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MASTER

ORNL-5529

**Recommendations Concerning
Models and Parameters Best Suited
to Breeder Reactor Environmental
Radiological Assessments**

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Health and Safety Research Division

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HIGHLIGHTS

A project is underway at Oak Ridge National Laboratory which has as its objective the identification of models available for environmental radiological assessments; evaluation of model structure, simplifying assumptions and data bases; estimation of their uncertainties; and, if possible, the recommendation of the models and parameters which are best suited to particular assessment situations. When needs are identified, recommendations are also made for further environmental and biomedical research.

The purpose of this report is to present recommendations concerning the models and parameters best suited for assessing the impact of radionuclide releases to the environment by breeder reactor facilities. These recommendations are based on the model and parameter evaluations performed during this project to date. Seven different areas are covered in separate sections of the report. These sections may be summarized as follows:

The Gaussian plume model continues to be appropriate for most situations of interest. Dispersion parameters which are used to account for the release height and terrain conditions under consideration. Ground deposition and plume depletion processes can be parameterized through the use of the deposition velocity.

The compartmental flow model assumes equilibrium conditions and requires the input of empirically derived transfer coefficients. Wherever possible, this report describes the statistical distribution of such transfer coefficients for a given radionuclide.

A two dimensional, R-theory model assuming no sorption of the radionuclide^{1,2} seems appropriate for many hydrologic transport calculations.

If sorption and sediment transport are important considerations, however, more complicated models^{3,4} are needed to estimate the transport in surface water systems.

For assessment purposes, the bioaccumulation factor may be used to relate the concentration of a radionuclide in water to its concentration in aquatic foodstuffs. Bioaccumulation factors associated with 32 elements are tabulated for marine and freshwater fish and invertebrates.

The internal radiation dose to man depends in part on the quantity of radionuclides incorporated into the body from food, water, and/or air, and hence intake rates of these materials are important.

Adult intake factors are presented for fruits, vegetables, grains, meats, fish, poultry, milk, other liquids, and air.

Fifty-year dose commitment factors from inhalation and ingestion of potentially significant radionuclides in nuclear reactor fuel cycles are presented for lungs, total body, ovaries, testis, and skeletal cells, and testes. These calculations are based on the latest intake factors provided by the International Commission on Radiological Protection and other non-splenic intake factors. The intake factors are based upon external dose conversion factors for lungs, ovaries, lungs, ovaries, skeleton, testes, and total body from exposure in contaminated air and exposure to contaminated foodstuffs.

The methodologies currently used to estimate the dose to man from airborne releases of ¹³¹I and ¹³⁷Cs are outlined. However, these methods also are presented using the simplified model as part of another project at ORNL. The methodology outlined in this report may be modified by a comparison of the results.

Work on the model evaluation project is continuing at this time. As this work proceeds, modification of the recommendations presented in this report may be made to reflect the latest findings of the project staff.

REFERENCES

1. Yotsukura, N. and W. W. Sayre. 1976. Transverse mixing in natural channels. *Water Resour. Res.* 12(4):695-704.
2. Yotsukura, N. and E. D. Cobb. 1972. Transverse Diffusion of Solutes in Natural Streams. *U.S. Geol. Survey Prof. Paper* 582-C, U.S. Government Printing Office.
3. Onishi, Y. and S. E. Wise. 1978. Mechanisms of Transport of Sediments and Solutes in the Lower River Rhône. PNL-2731.
4. Onishi, Y. 1977. Finite Element Modeling of River and Estuarine Transport in Surface Waters - Transport of Sediments and Solutes in the Rhône River. BNWL-2227.

1. INTRODUCTION

Stephen A. Miller

Currently, a number of environmental transport and dose calculation models employing different data bases and assumptions are available for the assessment of the radiological impact of routine and accidental discharges. The model evaluation project initiated at Oak Ridge National Laboratory (ORNL) has had as its objectives identification of these models; evaluation of their structure, simplifying assumptions and data bases; estimation of their uncertainties; and, if possible, recommendation of the models and parameters which are best suited to particular assessment situations. Where needs are identified, recommendations are also made for further environmental and biomedical research. The models examined in this project to date include those developed for the prediction of atmospheric and hydrologic transport and deposition, terrestrial and aquatic food-chain bioaccumulation, and internal and external dosimetry.

The purpose of this particular report is to make recommendations concerning models and parameters for use in assessing the impact of breeder reactor radioactive discharges to the environment. These recommendations are based on the results of the model evaluations project to date. It should be noted, however, that this project is still underway, and these recommendations may be adjusted as new information is analyzed.

This report contains summaries of the recommendations for each of the transport and dosimetry areas listed above. These summaries reflect the work of those individuals who have contributed significantly to the

project. Further details concerning the models and parameter values discussed may be found in the literature cited.

For assessment purposes, the simplest model which can be acceptably validated is deemed the most suitable for a given set of conditions.¹ The models considered in this report are based heavily on the recommendations made by the model evaluation workshop² held as part of this project. Additional analyses of model uncertainty were also used in selecting models for inclusion in this report.³

Whenever possible, the input parameter values used in assessment models should be based on site-specific information. This, however, is often not practical. In the absence of site-specific information, default values must be chosen for each model input parameter. Often, only a single value is chosen for each parameter of interest. In reality, however, each input parameter has a distribution of values associated with it.¹

During the course of the model evaluation project, data for a number of parameters have been examined in an attempt to estimate the distribution of values associated with a given parameter. However, it was generally found that few of the parameters analyzed have a data base large enough to represent the true distribution. The assumption that the available data are representative of the true population of the parameter value of interest is also crucial as the data considered do not necessarily represent unbiased samples.⁴

In this report, information on the estimated distribution associated with a given parameter value is presented wherever possible. If the data for a given parameter were lognormally distributed, the data were

log-transformed to produce a normal distribution, and estimates of the population mean μ and standard deviation σ of the logarithms were determined. If the data were found to be normally distributed, the population mean $\bar{X}$ and standard deviation (S.D.) were estimated. If this information is available for a given parameter, the cumulative probability associated with any value chosen for the parameter can be determined.^{1,4} The values of the various parameters to be used in an assessment calculation will depend on the cumulative probability desired in the final answer. Therefore, recommendation of a single value for any parameter for use in assessment calculations biases model predictions. Before choosing a parameter value an assessor should consult the cited literature to determine the potential effect of the limitations noted above on the final results.

Such an in-depth analysis has not yet been performed for all parameters and all elements considered in this report. For these situations, the parameter values currently in use at ORNL have been entered and their origin cited. These values may change, however, as this project continues.

