release to world*
Atmosphere	24.3	6.5	5.4	3.0
Surface soil water	17.0	5.3	4.4	2.4
Freshwater	6.9	4.1	3.6	1.9
Saline water	0.12	0.12	0.10	0.055
Groundwater	0.83	0.26	0.22	0.12
Ocean surface	2.9	0.66	0.54	0.30
Deep ocean	0.059	0.014	0.011	0.0061
Man	14.7	4.9	4.3	2.3
Effective dose commitment (nSv)	0.38	0.13	0.11	0.06

a Obtained with use of seven-compartment model [N3].

Table 33
Distribution of tritium and dose from release of 1 PBq to the atmosphere of the 30°-50° N band of the northern hemisphere; results of seven-compartment model

Time after release (years)	*Activity (TBq)*								*Integrated concentration in man (Bq a m^{-3})*	*Effective dose (nSv)*	
	Atmosphere	*Surface soil water*	*Freshwater*	*Saline water*	*Groundwater*	*Ocean surface*	*Deep ocean*	*Total*		*Annual*	*Cumulative*
0	1 000	0	0	0	0	0	0	1 000			
1	2.9	75.0	63.6	0.54	60	704	39	945	10.23	0.26	0.26
2	0.90	17.8	62.9	0.53	71	664	76	893	1.43	0.037	0.30
3	0.46	5.4	51.5	0.52	71	607	108	844	0.65	0.017	0.32
4	0.33	2.6	40.4	0.50	68	550	136	798	0.41	0.011	0.33
5	0.28	1.9	31.5	0.48	65	497	158	754	0.31	0.0080	0.33
6	0.25	1.6	24.5	0.46	62	448	177	713	0.25	0.0064	0.34
7	0.22	1.4	19.2	0.44	59	403	191	674	0.20	0.0052	0.35
8	0.20	1.3	15.1	0.42	56	362	202	637	0.17	0.0043	0.35
9	0.18	1.1	11.9	0.40	53	325	211	602	0.14	0.0036	0.35
10	0.16	1.0	9.4	0.38	51	291	217	569	0.12	0.0030	0.36
15	0.09	0.60	3.28	0.30	39	167	219	430	0.054	0.0014	0.37
20	0.053	0.36	1.37	0.24	30	95	197	324	0.028	0.00072	0.37
25	0.031	0.21	0.67	0.18	22	55	166	245	0.016	0.00041	0.37
30	0.018	0.13	0.37	0.14	17	32	135	184	0.0093	0.00024	0.38
35	0.011	0.082	0.22	0.11	13	18	108	139	0.0057	0.00015	0.38
40	0.0064	0.052	0.13	0.09	10	11	84	105	0.0036	0.000092	0.38
45	0.0039	0.034	0.08	0.07	7.2	6.5	65	79	0.0023	0.000059	0.38
50	0.0024	0.023	0.06	0.05	5.4	3.9	50	60	0.0015	0.000039	0.38
55	0.0016	0.015	0.04	0.04	4.0	2.4	39	45	0.0010	0.000026	0.38
60	0.0010	0.011	0.03	0.03	3.0	1.6	29	34	0.00070	0.000018	0.38
65	0.00068	0.0074	0.02	0.02	2.3	1.0	22	26	0.00049	0.000013	0.38
70	0.00047	0.0053	0.01	0.02	1.7	0.7	17	19	0.00035	0.000009	0.38
Total									14.7	0.38	

Table 34
Results of model calculation of the release of 1 PBq of ^{14}C to the environment
[T1]

Year	*Inventory of ^{14}C (TBq)*		*Inventory of stable carbon (10^{12} g) ground vegetation* [a]	*Integrated specific activity (Bq g^{-1})*			*Effective dose (μSv)*	
	Atmosphere	*Ground vegetation*		*Atmosphere* [b]	*Ground vegetation* [b]	*Ground vegetation* [a]	*Annual dose* [a]	*Cumulative dose* [a]
Release to atmosphere								
1	885	18.6	69 100	0.00059	0.00013	0.00013	0.0076	0.0076
2	712	40.4	69 300	0.0017	0.00056	0.00056	0.024	0.032
5	437	46.4	70 000	0.0039	0.0026	0.0026	0.038	0.13
10	253	28.7	70 900	0.0061	0.0053	0.0052	0.024	0.27
20	137	13.9	71 800	0.0085	0.0081	0.0080	0.011	0.41
50	76.3	7.18	73 200	0.012	0.012	0.012	0.0056	0.61
100	50.0	4.65	74 500	0.017	0.016	0.016	0.0035	0.85
200	33.0	3.05	76 000	0.022	0.022	0.021	0.0023	1.1
500	21.1	1.94	78 500	0.032	0.032	0.030	0.0014	1.6
1 000	16.3	1.50	80 400	0.045	0.045	0.040	0.0011	2.1
2 000	13.7	1.26	82 300	0.064	0.064	0.057	0.00086	3.0
5 000	9.41	0.866	84 400	0.11	0.11	0.095	0.00058	4.9
10 000	5.02	0.462	84 600	0.16	0.16	0.13	0.00031	6.9
20 000	1.43	0.131	83 900	0.20	0.20	0.17	0.000088	8.5
50 000	0.033	0.0030	82 000	0.22	0.22	0.18	0.000002	9.2

Year	*Inventory of ^{14}C (TBq)*		*Inventory of stable carbon (10^{12} g) ground vegetation* [a]	*Integrated specific activity (Bq g^{-1})*			*Effective dose (μSv)*	
	Ocean surface	*Ground vegetation*		*Ocean surface* [b]	*Ground vegetation* [b]	*Ground vegetation* [a]	*Annual dose* [a]	*Cumulative dose* [a]
Release to ocean surface								
1	977	0.124	69 100	0.00047	0.00000	0.000001	0.00005	0.00005
2	914	0.69	69 300	0.0014	0.00001	0.00001	0.00033	0.00050
5	777	3.2	70 000	0.0038	0.00009	0.00009	0.0016	0.0050
10	599	5.6	70 900	0.0071	0.00041	0.00040	0.0035	0.022
20	368	6.3	71 800	0.012	0.0013	0.0012	0.0047	0.067
50	137	4.6	73 200	0.019	0.0036	0.0035	0.0042	0.18
100	81	3.5	74 500	0.024	0.0066	0.0062	0.0031	0.32
200	53	2.6	76 000	0.030	0.011	0.010	0.0023	0.53
500	30	1.8	78 500	0.042	0.021	0.019	0.0016	1.0
1 000	23	1.4	80 400	0.055	0.032	0.029	0.0012	1.5
2 000	19	1.2	82 300	0.075	0.052	0.045	0.0009	2.4
5 000	14	0.90	84 400	0.12	0.098	0.083	0.0007	4.3
10 000	7.0	0.46	84 600	0.17	0.15	0.12	0.0005	6.3
20 000	2.0	0.13	83 900	0.21	0.19	0.16	0.0002	7.9
50 000	0.046	0.0030	82 000	0.24	0.22	0.18	0.00005	8.6

a Assuming variable inventory of carbon in environment as a result of input from burning of fossil fuels.
b Assuming fixed inventories of carbon in the environment: 750 10^{15} g (atmosphere); 1,050 10^{15} g (Atlantic Ocean surface) and 69 10^{15} g (ground vegetation).

Table 35
Estimates of collective dose to the world population per unit release of ^{129}I to different environmental compartments calculated using a global circulation model
[T1]

Time (years)	*Collective effective dose per unit release (man Sv TBq^{-1})*				
	Release to land atmosphere	*Release to ocean atmosphere*	*Release to ocean mixed layer*	*Release to deep ocean*	*Release to solid soil*
1	67.1	13.9	0.00432	0.0000036	0.0292
2	81.7	19.3	0.0158	0.000032	0.0977
5	81.8	19.4	0.0420	0.000288	0.305
10	82.0	19.4	0.0649	0.00106	0.649
20	82.4	19.5	0.0804	0.00313	1.34
50	83.4	19.8	0.0908	0.00996	3.40
100	85.1	20.2	0.104	0.0216	6.80
200	88.4	21.0	0.130	0.0453	13.5
500	98.0	23.3	0.212	0.121	32.9
1 000	113	26.9	0.361	0.259	62.9
2 000	138	33.2	0.698	0.577	115
5 000	192	46.8	1.94	1.78	223
10 000	236	59.0	4.43	4.24	309
20 000	263	69.5	9.82	9.62	359
50 000	282	85.6	25.0	24.8	382
100 000	303	106	45.8	45.6	403
1 000 000	450	254	193	193	551
10 000 000	643	446	385	385	744
100 000 000	727	530	469	469	828

References

A1 Aarkrog, A. Radioecological lessons learned from Chernobyl. p. 129-134 in: Proceedings of the XVth Regional Congress of IRPA, September 1989, Visby, Gotland, Sweden (1989).