The dose conversion factors given in this report are based on model calculations rather than observations. It is not possible to validate these models because of the impracticality of measuring dose in human subjects. These models, however, are based on the best animal and human data currently available.

A separate section in this report is devoted to the dosimetry of ^3H and ^{14}C . These nuclides are often of special interest because of the importance of hydrogen and carbon in biological systems and because they are important effluents of various fuel cycles. Methodologies for assessing the dose to man from these nuclides are currently being evaluated

by another project at ORNL. This report presents the methodology currently used for assessment purposes at ORNL. This methodology may be modified, however, as the evaluation continues.

As mentioned previously, work on this project is continuing.

Current efforts are focused in three major areas:

1. comparison of predictions from various environmental transport models with measured field data to estimate the uncertainty in model output;
2. continued determination of distributions associated with model input parameter values;
3. development of recommendations for needed environmental and biomedical research

As this work continues it may result in modification of the information contained in this report.

REFERENCES

1. Shaeffer, D. L. 1979. *A Model Evaluation Methodology Applicable to Environmental Assessment Models*. ORNL-5507.
2. Hoffman, F. O., D. L. Shaeffer, C. W. Miller and C. T. Garten, Jr. 1978. *Proceedings of a Workshop on the Evaluation of Models used for the Environmental Assessment of Radionuclide Releases*. Gatlinburg, Tennessee, September 6-9, 1977. CONF-770901.
3. Little, C. A., and C. W. Miller. 1979. *The Uncertainty Associated with Selected Environmental Transport Models*. ORNL-5528.
4. Hoffman, F. O., D. L. Shaeffer, C. F. Baes III, C. A. Little, C. W. Miller, D. E. Dunning, Jr., E. M. Rupp, and R. W. Shor. 1979. An evaluation of uncertainties in radioecological models. IN *Proceedings of the German-Swiss Fachverband für Strahlenschutz*. Norderney, Federal Republic of Germany, October 2-6, 1978.

2. ATMOSPHERIC DISPERSION AND DEPOSITION

Charles W. Miller

One of the principal ways in which radionuclides from breeder reactor facilities reach the environment is through discharges to the atmosphere. These discharges result in doses to man as a result of:

1. exposure to the contaminated air; and
2. exposure to surfaces and ingestion of foodstuffs contaminated by radionuclides removed from the plume by deposition processes.

2.1 Atmospheric Dispersion

Atmospheric dispersion calculations provide estimates of air concentration resulting from atmospheric discharges of radionuclides. These air concentrations are then used to calculate doses to man.

2.1.1 The Gaussian plume model

The Gaussian plume model¹ is the most widely used method of estimating downwind air concentrations of radionuclides released to the atmosphere.^{2,3} It is also the most often verified atmospheric dispersion model.⁴ For a continuous point source and invariant meteorology, this model is given by

$$x = \frac{Q}{\pi u \sigma_y \sigma_z} \exp \left[-\frac{1}{2} \left(\frac{H}{\sigma_z} \right)^2 - \frac{1}{2} \left(\frac{y}{\sigma_y} \right)^2 \right] , \quad (2.1)$$

where

x = ground-level air concentration (Ci/m³),

Q = release rate (Ci/sec),

H = height of release (m),

u = mean wind speed over the height of the mixing layer (m/sec),
 σ_y, σ_z = standard deviation of a Gaussian distribution in the cross
 wind y and vertical z directions, respectively (m).

For estimating the impact of routine (i.e., continuous) radionuclide releases, the average ground-level concentration in air over a sector 22.5° wide is often used. It is given by

$$x = \frac{2.032 Q}{\bar{u} \sigma_z} \exp \left\{ -1/2 \left(\frac{H}{\sigma_z} \right)^2 \right\} \quad (2.2)$$

where

x = the downwind distance of interest, m;

$\bar{u}$ = average wind speed during time period of interest.

The Gaussian plume has a number of theoretical limitations.⁵

However, when properly used it has been found to be a very practical tool for dispersion modeling. It is

1. mathematically simple and flexible,
2. in accord with much, but not all, diffusion theory, and
3. a reliable framework for the correlation of both field studies and mathematical and physical modeling studies of atmospheric diffusion.⁵

2.1.1.1 Dispersion parameters. It has been found that the dispersion parameters, s_y and s_z , need to be carefully specified when using the Gaussian plume model.^{6,7} A number of empirically determined graphs of s_y and s_z as a function of downwind distance and atmospheric stability have been proposed.⁸ Geiß et al.⁹ have published a useful set of comparisons between some of these sets of graphs.

Whenever possible, site-specific measurements should be used to specify σ_y and σ_z .¹⁰ When sets of standard curves are used instead, at least two such sets should be used, one for surface releases and one for elevated releases. The Pasquill-Gifford curves¹ adjusted for averaging time and surface roughness have been suggested for use with the former^{10,11} (Table 2.1). For elevated releases, the curves suggested by Geiß et al. or those measured at Brookhaven National Laboratory¹² have been recommended^{10,11} (Table 2.2).

As noted above and in Tables 2.1 and 2.2, σ_y and σ_z depend on atmospheric stability. A number of different methods for classifying stability have been proposed, but these methods often give significantly different results when applied to the same meteorological data set.^{13,14} Until enough data become available to select the best method of classifying stability, the user of the Gaussian plume model must exercise care in choosing stabilities. The selection of a stability category alone can result in a factor of four difference between the lowest and highest annual average air concentration estimated from a given set of σ_y and σ_z curves.^{13,14}

2.1.1.2 Release height. The effective release height, H , is also a critical parameter in Eqs. (1) and (2).^{6,7} The value of H depends on more than just the physical height of the stack, h .¹⁵

$$H = h + h_{pr} - h_t - C, \quad (2.3)$$

where

h_{pr} = rise of plume above the release point due to buoyancy and momentum,

Table 2.1. Coefficients for specifying the Pasquill-Gifford⁹ system of dispersion parameters for six stability categories

Coefficient ^a	Atmospheric stability category					
	A	B	C	D	E	F
a_1	-0.023	-0.015	-0.012	-0.0059	-0.0059	-0.0029
a_2	0.35	0.25	0.18	0.11	0.0881	0.0541
b_1	0.88	-0.99	-1.19	-1.35	-2.88	-3.80
b_2	-0.15	0.82	0.85	0.79	1.26	1.42
b_3	0.15	0.017	0.0045	0.0022	-0.0421	-0.0551

$$\sigma_y = (a_1 \ln x + a_2)x$$

$$\sigma_z = \frac{1}{2.15} \exp(b_1 + b_2 \ln x + b_3 \ln^2 x)$$

x = downwind distance, m.