A2 Aarkrog, A. Chernobyl-related monitoring and comparison with fallout data. p. 229-249 in: Proceedings of Seminar on Project "MARINA". CEC, Luxembourg, 1989.

A3 Aarkrog, A., L. Bøtter-Jensen, Q.J. Chen et al. Environmental radioactivity in Denmark in 1990 and 1991. Risø-R-621 (1992).

A4 Aarkrog, A. Variation of direct plutonium contamination in Danish cereal grains. Health Phys. 35: 489-491 (1978).

A5 Aarkrog, A. Environmental studies on radioecological sensitivity and variability with special emphasis on the fallout radionuclides ^{90}Sr and ^{137}Cs. Risø-R-437 (1979).

A6 Alevra, A.V. Neutron spectrometry with Bonner spheres: Applications in physics and dosimetry. Physikalisch Technische Bundesanstalt, Braunschweig (1996).

B1 Barry, P.J. Energy for tomorrow - nuclear power and the environment. AECL-5011 (1974).

B2 Bennett, B.G. Environmental aspects of americium. EML-348 (1978).

B3 Bennett, B.G. Environmental tritium and the dose to man. in: Proceedings of the Third International Congress of the International Radiation Protection Association (IRPA), Washington D.C. (1973).

B4 Bergman, R., U. Bergström and S. Evans. Environmental transport and long-term exposure for tritium released in the biosphere. p. 535-554 in: Behaviour of Tritium in the Environment. STI/PUB/498. IAEA, Vienna, 1979.

B5 Brauer, F.P. and R.S. Strebin Jr. Environmental concentration and migration of ^{129}I. p. 465-480 in: Environmental Migration of Long-lived Radionuclides. STI/PUB/597. IAEA, Vienna, 1982.

B6 Bainbridge, A. Geosecs Atlantic Expedition. Volume 2. Sections and Profiles. National Science Foundation, United States Government Printing Office, Washington, 1981.

B7 Belot, Y., K. Ganthier, H. Camus et al. Prediction of the flux of tritiated water from the air to plant leaves. Health Phys. 37: 575 (1979).

B8 Beck, H.L. The physics of environmental gamma radiation fields. p. 101-133 in: The Natural Radiation Environment II (J.A.S. Adams, W.M. Lowder and T.F. Gesell, eds.). CONF-720805 (1972).

B9 Beck, H.L. Exposure rate conversion factors for radionuclides deposited on the ground. EML-378 (1980).

B10 Bernström, B. Radioactivity from nuclear explosions in ground-level air and precipitation in Sweden. NaI (Tl) measurements from 1972 to the end of 1975. FOA/C40080-T2(A1) (1978) *and* EML-349 (1979).

B11 Bennett, B.G. Fallout ^{239}Pu dose to man. p. I-42-I-63 in: Health and Safety Laboratory Fallout Program Quarterly Summary Report. HASL-278 (1974).

B12 Bouville, A. and W.M. Lowder. Human population exposure to cosmic radiation. Radiat. Prot. Dosim. 24: 293-299 (1988).

B13 Briggs, G.A. Diffusion estimation for small emissions. US Department of Commerce, NOAA-ERL-ARATDL Contribution no. 79 (1973).

B14 Birattari, C., B. Moy, T. Rancati et al. Neutron measurements at some environmental monitoring stations. Internal Report, CERN, TIS-RP/IR/96-13 (1996).

C1 Chamberlain, A.C. Aspects of the deposition of radio-active and other gases and particles. Int. J. Air Pollut. 3: 63-88 (1960).

C2 Chamberlain, A.C. Interception and retention of radio-active aerosols by vegetation. Atmos. Environ. 4: 57-78 (1970).

C3 Chamberlain, A.C. and J.A. Garland. Interception and retention of radioactive fallout by vegetation. AERE-R-13826 (1991).

C4 Cannell, M.G.R. and M.D. Hooper. The greenhouse effect and terrestrial ecosystems of the UK. Institute of Terrestrial Ecology, Grange-over-Sands. Research Publication No. 4. HMSO, London (1990).

C5 Commission of the European Communities. Underlying data for derived emergency reference levels. Post-Chernobyl action (J. Sinnaeve and G. Gerber, eds.). EUR 12553 (1991).

C6 Commission of the European Communities. The radiological exposure on the population of the European community from radioactivity in North European marine waters. Project MARINA. EUR 12483 (1990).

C7 Cambray, R.S., E.M.R. Fisher, W.L. Brooks et al. Radioactive fallout in air and rain: Results to the middle of 1970. AERE-R6556 (1970).