Table 2.2. Coefficients for specifying two systems of dispersion parameters for six stability categories⁹

System	Coefficient ¹	Atmospheric stability category					
		A	B	C	D	E	F
Julich ⁹ (H ≤ 75m)	σ_y^2	0.66	0.66	0.63	0.53	0.41	7.56
	σ_z^2	0.83	0.83	0.80	0.80	0.87	0.52
	σ_y	0.14	0.14	0.21	0.26	0.13	0.56
	σ_z	1.09	1.09	0.98	0.89	0.83	0.55
Julich ⁹ (H > 75m)	σ_y^2	0.34	0.37	0.40	0.43	0.46	7.56
	σ_z^2	1.00	0.94	0.88	0.82	0.76	0.52
	σ_y	0.037	0.076	0.16	0.32	0.66	1.37
	σ_z	1.23	1.12	0.96	0.88	0.63	0.47
		(B ₂)	(B ₁)	(C)	(D)		
Brookhaven ¹¹ (measured at H = 108m)	σ_y^2	0.40	0.36	0.32	0.31	0.31	0.31
	σ_z^2	0.91	0.86	0.78	0.71	0.71	0.71
	σ_y	0.41	0.33	0.22	0.062	0.062	0.062
	σ_z	0.91	0.86	0.78	0.71	0.71	0.71

$$\sigma_y^2 = \sigma_y^2 x^2$$

$$\sigma_z^2 = \sigma_z^2 x^2$$

x = downwind distance, m.

h_t = maximum terrain height (above the stack base) between the release point and the point for which the calculation is being made ($h_t \geq 0$),

C = downwash correction factor.

Values of h_{pr} are usually calculated through the use of models such as those suggested by Briggs.¹⁶ Models have also been proposed for calculating C.¹⁵ The need to include estimates of h_{pr} , h_t , and C in estimates of H for a given radiological assessment will depend on the particular facility under consideration and its location.

2.1.1.3 Radioactive decay. The Gaussian model, as expressed by Eqs. (2.1) and (2.2), assumes no change in the species of material released as it is being transported downwind. This, of course, is not the case when the released material is radioactive. As a result, air concentrations of radionuclides calculated using the model should be corrected to account for decay of the material during transit and subsequent buildup of daughter products.¹⁷

2.1.2 Other atmospheric dispersion models

There are a number of situations commonly encountered in radiological assessments for which the Gaussian plume model may not apply or may be difficult to parameterize. These include instances involving complex terrain, long range transport, time-varying meteorology, and variable release rates. As a result, a large number of more complex, seemingly more realistic, dispersion models have been developed. However, to run properly, these models often require a much more extensive input data base than the Gaussian model, a computer with large storage capacity, and a long computer running time for each simulation desired. These

conditions can severely limit the practicality of using a more complex model in assessment activities. The trajectory model¹⁸ is gaining acceptance as a tool for estimating dispersion on a continental scale.¹⁹ In general, however, more field data are needed to specify clearly when a given complex model should supplement the Gaussian plume model in radiological assessments.

2.2 Deposition

Particulates and reactive gases may deposit on the surface of the earth through the processes of dry and wet deposition. These processes affect doses to man from atmospheric releases of radionuclides in two ways:

1. deposited material serves as a source of surface and/or food chain contamination;
2. deposition results in a reduction in the amount of material transported downwind in air through plume depletion.

2.2.1 Dry deposition

Dry deposition is the process by which particles and reactive gases deposit on various surfaces (soil, grass, leaves, etc.) via impingement, electrostatic interactions, chemical reactions, and other processes.

The rate of deposition $d(\text{Ci}/\text{m}^2 \cdot \text{sec})$ is given by²⁰

$$d = x v_d \quad , \quad (2.4)$$

where

x = ground-level air concentration (Ci/m^3),

v_d = deposition velocity (m/sec).

The deposition velocity defined in Eq. (2.4) is a transfer factor relating an air concentration to a surface deposition rate. Field measurements of v_d , however, are generally based on measured concentrations in vegetation cut at a specific height above ground.^{21,22} Thus, an estimate of v_d appropriate for the total deposit on a unit area basis is derived from a v_d specific for deposition onto vegetation by¹⁷

$$v_d (\text{total}) = v_d (\text{vegetation})/R \quad (2.5)$$

where

R = fraction of the total material being deposited which is intercepted by the surface of interest.

Values of v_d (vegetation) for forage grasses have been determined:²³

1. 0.02 m/sec for reactive gases (molecular iodine),
2. 0.001 m/sec for small particulates (<4 μm diam), and
3. 0.0001 m/sec for relatively unreactive gases (CH_3I).

Using a mean forage grass interception fraction value of $R = 0.57$ (Sect. 3) results in the following values of v_d (total):

1. 0.035 m/sec for reactive gases,
2. 0.0018 m/sec for small particulates, and
3. 0.00018 m/sec for relatively unreactive gases.

If the plume traverses surfaces other than grasslands, other values of v_d (total) should be considered if they are available.²²

2.2.2 Wet deposition

Wet deposition is the process by which particles or gases are scavenged from a plume by rain or snow and deposited on ground surfaces. The rate of deposition on ground surfaces from these processes, w , is often estimated by²⁴

$$\omega = \phi x_{va} L, \quad (2.6)$$

where

ϕ = fraction of material scavenged from a vertical column of air per unit time (sec^{-1}),

x_{va} = average concentration in vertical column (Ci/m^3),

L = height of vertical column (m).

The value chosen for L is often the depth of the tropospheric mixing layer.²⁴

It has also been suggested that the wet deposition rate for long-term average situations can be calculated by using Eq. (2.4) and a wet deposition velocity v_w which is given by¹¹

$$v_w = \frac{k_0 p}{x_0} = w_r p, \quad (2.7)$$

where

k_0 = surface level concentration in the precipitation (Ci/m^3),

x_0 = surface level air concentration (Ci/m^3),

p = amount of precipitation per unit time (m/year),

$w_r = \frac{k_0}{x_0}$ = washout ratio.

A comparison of results from these two methods (Eqs. 2.6 and 2.7) using annual average meteorological data for Oak Ridge, Tennessee, indicates that beyond 1 km downwind from the source there is little difference between them.²⁵ For considering wet deposition from a single event, however, Eq. (2.6) is the more appropriate method to use.¹¹

2.2.3 Plume depletion

Deposition processes deplete the airborne plume as it travels downwind. The most common method of accounting for this removal is

reducing the release rate of the material. In the case of the Gaussian model [Eq. (2.1)] this new release rate Q' is given for dry deposition by

$$Q' = Q \exp \left\{ - \left(\frac{2}{\pi} \right)^{1/2} \frac{v_d(\text{total})}{u} \int_0^x \frac{\exp \left[- 1/2 \left(\frac{H}{\sigma_z} \right)^2 \right]}{\sigma_z} dx' \right\}. \quad (2.8)$$

A number of other methods of accounting for plume depletion have been proposed, but until validation data become available for determining which one is really best, the continued use of Eq. (2.7) seems reasonable.²⁵

If wet deposition is estimated using Eq. (2.6) and a scavenging coefficient ϕ' , the reduced release rate is given by²⁴

$$Q' = Q \exp(-\phi' t), \quad (2.9)$$

where t is the time required for the plume to reach a given point downwind, and ϕ' is averaged over the entire time period of interest, including periods without precipitation. If Eq. (2.7) is used to estimate wet deposition, however, Eq. (2.8) may be used to estimate plume depletion by substituting v_w for v_d .

2.2.4 Gravitational settling

Equations (2.4) and (2.8) apply when the gases or particulates in the plume are small enough not to be significantly affected by gravity as they travel downwind. If the plume contains larger particles, however, the gravitational settling may be approximated by tilting the plume downward. This is done by replacing the effective stack height H by²⁴

$$H = \frac{v_g x}{u},$$

where

v_g = gravitational fall velocity (m/sec),

x = downwind distance of interest (m).

It must be specified that $H - \frac{v_g x}{u} \geq 0$.

2.3 Duration of Release

The point-source Gaussian plume model, Eq. (2.1) and (2.2) with the modifications discussed above, attempts to define a mean concentration discussed above, attempts to define a mean concentration field relative to a fixed location over some relatively long period of time.⁵ The averaging time of the model is that of the diffusion data upon which dispersion parameters are based. Not all sets of dispersion parameters have the same release or sampling time. For example, the Pasquill-Gifford curves are based on a sampling duration of 3 minutes¹⁰ while the Julich parameters are based on releases of generally 1 hr duration.⁹

Often times one is interested in calculating concentrations for averaging times other than the one applicable for the dispersion parameters being used. It has been suggested^{5,10} that this can be accomplished by modifying σ_y according to the following relationship:

$$\frac{\sigma_{yA}}{\sigma_{yB}} = \left(\frac{t_A}{t_B} \right)^q \quad (2.10)$$

where

σ_{yA} = horizontal dispersion parameter average over some time period of interest, t_A , and

σ_{yB} = original value of the horizontal dispersion parameter based on diffusion data averaged over some time period t_B .

Reasonable values for q have been found to be in the range of 0.25 - 0.3 for $1 \text{ hr} < t_A < 100 \text{ hr}$ and approximately equals 0.2 for $3 \text{ min} < t_A < 1 \text{ hr}$.^{5,10} A similar relationship could be applied to σ_z , but such variations in σ_z should not extend beyond a few kilometers downwind of the release point.

For calculating average concentrations over long time periods, such as a season or a year, Eq. (2.2) is generally used with a weighting for the fraction of the time that the wind blows toward the point of interest.⁵ Joint frequency distributions of wind direction, wind speed, and atmospheric stability can be constructed for many locations from climatological records.

The methods discussed in this chapter must be further modified to calculate dispersion from sudden, explosive, or very short term (i.e., $t_B < 3 \text{ min}$) releases of material to the atmosphere. In general, the values of σ_y and σ_z used for such releases are considered to be different than the values presented above for longer time releases.¹ Also, dispersion in the downwind direction must be taken into consideration for very short term releases. However, virtually nothing is known about the downwind dispersion parameter, σ_x .⁵ Accidental releases from breeder reactor facilities have generally been considered to be of long enough duration for the methods considered in this section to be applied.²⁶

REFERENCES

1. Gifford, F. A., Jr. 1968. An outline of theories of diffusion in the lower atmosphere. Chapter 3. IN Siade, D. (ed.), *Met. W-ology and Atomic Energy*. Vol. USAEC TID-24190.
2. Hoffman, F. O., C. W. Miller, D. L. Shaeffer, C. T. Garten, Jr., R. W. Shor, and J. T. Ensminger. 1977. A compilation of environmental computer codes applicable to environmental assessment of pollution releases. ORNL/TM-5830.
3. Hoffman, F. O., C. W. Miller, D. L. Shaeffer, and C. T. Garten, Jr. 1977. A compilation of computer codes for the assessment of radio-nuclides released to the environment. *Env. Sci. Tech.* 11(3):343-54.
4. Little, C. A., and C. W. Miller. 1979. *Environmental Assessment of Pollution Releases*. ORNL-5528.
5. Gifford, F. A., Jr. 1975. Atmospheric dispersion models for environmental pollution applications. Chapter 2. IN Haugen, D. A. (coord.), *Environmental Pollution Models and Environmental Impact Assessment*. American Meteorological Society, Boston.
6. Pasquill, F. 1974. *Atmospheric Diffusion* (Second Edition). Wiley, New York.
7. Weber, A. H. 1976. *Atmospheric Dispersion Models for Pollution Abatement Modeling, Part I: Review of Current Systems and Available Models*. EPA-600/4-76-030a.
8. Gifford, F. A., Jr. 1976. Turbulent diffusion-typing schemes: A review. *Env. Sci. Tech.* 10(1):68-86.

9. Geiß, H., K. J. Vogt, H. S. Ehrlich, and G. Polster. 1979. Recent results of dispersion experiments at emission heights of 50 and 100 m, IN *Proceedings of the 12th Annual Symposium of the German-Swiss Fachverband für Strahlenschutz*, October 2-6, 1978, Norderney, Federal Republic of Germany (in German). Also translated in OLS-79-43, Oak Ridge National Laboratory.
10. Hanna, S. R., G. A. Briggs, J. Deardorff, B. A. Egan, F. A. Gifford, and F. Pasquill. 1977. AMS workshop on stability classification schemes and sigma curves - summary of recommendations. *Bull. Am. Meteorol. Soc.* 58(12):1305-9.
11. Hoffman, F. O., D. L. Shaeffer, C. W. Miller, and C. T. Garten, Jr. 1978. *Proceedings of a Workshop on the Evaluation of Models used for the Environmental Assessment of Radionuclide Releases to the Environment*. Gatlinburg, Tennessee, September 6-9, 1977. CONF-770901.
12. Singer, I. A., and M. E. Smith. 1966. Atmospheric dispersion at Brookhaven National Laboratory. *Int. J. Air Water Pollut.* 10(2): 125-35.
13. Pendergast, M. M. 1976. *Estimating Diffusion Coefficients from Meteorological Data*. DP-MS-76-64.
14. Miller, C. W. 1978. A critique of the determination of atmospheric stability categories for assessing airborne releases of radionuclides. *Health Phys.* 34(5):489-92.
15. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.111, Methods for estimating atmospheric transport and dispersion of gaseous effluents in routine releases from light-water-cooled reactors, Revision 1.