C8 Civil Defence, Sweden. Protection factors for radioactive deposition. Communication to the UNSCEAR Secretariat by K. Edvarson (1988).

C9 Christensen, G.C. and R. Mustonen. The filtering effects of buildings on airborne particles. Radiat. Prot. Dosim. 21: 125-128 (1987).

C10 Canadian Standards Association. Guidelines for calculating radiation doses to the public from a release of airborne radioactive material under hypothetical accident conditions in nuclear reactors. CAN/CSA-N288.2-M91, Canada (1991).

D1 Davis, P.A. Communication to the UNSCEAR Secretariat (1998).

D2 Dunstall, T.G., G.L. Ogram and F.S. Spencer. Elemental tritium deposition and conversion in the terrestrial environment. Fusion Technol. 8: 2551-2556 (1985).

D3 Diabate, S. and S. Strack. Organically bound tritium. Health Phys. 65: 698-712 (1993).

D4 Dahlgaard, H. (ed.). Nordic Radioecology. The Transfer of Radionuclides through Nordic Ecosystems to Man. Elsevier, Amsterdam, 1994.

D5 DeGeer, L.E., R. Arntsing, I. Vintersved et al. Particulate radioactivity, mainly from nuclear explosions, in air and precipitation in Sweden mid-year 1975 to mid-year 1977. FOA/C40089-T2(A1) (1978) *and* p.I-49–I-124 in EML-349 (1979).

D6 Diem, K. and C. Lentner (eds.), Documents Geigy: Scientific Tables, 8th edition. Ciba-Geigy Ltd., Basel, 1981.

E1 Emanuel, W.R., G.G. Killough and J.S. Olson. Modelling the circulation of carbon in the world's terrestrial ecosystems. in: Carbon Cycle Modelling (B. Bolin, ed.). John Wiley & Sons, New York, 1981.

E2 Emanuel, W.R., G.G. Killough, W.M. Post et al. Computer implementation of a globally averaged model of the world carbon cycle. DOE/NBB-0062 (1984).

E3 Engelmann, R.J. Scavenging prediction using ratios of concentrations in air and precipitation. J. Appl. Meteorol. 10: 493-497 (1971).

E4 Eriksson, Å. Fissions Product in the Swedish Environment. Institute for Radiobiology in the Agricultural University, Uppsala, 1977. (In Swedish).

E5 Edvarson, K. Fallout over Sweden from the Chernobyl accident. p. 47-65 in: The Chernobyl Fallout in Sweden (L. Moberg, ed.). Swedish Radiation Protection Institute, Stockholm, 1991.

E6 Ehalt, D.H. Methane in the atmosphere. in: Carbon and the Biosphere. Proceedings of the 24th Brookhaven Symposium in Biology (G.M. Woodwell and E.V. Pecan, eds). AEC Symposium Series, CONF-720510 (1973).

E7 Eckerman, K.F. and J.C. Ryman. External exposure to radionuclides in air, water, and soil. Federal Guidance Report No. 12. EPA 402-R-93-081 (1993).

F1 Food and Agriculture Organization. FAO Food Balance Sheets, 1979-1981 (1984).

F2 Food and Agriculture Organization. Fishery Statistics, Volumes 70 and 71 (1992).

F3 Florek, M., J. Masarik, I. Szarka et al. Natural neutron fluence rate and the equivalent dose in localities with different elevation and latitude. Radiat. Prot. Dosim. 67: 187-192 (1996).

G1 Garland, J.A. Transfer of tritiated water vapour to and from land surfaces. p. 349-359 in: Behaviour of Tritium in the Environment. STI/PUB/498. IAEA, Vienna, 1979.

G2 Gesellschaft für Strahlen- und Umweltforschung. Umweltradioaktivität und Strahlenexposition in Südbayern durch den Tschernobyl Unfall. GSF-Bericht 16/86 (1986).

G3 Goldhagen, P. Overview of aircraft radiation exposure and recent ER-2 measurements. in: Cosmic Radiation Exposure of Airline Crews, Passengers, and Astronauts. NCRP, Bethesda, 1999.

H1 Harrison, R.M. Atmospheric pathways. p. 56-100 in: Radioecology after Chernobyl (F. Warner and R.M. Harrison, eds.). John Wiley & Son, Chichester, 1993.

H2 Holm, E., P. Roos, R.B.R. Persson et al. Radiocaesium and plutonium in atlantic surface waters from 73°N to 72°S. p. 3-11 in: Radionuclides in the Study of Marine Processes (P.J. Kershaw and D.S. Woodhead, eds.). Elsevier Applied Science, London, 1991.

H3 Hamby, D.M. and L.R. Bauer. The vegetation-to-air concentration ratio in a specific activity atmospheric tritium model. Health Phys. 66: 339-342 (1994).

H4 Hamby, D.M. A probabilistic estimation of atmospheric tritium dose. Health Phys. 65: 33-40 (1993).

H5 Horton, J.H., J.C. Corey and R.M. Wallace. Tritium loss from water exposed to the atmosphere. Environ. Sci. Technol. 5: 338-343 (1971).

H6 Hoffman, F.O., D.L. Shaeffer, C.W. Miller et al. Proceedings of a Workshop on the Evaluation of Models Used for the Environmental Assessment of Radionuclide Releases. CONF-770901 (1977).

H7 Horst, T.W. A surface depletion model for deposition from a Gaussian plume. Atmos. Environ. 11: 41-46 (1977).

H8 Hosker, R.P. Estimates of dry deposition and plume depletion over forests and grassland. p. 291-309 in: Physical Behaviour of Radioactive Contaminants in the Atmosphere. STI/PUB/354. IAEA, Vienna, 1974.

I1 International Commission on Radiological Protection. 1990 Recommendations of the International Commission on Radiological Protection. ICRP Publication 60. Annals of the ICRP 21(1-3). Pergamon Press, Oxford, 1991.

I2 International Commission on Radiological Protection. Age-dependent doses to members of the public from intake of radionuclides: Part 2. Ingestion dose coefficients. ICRP Publication 67. Annals of the ICRP 23(3/4). Pergamon Press, Oxford, 1993.

I3 International Commission on Radiological Protection. Age-dependent doses to members of the public from intake of radionuclides: Part 3. Ingestion dose coefficients. ICRP Publication 69. Annals of the ICRP 25(1). Pergamon Press, Oxford, 1995.

I4 International Commission on Radiological Protection. Age-dependent doses to members of the public from intake of radionuclides: Part 4. Inhalation dose coefficients. ICRP Publication 71. Annals of the ICRP 25(3-4). Pergamon Press, Oxford, 1995.

I5 International Commission on Radiological Protection. Age-dependent doses to members of the public from intake of radionuclides: Part 5. Compilation of ingestion and inhalation dose coefficients. ICRP Publication 72. Annals of the ICRP 26(1). Pergamon Press, Oxford, 1996.

I6 International Commission on Radiological Protection. Human respiratory tract model for radiological protection. ICRP Publication 66. Annals of the ICRP 24(1-3). Pergamon Press, Oxford, 1994.

I7 International Commission on Radiological Protection. Report of the Task Group on Reference Man. ICRP Publication 23. Pergamon Press, Oxford, 1975.

I8 International Commission on Radiation Units and Measurements. Fundamental quantities and units for ionizing radiation. ICRU Report 60 (1998).

I9 International Atomic Energy Agency. Handbook of parameter values for the prediction of radionuclide transfer in temperate environments. IAEA Technical Report Series No. 364 (1994).

I10 International Atomic Energy Agency. Sediment K_ds and concentration factors for radionuclides in the marine environment. IAEA Technical Report Series No. 247 (1985).

I11 International Commission on Radiological Protection. Recommendations of the ICRP. ICRP Publication 26. Annals of the ICRP 1(3). Pergamon Press, Oxford, 1977.

I12 International Commission on Radiation Units and Measurements. Quantities and units in radiation protection dosimetry. ICRU Report 51 (1993).

I13 International Commission on Radiological Protection. Protection against radon-222 at home and at work. ICRP Publication 65. Annals of the ICRP 23(2). Pergamon Press, Oxford, 1993.

I14 International Commission on Radiological Protection. Statement from the 1978 Stockholm meeting of the ICRP. The principles and general procedures for handling emergency and accidental exposures of workers. ICRP Publication 28. Annals of the ICRP 2(1). Pergamon Press, Oxford, 1978.

I15 International Atomic Energy Agency. Atmospheric dispersion in nuclear power plant siting: A safety guide. Safety Series No. 50-SG-S3 (1980).

I16 International Atomic Energy Agency. Validation of models using Chernobyl fallout data from Southern Finland. Scenario S. IAEA-TECDOC-904 (1996).

I17 International Atomic Energy Agency. Modelling of radionuclide interception and loss processes in vegetation and of transfer in semi-natural ecosystems. IAEA-TECDOC-857 (1996).