16. Briggs, G. A. 1969. *Plume Rise*. AEC Critical Review Series, TID-25075.
17. Moore, R. E., C. F. Baes III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller. 1979. *AIRDCS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides*. ORNL-5532.
18. Heffter, J. L., and A. D. Taylor. 1975. *A Regional-Continental Scale Transport, Diffusion, and Deposition Model, Part I: Trajectory Model*. NOAA Technical Memorandum ERL-ARL-50.
19. Travis, C. C., A. P. Watson, L. M. McDowell-Boyer, S. J. Cotter, M. L. Randolph, and D. E. Fields. 1979. *A Radiological Assessment of Radon-222 Released from Uranium Mills and Other Natural and Technologically Enhanced Sources*. NUREG/CR-0573. ORNL/NUREG-55.
20. Van der Hoven, I. 1968. Deposition of particles and gases. pp. 202-8. IN Slade, D. (ed.), *Meteorology and Atomic Energy 1968*. USAEC TID-24190.
21. Hoffman, F. O. 1977. A reassessment of the deposition velocity in the prediction of the environmental transport of radioactivity releases. *Health Phys.* 32(5):437-41.
22. Miller, C. W., F. O. Hoffman, and D. L. Shaeffer. 1978. The importance of variations in the deposition velocity assumed for the assessment of airborne radionuclide releases. *Health Phys.* 34(6):730-4.
23. Heinemann, K., and K. J. Vogt. 1979. Messungen zur ablagerung and biologischen halbwertszeit von jod auf vegetation. IN *Proceedings of the 12th Annual Symposium of the German-Swiss Fachverband für Strahlenschutz*. Norderney, Federal Republic of Germany, October 2-6, 1978.

24. Moore, R. E. 1977. *The AIRDOS-II Computer Code for Estimating Radiation Dose to Man from Airborne Radionuclides in Areas Surrounding Nuclear Facilities.* ORNL-5245.
25. Miller, C. W., and F. O. Hoffman. A critique of methods for estimating plume depletion and deposition of airborne radionuclides. *IN Proceedings of the 12th Annual Symposium of the German-Swiss Fachverband für Strahlenschutz.* Norderney, Federal Republic of Germany, October 2-6, 1978.
26. U.S. Atomic Energy Commission. 1974. *Environmental Statement, Liquid Metal Fast Breeder Reactor Program.* WASH-1535.

3. TERRESTRIAL FOOD CHAIN TRANSPORT

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A number of computer codes are available with which terrestrial transport of radionuclides through food chains following release to the atmosphere may be quantitatively estimated. The majority of these models were derived from the HERMES computer code,¹ which was developed for assessing transport in chronic, or routine, release situations.² Distinct from these HERMES-based models is TERMOD,³ a model developed for assessing terrestrial transport under routine release conditions or following acute, or accidental, radionuclide releases to the environment. All of these models, however, utilize equilibrium transfer coefficients to quantify the transport between food chain compartments following deposition of airborne radionuclides on soil or interception by vegetation. The purposes of this section are first to define the parameters for which values are needed in currently implemented terrestrial transport models at Oak Ridge National Laboratory (ORNL), and secondly, to present the parameter values used in these models.

3.1 Terrestrial Models

Terrestrial transport models currently in use at ORNL for assessing routine radionuclide releases are implementations of models provided in the October 1977 draft of the U. S. Nuclear Regulatory Commission (NRC) Regulatory Guide 1.109, Appendix C.⁴ These latter models were originally provided for assessing transport of light-water reactor effluents but have been used to assess terrestrial nuclide transport in

general. For breeder reactor effluents, it may be desirable to consider resuspension of nuclides following discontinuation or interruption of plant operation, when this phenomenon may be a significant source of airborne nuclides, and to consider buildup of daughter nuclides as was mentioned previously (Sect. 2.1.1.3). These models consider the concentration of radionuclides in vegetation as a result of deposition onto plant tissues and root uptake of activity initially deposited on soil, as well as concentrations in milk and beef following grazing on contaminated vegetation by dairy and beef cattle. Many shortcomings of equilibrium models such as the following are recognized, and thus, work is underway to clarify, evaluate, and possibly rectify some of these shortcomings. However, at present, our discussions will be restricted to these currently implemented models.

3.1.1 Concentrations in vegetation

The following equation⁴ is used for estimating the concentration $C_i^v(r, \theta)$ of nuclide i in and on vegetation at the location (r, θ) .

$$C_i^v(r, \theta) = d_i(r, \theta) \frac{R[1 - \exp(-\lambda_{Ei} t_e)]}{\lambda_{Ei}} + \frac{B_{iv}[1 - \exp(-\lambda_i t_b)]}{\lambda_i} \exp(-\lambda_i t_h), \quad (3.1)$$

where

$C_i^v(r, \theta)$ is measured in pCi/kg;

$d_i(r, \theta)$ is the deposition rate of radionuclide i onto ground at location (r, θ) , in pCi/m²-hr;

R is the fraction of depositing activity intercepted by crops, dimensionless;

λ_i is the radioactive decay constant of nuclide i , in hr⁻¹;

λ_{Ei} is the effective removal rate constant for radionuclide i from crops, in hr^{-1} , where $\lambda_{Ei} = \lambda_i + \lambda_w$,

λ_w is the removal rate constant for physical loss by weathering, in hr^{-1} ;

t_e is the time period that crops are exposed to contamination during the growing season, in hours;

Y_v is the agricultural productivity (yield) of the edible portion, in kg/m^2 ;

B_{iv} is the concentration factor for uptake of radionuclide i from soil by edible parts of crops, in pCi/kg plant tissue per pCi/kg dry soil;

t_b is the period of long-term buildup for activity in soil, in hours;

P is the effective "surface density" for the top 15 cm of soil, in $\text{kg (dry soil)}/\text{m}^2$;

t_h is a holdup time that represents the time interval between harvest and consumption of the food, in hours.

Regulatory Guide 1.109 (ref. 4) makes a distinction between the value of t_e , Y_v , and t_h appropriate for forage grasses and those appropriate for crops and leafy vegetables. The ORNL implementation of Eq. (3.1) (ref. 5) further distinguishes the C_i^V calculated for pasture grasses from a C_i^V calculated for fresh vegetables and produce consumed by man by inputting separate values of B_{iv} and R or R/Y_v for each of these categories. For pasture grasses, $Y_{v(1)}$ and $B_{iv(1)}$, based on dry weight concentrations in both vegetation and soil, are used for Y_v and B_{iv} . When calculating concentrations in fresh produce consumed by man, $Y_{v(2)}$ and $B_{iv(2)}$ are used

for Y_v and B_{iv} values, based on fresh-weight concentrations in vegetation and dry-weight soil concentrations of a nuclide.

3.1.2 Concentrations in milk

The concentration of radionuclide i in milk depends upon the amount and contamination level of the feed consumed by the animal. The concentration of radionuclide i in the animal's feed is calculated by use of the equation

$$C_i^V(r, \theta) = f_p f_s C_i^P(r, \theta) + (1 - f_p f_s) C_i^S(r, \theta) \quad (3.2)$$

where

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed, in pCi/kg;

$C_i^P(r, \theta)$ is the concentration of radionuclide i on pasture grass (calculated using Eq. (3.1) with $t_h = 0$), in pCi/kg;

$C_i^S(r, \theta)$ is the concentration of radionuclide i in stored feeds (calculated using Eq. (3.1) with $t_h = 2160$ hours (90 days), in pCi/kg;

f_p is the fraction of the year that animals graze on pasture, dimensionless;

f_s is the fraction of daily feed that is pasture grass when the animals graze on pasture, dimensionless.

Using the value of $C_i^V(r, \theta)$ calculated by use of this equation, the concentration of radionuclide i in milk is estimated as

$$C_i^M(r, \theta) = F_m C_i^V(r, \theta) Q_F \exp(-\lambda_i t_f), \quad (3.3)$$

where

$C_i^M(r, \theta)$ is the concentration in milk of nuclide i in pCi/liter;

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed,
in pCi/kg;

F_m is the average fraction of the animal's daily intake of
radionuclide i which appears in each liter of milk, in
days/liter;

Q_F is the amount of dry feed consumed by the animal per day, in
kg/day;

t_f is the average transport time of the activity from the feed into
the milk and to the receptor, in days;

λ_i is the radiological decay constant of nuclide i , in days⁻¹.

3.1.3 Concentrations in meat

The radionuclide concentration in meat (usually beef) depends upon
the amount and contamination level of the feed consumed by the animal,
as in the milk pathway. Using the value of $C_i^V(r, \theta)$ as calculated in
Eq. (3.2), the radionuclide concentration in meat is estimated as

$$r_i^F(r, \theta) = F_f C_i^V(r, \theta) Q_F \exp(-\lambda_i t_s), \quad (3.4)$$

where

$C_i^F(r, \theta)$ is the concentration of nuclide i in animal flesh, in pCi/kg;

F_f is the fraction of the animal's daily intake of nuclide i
which appears in each kilogram of flesh, in days/kg;

$C_i^V(r, \theta)$ is the concentration of radionuclide i in the animal's feed,
in pCi/kg;

Q_F is the amount of dry feed consumed by the animal per day, in
kg/day;

λ_i is the radiological decay constant of nuclide i , in days⁻¹;

t_s is the average time (days) from slaughter to consumption.

For concentrations in beef, it is assumed that beef cattle are on open pasture for the same grazing periods as given for milk cattle.

3.2 Terrestrial Transport Input Parameters

Following are lists of input parameter values currently used at ORNL in assessing terrestrial transport resulting from routine releases. These values were derived from empirical data obtained from an ongoing review of available literature,⁵ and represent average values over a wide range of experimental and environmental conditions. Values listed for the transfer coefficients, $B_{iv(1)}$, $B_{iv(2)}$, F_m , and F_f , reflect the current status of this literature review of element-specific data pertinent to terrestrial transport, and may change when new references become available and are reviewed.

Not all transfer coefficients reported here have been evaluated in this project. Values given for these parameters are taken from other widely used documents.^{6,7} The derivations of transfer coefficients for nuclides which have been evaluated are element, rather than nuclide, specific, so that the data base for some nuclides is not severely restricted.

The parameters used in the terrestrial transport models described are designed to represent annually averaged values at equilibrium over a wide range of environmental conditions. Thus, these average parameters are best suited for generic assessments, but may be used to preliminarily assess specific sites when site specific parameters are not available. Empirical data used in deriving values for these parameters may not

always reflect annual average or equilibrium conditions, nor field conditions when laboratory or otherwise controlled experiments were conducted. For technetium, appreciable uptake of the element from soil by plant roots may result in nonequilibrium soil concentrations in laboratory studies concerning technetium uptake. For this reason and because the transport model used does not consider soil loss of nuclides via crop harvest or leaching, the assessment of technetium transport using the concentration factor approach may result in overestimates of plant concentrations. Furthermore, empirical data may not adequately reflect the true distribution of values associated with each parameter under various conditions. Thus, caution should be used in the interpretation of parameter values presented and results generated in implementation of the models.

Statistical distributions have been estimated for Y_v , f_p , f_s , Q_F , and P ,⁸ and values describing the distributions for each of these parameters are provided here. For B_{iv} , F_m , and F_f , an arithmetic mean value was calculated from data obtained from each reference cited. These mean values were used in deriving an arithmetic mean for all references collected for a particular element. Statistical distributions have not yet been developed for these latter parameters.

Terrestrial transport of ^3H and ^{14}C is addressed in Sect. 8 of this report, and thus, transfer coefficients for these nuclides are not included in this section.

3.2.1 Agricultural productivity by unit area, $Y_{v(1)}$ and $Y_{v(2)}$

Agricultural productivity, in kg/m^2 , is given in dry weight in this report for pasture grasses, $Y_{v(1)}$, and in fresh weight for leafy

vegetables and produce ingested directly by man, $Y_v(2)$. This method of reporting parameter values for Y_v has been found to be most directly useful in the transport models used. Table 3.1 presents mean values of $Y_v(1)$ and $Y_v(2)$ derived from the review and analysis of agricultural productivity by Hoffman and Baes along with the μ and σ , representing the mean and standard deviation of the log-transformed distribution, respectively.