I18 International Commission on Radiological Protection. Lung cancer risk from indoor exposures to radon daughters. ICRP Publication 50. Annals of the ICRP 17(1). Pergamon Press, Oxford, 1987.

J1 Jones, J.A. ESCLOUD: A computer programme to calculate the air concentration, deposition rate and external dose rate from a continuous discharge of radioactive material to atmosphere. NRPB-R101 (1980).

J2 Jacob, P., R. Meckbach and H.M. Müller. Reduction of external exposures from deposited Chernobyl activity by run-off, weathering, street-cleaning and migration in the soil. Radiat. Prot. Dosim. 21: 51-57 (1987).

J3 Johnson, W.B. Interregional exchanges of air pollution: Model types and applications. J. Air Pollut. Control Assoc. 3: 563-574 (1983).

K1 Kossina, E. Die Tiefen des Weltmeers. Veröffentlichungen des Institut für Meereskunde, Vol. 9. E.S. Mittler & Sohn, Berlin, 1921.

K2 Killough, G.G. and D.C. Kocher. Global environmental transport models for tritium. Fusion Technol. 14: 1115-1120 (1988).

K3 Kelly, G.N., J.A. Jones, P.M. Bryant et al. The predicted radiation exposure of the population of the European Community resulting from discharges of ^{85}Kr, ^{3}H, ^{14}C and ^{129}I from the nuclear power industry to the year 2000. CEC Doc. No. V/2676/75 (1975).

K4 Killough, G.G. and D.C. Kocher. Global environmental transport models for tritium. Fusion Technol. 8: 2569-2574 (1985).

K5 Killough, G.G. and P.S. Rohwer. A new look at the dosimetry of ^{14}C released to the atmosphere as carbon dioxide. Health Phys. 34: 141-159 (1978).

K6 Karlberg, O. Weathering and migration of Chernobyl fallout in Sweden. Radiat. Prot. Dosim. 21: 75-78 (1987).

K7 Kocher, D.C. Dose-rate conversion factors for external exposure to photons and electrons. Health Phys. 45: 665-686 (1983).

K8 Kocher, D.C. A dynamic model of the global iodine cycle for the estimation of doses to the world population from releases of iodine-129 to the environment. ORNL/NUREG-59 (1979).

K9 Kocher, D.C. A dynamic model of the global iodine cycle and estimatioin of dose to the world population from releases of iodine-129 to the environment. Environ. Int. 5:15-31 (1981).

L1 Lovelock, J.E., R.J. Maggs and R.J. Wade. Halogenated hydrocarbons in and over the Atlantic. Nature 241: 194-196 (1973).

L2 LaMarre, J.R. Annual summary and assessment of environmental radiobiological data for 1998. Ontario Power Generation Report N-REP-03419-0597 R00 (1999).

M1 Mobbs, S.F., D. Charles, C.E. Delow et al. PAGIS Performance Assessment of Geological Isolation Systems for radioactive waste. Disposal into the sub-seabed. EUR 11779 (1988).

M2 Middleton, L.J. Radioactive strontium and caesium in the edible parts of crop plants after foliar contamination. Int. J. Radiat. Biol. 4: 387-402 (1959).

M3 Marine Environment Laboratory. Sources of radioactivity in the marine environment and their relative contributions to overall dose assessment from marine radioactivity (MARDOS). Final Report of the CRP. IAEA-MEL-R2/94 (1994).

M4 Murphy, C.E. Jr. The relationship between tritiated water activities in air, vegetation and soil under steady-state conditions. Health Phys. 47: 635-639 (1984).

M5 McCartney, M., M.S. Baxter and E.M. Scott. Carbon-14 discharges from the nuclear fuel cycle: 1. Global effects. J. Environ. Radioact. 8: 143-188 (1988).

M6 Meckbach, R., P. Jacob and H.G. Paretzke. Shielding of gamma radiation by typical European houses. Nucl. Instrum. Methods Phys. Res. Sect. A 255: 160-164 (1987).

M7 Muramatsu, Y. and Y. Ohmomo. Iodine-129 and iodine-127 in environmental samples collected from Tokaimura/ Ibaraki, Japan. Sci. Total Environ. 48: 33-43 (1986).

M8 Muramatsu, Y. and K.H. Wedepohl. The distribution of iodine in the Earth's crust. Chem. Geol. 147: 201-216 (1998).