3.2.2 Interception fractions for above-ground plant portions, R_1 and R_2

The fraction of atmospherically depositing radionuclides intercepted and initially retained on above-ground portions of either forage crops or leafy vegetables and fresh produce ingested by man is symbolized by R_1 and R_2 , respectively, where $R_1, R_2 \leq 1$. The remaining fraction deposited on the soil and surfaces other than the vegetation is merely $(1-R_1)$ and $(1-R_2)$.

Measurements of R_1 for specific values of $Y_v(1)$ have been made and reviewed by Hoffman and Baes and the resultant mean value of $R_1/Y_v(1)$ is given in Table 3.1. Measurements of the interception fraction R_2 specific for edible portions of leafy vegetables and fresh produce ingested directly by man are unavailable. It is expected that the value of R_2 will be less than that of R_1 since vegetable crops are usually cultivated in rows. On a unit area basis, this spacing of vegetable crops exposes more surface soil than the dense spacing of forage crops. Therefore, a value of 0.2 for R_2 , provided in the USNRC Regulatory Guide 1.109 (ref. 4) is currently being used. These R_1 and R_2 values are being used for both wet and dry deposition.

Table 3.1. Estimated values of above-ground agricultural productivity and interception fractions for forage grasses and edible portions of vegetable crops

Parameter	Mean value (kg/m ²)	μ	σ	Ref.
R_1/Y_{v1} (Y_{v1} measured in dry weight- forage grasses)	2.0	0.61	0.44	8
Y_{v2} measured in fresh weight (leafy vegetables)	1.9	-0.73	0.46	8
Y_{v2} measured in fresh weight (nonleafy vegetables)	0.57	0.43	0.52	8
R_2 (vegetables)	0.20			4

R_1 defined as interception fraction for forage grasses.

The estimates of the mean are the inverse of values derived by Hoffman and Baes⁸ (μ and σ are specific for $1/Y_{v2}$).

The distribution described by Hoffman and Baes⁶ is based on edible portions of cabbage lettuce, and spinach.

The distribution described by Hoffman and Baes⁶ is based on edible portions of broccoli, cauliflower, green peas, lima beans, and sweet corn.

R_2 defined as interception fraction for edible portions of leafy vegetables and fresh produce. No statistical parameters are available.

3.2.3 Plant/soil bioaccumulation factor, B_{iv}

Tables 3.2 and 3.3 contain values of $B_{iv(1)}$ and $B_{iv(2)}$, representing the transfer of elements from soil to grass leaves and portions of vegetables and produce considered edible by man, respectively. The following definitions of $B_{iv(1)}$ and $B_{iv(2)}$ apply:

$$B_{iv(1)} = \frac{\text{radionuclide concentration in entire above-ground portion of plant at maturity per unit dry wt}}{\text{radionuclide concentration in soil per unit dry wt.}}$$

$$B_{iv(2)} = \frac{\text{radionuclide concentration in edible portion of plant at maturity per unit fresh wt.}}{\text{radionuclide concentration in soil per unit dry wt.}}$$

As was mentioned previously, arithmetic means of results obtained from empirical studies concerning root uptake for each element are currently being used for B_{iv} values.

The parameter B_{iv} is interpreted as the elemental concentration in plant tissues at maturity resulting from an equilibrium soil concentration to which the root are exposed during the growing season. For Tc, however, appreciable uptake of the element from soil by plant roots may result in non-equilibrium soil concentrations in laboratory studies concerning Tc uptake.¹⁴ Because no field data have been reported for Tc, the appropriate B_{iv} cannot be directly derived. The $B_{iv(2)}$ for Tc (Table 3.2) is simply a conversion of the $B_{iv(2)}$ value to represent a dry-weight concentration in forage. No distinction between plant portions or species was made in this case, due to the lack of empirical data on different plant types.

Table 3.2. Values of $B_{iv(l)} [(element)_{forage} : (element)_{soil}]$ for forage and feed

Element	$B_{iv(l)}^a$	References
Na	2.1×10^{-1}	6^b
P	4.4	6^b
S	2.4	6^b
Mn	1.2×10^{-1}	6^b
Fe	2.6×10^{-3}	6^b
Co	3.8×10^{-2}	6^b
Ni	7.6×10^{-2}	6^b
Zn	1.6	6^b
Sr	1.2	9-12
Y	1.1×10^{-2}	9,13
Zr	6.8×10^{-4}	6^b
Nb	3.8×10^{-2}	6^b
Tc	2.0×10^2	14
Ru	1.7×10^{-1}	9,13
Ag	6.0×10^{-1}	6^b
Sb	4.4×10^{-2}	6^b
Te	5.2	6^b
I	2.0×10^{-1}	15
Cs	1.5×10^{-1}	9,11,13,16-20
Ce	3.9×10^{-2}	9,13
Pm	1.3×10^{-3}	13
Sm	1.0×10^{-2}	6^b
Eu	1.0×10^{-2}	6^b
Pb	1.4×10^{-1}	21-24
Bi	6.0×10^{-1}	6^b
Po	4.2×10^{-3}	25

Table 3.2. (continued)

Element	$B_{iv(1)}^a$	References
Ra	9.1×10^{-2}	26-28
Ac	1.0×10^{-2}	6 ^b
Th	2.7×10^{-3}	29
Pa	1.0×10^{-2}	6 ^b
U	6.1×10^{-3}	29,30
Np	1.0×10^{-2}	6 ^b
Pu	2.0×10^{-3}	11,13,30-34
Am	2.1×10^{-3}	29-31,34
Cm	4.8×10^{-4}	32

^a $B_{iv(1)}$ is derived for dry-weight concentrations in forage, hay, or feed and in soil.

^bDerived by converting fresh-weight plant concentrations (C_p) to dry-weight, assuming 25% dry matter content, and dividing this value by dry-weight soil concentrations, C_s .

^cConverted from a given fresh-weight concentration factor to dry-weight, by assuming 25% dry matter for plants.

^dSee preceding discussion.