M9 Miskel, J.A. Production of tritium by nuclear weapons. p.79-85 in: Tritium (A.A. Moghissi and M.W. Carter, eds.) Messesger Graphics, Las Vegas, 1973.

N1 Noordijk, H. and J.M. Quinault. The influence of food processing and culinary preparation on the radionuclide content of foodstuffs: A review of available data. p.35-59 in: Modelling of Resuspension, Seasonality and Losses During Food Processing. IAEA-TECDOC-647 (1992).

N2 National Radiological Protection Board and Commissariat à l'Énergie Atomique. Methodology for evaluating the radiological consequences of radioactive effluents released in normal operations. CEC Doc. No. V/3865/79 (1979).

N3 National Council on Radiation Protection and Measurements. Tritium in the environment. NCRP Report No. 62 (1979).

N4 Nakamura, Y. and Y. Ohmomo. Factors used for the estimation of gaseous radioactive iodine intake through vegetation - I. Uptake of methyliodide by spinach leaves and II. Uptake of elemental iodine by spinach leaves. Health Phys. 38: 307-314 and 315-320 (1980).

N5 National Research Council. Radon in Drinking Water. National Academy Press, Washington, 1998.

N6 Niemi, T. and N. Soonawala. 1998 annual report of radiological monitoring results for the Chalk River and Whiteshell Laboratories sites. Vol. 3, Environmental monitoring. AECL-MISC-362-98 (1999).

N7 Nisbet, A. and R. Woodman. Soil-to-plant transfer factors for radiocaesium and radiostrontium in agricultural systems. Health Phys. 78: 279-288 (2000).

P1 Pan, Z., Z. Wang, Z. Chen et al. Radiological environmental impact of the nuclear industry in China. Health Phys. 71(6): 847-862 (1996).

P2 Pröhl, G. Modellierung der Radionuclidausbreitung in Nahrungsketten nach Deposition von Sr-90, Cs-137 und J-131 aus landwirtschaftlich genützte Flächen. GSF-Bericht 29/90 (1990).

P3 Poston, J.W. and W.S. Snyder. A model for exposure to a semi-infinite cloud of a photon emitter. Health Phys. 26: 287-293 (1974).

P4 Petoussi, N., M. Zankl, K. Saito et al. Organ doses to adults and children from environmental gamma rays. p. 372-377 in: The Radioecology of Natural and Artificial Radionuclides (W. Feldt, ed.). Verlag TüV Rheinland GmbH, Köln, 1989.

P5 Petoussi, N., P. Jacob, M. Zankl et al. Organ doses for foetuses, babies, children and adults from environmental gamma rays. Radiat. Prot. Dosim. 37: 31-34 (1991).

P6 Pasquill, F. The estimation of the dispersion of windborne material. Meteorol. Mag. 90: 33-49 (1961).

R1 Raskob, W. Description of NORMTRI: A computer program for assessing the off-site consequences from airborne releases of tritium during normal operation of nuclear facilities. Kernforschungszentrum Karlsruhe, Report KfK-5364 (1994).

R2 Roed, J. and R. Cannell. Relationship between indoor and outdoor aerosol concentration following the Chernobyl accident. Radiat. Prot. Dosim. 21: 107-110 (1987).

R3 Roesler, S., W. Heinrich and H. Schraube. Calculation of radiation fields in the atmosphere and comparison to experimental data. Radiat. Res. 149: 87-97 (1998).

R4 Robertson, E. and P.J. Barry. The validity of a Gaussian plume model when applied to elevated releases at a site on the Canadian shield. Atmos. Environ. 23: 351-362 (1989).

R5 Revzan, K.L. and W.J. Fisk. Modelling radon entry into houses with basements: the influence of structural factors. Indoor Air 2: 40-48 (1992).

R6 Rao, U. and U. Fehn. Source and reservoirs of anthropogenic iodine-129 in Western New York. Geochim. Cosmochim. Acta 63: 1927-1938 (1999).

S1 Smith, F.B. A scheme for estimating the vertical dispersion of a plume from a source near ground level. Proceedings of the Third Meeting of the Expert Panel on Air Pollution Modelling. NATO Committee on the Challenges of Modern Society, Report No. 14, Brussels (1972).

S2 Simmonds, J.R., G. Lawson and A. Mayall. Methodology for assessing the radiological consequences of routine releases of radionuclides to the environment. EUR 15760 EN (1995).

S3 Siegenthaler, U. Carbon-14 in the oceans. in: Handbook of Environmental Isotope Geochemistry 3, The Marine Environment (P. Fritz and J.Ch. Fontes, eds.). Elsevier, Amsterdam, 1989.

S4 Smith, G.M. and I.F. White. A revised global-circulation model for iodine-129. NRPB-M81 (1983).

S5 Saito, K., N. Petoussi, M. Zankl et al. Calculation of organ doses from environmental gamma rays using human phantoms and Monte Carlo methods. Part I. Monoenergetic sources and natural radionuclides in the ground. GSF-Bericht 2/90 (1990).

S6 Strand, T., E. Strandon and A.L. Rudjord. External radiation doses to the Norwegian population from the Chernobyl fallout. Radiat. Prot. Dosim. 20: 231-236 (1987).

S7 Steinhäusler, F., W. Hoffmann, F. Daschil et al. Chernobyl and Its Radiological and Socio-economic Consequences for the Province of Salzburg, Austria. Division of Biophysics, University of Salzburg, Austria, 1987.

S8 Schraube, H., J. Jakes, A. Sannikov et al. The cosmic ray induced neutron spectrum at the summit of the Zugspitze (2963 m). Radiat. Prot. Dosim. 70: 405-408 (1997).

S9 Schraube, H., G. Leuthold, S. Roesler et al. Neutron spectra at flight altitudes and their radiological estimation. Adv. Space Res. 21: 1727-1738 (1998).

S10 Saito, K. and P. Jacob. Gamma ray fields in the air due to sources in the ground. Radiat. Prot. Dosim. 58: 29-45 (1995).

S11 Saito, K., N. Petoussi-Henss and M. Zankl. Calculation of the effective dose and its variation from environmental gamma ray sources. Health Phys. 74: 698-706 (1998).

T1 Titley, J.G., T. Cabianca, G. Lawson et al. Improved global dispersion models for iodine-129 and carbon-14. EUR 15880 EN (1995).

T2 Taeschner, M., B. Wiener and C. Bunnenberg. HT dispersion and deposition in soil after experimental releases of tritiated hydrogen. Fusion Technol. 14: 1264-1273 (1988).

T3 Takeda, H. and Y. Kasida. Biological behavior of tritium after administration of tritiated water in the rat. J. Radiat. Res. 20: 174-185 (1979).

U3 United Nations. Sources and Effects of Ionizing Radiation. United Nations Scientific Committee on the Effects of Atomic Radiation, 1993 Report to the General Assembly, with scientific annexes. United Nations sales publication E.94.IX.2. United Nations, New York, 1993.

U4 United Nations. Sources, Effects and Risks of Ionizing Radiation. United Nations Scientific Committee on the Effects of Atomic Radiation, 1988 Report to the General Assembly, with annexes. United Nations sales publication E.88.IX.7. United Nations, New York, 1988.

U6 United Nations. Ionizing Radiation: Sources and Biological Effects. United Nations Scientific Committee on the Effects of Atomic Radiation, 1982 Report to the General Assembly, with annexes. United Nations sales publication E.82.IX.8. United Nations, New York, 1982.

U7 United Nations. Sources and Effects of Ionizing Radiation. United Nations Scientific Committee on the Effects of Atomic Radiation, 1977 Report to the General Assembly, with annexes. United Nations sales publication E.77.IX.1. United Nations, New York, 1977.

U8 United Nations. Ionizing Radiation: Levels and Effects. Volume I: Levels, Volume II: Effects. United Nations Scientific Committee on the Effects of Atomic Radiation, 1972 Report to the General Assembly, with annexes. United Nations sales publication E.72.IX.17 and 18. United Nations, New York, 1972.

U13 United Nations. Report of the United Nations Scientific Committee on the Effects of Atomic Radiation. Official Records of the General Assembly, Thirteenth Session, Supplement No. 17 (A/3838). New York, 1958.

U14 United States Geological Survey. Estimated world water supply and budget. Tables 2-3 in: The Water Encyclopedia (D.K. Todd, ed.). Water Information Center, Inc. Port Washington, New York, 1970.

U15 United States Bureau of the Census. International data base. Internet site: www.census.gov/cgi-bin/ipc/idbagg (1999).

V1 Vogt, K.J. Empirical investigations of the diffusion of waste air plumes in the atmosphere. Nucl. Technol. 34: 43-57 (1977).

W1 Whitehead, D.C. The distribution and transportations of iodine in the environment. Environ. Int. 10: 321-339 (1984).

Y1 Yamartino, R.J. A new method for computing pollutant concentrations in the presence of limited vertical mixing. J. Air Poll. Control Ass. 27: 467-468 (1977).