Table 3.3. Values of $B_{iv(2)}$ [(element) edible tissue:
(element)_{soil}] for vegetables, fruits, and grains

Element	$B_{iv(2)}^a$	References
Na	5.2×10^{-2}	6 ^b
P	1.1	6 ^b
S	5.9×10^{-1}	6 ^b
Mn	2.9×10^{-2}	6 ^b
Fe	6.6×10^{-4}	6 ^b
Co	9.4×10^{-3}	6 ^b
Ni	1.9×10^{-2}	6 ^b
Zn	4.0×10^{-1}	6 ^b
Sr	2.9×10^{-1}	9,11,12,35
Y	4.3×10^{-3}	9
Zr	1.7×10^{-4}	6 ^b
Nb	9.4×10^{-3}	6 ^b
Tc	5.0×10^1	14 ^c
Ru	1.6×10^{-2}	9
Ag	1.5×10^{-1}	6 ^b
Sb	1.1×10^{-2}	6 ^b
Te	1.3	6 ^b
I	5.5×10^{-2}	15
Cs	1.1×10^{-2}	9,11,19,20,35,36
Ce	6.2×10^{-3}	9
Pm	2.5×10^{-3}	6 ^b
Sm	2.5×10^{-3}	6 ^b
Eu	2.5×10^{-3}	6 ^b
Pb	3.9×10^{-3}	21,23,37,38
Bi	1.5×10^{-1}	6 ^b
Po	2.6×10^{-4}	25

Table 3.3. (continued)

Element	$B_{iv(2)}^a$	References
Ra	1.3×10^{-2}	26,27,39,40
Ac	2.5×10^{-3}	6 ^b
Th	3.5×10^{-4}	41
Pa	2.5×10^{-3}	6 ^b
U	2.9×10^{-4}	30,41
Np	2.5×10^{-3}	6 ^b
Pu	2.2×10^{-4}	11,30,33,42-44
Am	4.0×10^{-4}	29
Cm	1.7×10^{-3}	15

^a $B_{iv(2)}$ is derived for a fresh-weight concentration in edible plant tissues and a dry-weight concentration in soils.

^bDerived by dividing the fresh-weight plant concentration (C_p) by dry-weight soil concentrations (C_s).

^cSee preceding discussion.

3.2.4 Milk transfer coefficient, F_m

The milk transfer coefficient, F_m , represents the fraction of the total daily intake of a nuclide which is transferred to a liter of the cow's milk at equilibrium. Values of F_m can be derived from a variety of reported data according to the methodologies described by Ng et al.⁷

Determinations of F_m (Table 3.4) were made from literature sources reporting empirically-derived results, whenever available. However, for many elements, the well-documented literature review by Ng et al.⁷ was the primary literature source. References in which the chemical form of the nuclide administered orally to the cow was clearly atypical of forms found in the environment were excluded from the analysis of F_m . As for the analysis of B_{iv} , the values of F_m given in Table 3.4 are element- rather than nuclide-specific.

3.2.5 Meat transfer coefficient, F_f

The meat-transfer coefficient F_f represents the fraction of the total daily intake of a nuclide which is transferred to a kilogram of muscle in the meat producing animal at equilibrium. It is assumed that equilibrium conditions exist when slaughter occurs.

A review of the available literature allowed the derivation of values of F_f for beef cattle (Table 3.5). Values of F_f based on immature cattle (less than 6 months of age) were excluded from analysis when data for adult cattle were available. Data for other ruminant species were included when literature references for cattle were unavailable.

Table 3.4. Values of F_m [(element)_{milk}: elemental daily intake], day/liter

Element	F_m^a	References
Na	3.5×10^{-2}	7
P	1.6×10^{-2}	7
S	1.6×10^{-2}	7
Mn	8.4×10^{-5}	7
Fe	5.9×10^{-5}	7
Co	2.0×10^{-3}	7
Ni	1.0×10^{-2}	7
Zn	1.0×10^{-2}	7
Sr	2.4×10^{-3}	45-48
Y	2.0×10^{-5}	7
Zr	8.0×10^{-2}	7
Nb	2.0×10^{-2}	7
Tc	9.9×10^{-3}	7
Ru	6.1×10^{-7}	7
Ag	3.0×10^{-2}	7
Sb	2.0×10^{-5}	7
Te	2.0×10^{-4}	7
I	1.0×10^{-2}	7,8
Cs	5.6×10^{-3}	49-51
Ce	2.0×10^{-5}	7
Pm	2.0×10^{-5}	7
Sm	2.0×10^{-5}	7
Eu	2.0×10^{-5}	7
Pb	9.9×10^{-5}	52-59
Bi	5.0×10^{-4}	7
Po	1.2×10^{-4}	58
Ra	5.9×10^{-4}	59
Ac	2.0×10^{-5}	7
Th	5.0×10^{-6}	7

Table 3.4. (continued)

Element	F_m^a	References
Pa	5.0×10^{-6}	7
U	1.2×10^{-4}	60
Np	5.0×10^{-6}	7
Pu	4.5×10^{-8}	61,62
Am	2.0×10^{-5}	7
Cm	2.0×10^{-5}	7

^aThe F_m is derived from fresh-weight concentrations in milk and dry-weight concentrations in forage, feed, or hay.

Table 3.5. Values of F_f [(element)_{beef}: elemental daily intake], day/kg

Element	F_f^a	References
Na	3.8×10^{-2}	6^b
P	5.7×10^{-2}	6^b
S	1.3×10^{-1}	6^b
Mn	1.0×10^{-3}	6^b
Fe	5.0×10^{-2}	6^b
Co	1.7×10^{-2}	6^b
Ni	6.7×10^{-3}	6^b
Zn	3.8×10^{-2}	6^b
Sr	3.0×10^{-4}	15
Y	5.8×10^{-3}	6^b
Zr	4.3×10^{-2}	6^b
Nb	3.5×10^{-1}	6^b
Tc	8.7×10^{-3}	6^b
Ru	1.8×10^{-3}	15
Ag	2.2×10^{-2}	6^b
Sb	5.0×10^{-3}	6^b
Te	9.6×10^{-2}	6^b
I	7.0×10^{-3}	15
Cs	1.4×10^{-2}	63
Ce	6.0×10^{-4}	15
Sm	6.3×10^{-3}	6^b
Eu	6.0×10^{-3}	6^b
Pb	9.1×10^{-4}	57
Bi	1.7×10^{-2}	6^b
Po	4.0×10^{-3c}	64-66

Table 3.5. (continued)

Element	F_f^a	References
Ra	5.5×10^{-4}	65,67 ^c
Ac	1.6×10^{-6}	d
Th	1.6×10^{-6}	d
Pa	1.6×10^{-6}	d
U	1.6×10^{-6}	d
Np	1.6×10^{-6}	d
Pu	4.1×10^{-7}	15,62
Am	1.6×10^{-6}	15
Cm	1.6×10^{-6}	15

^aThe F_f is derived from fresh-weight concentrations in beef and dry-weight concentrations in forage, feed, or hay.

^bDerived by converting fresh-weight plant concentrations (C_p) to dry-weight, assuming 25% dry matter content, such that

$$F_f = \frac{C_{\text{meat (fresh)}}}{C_p \text{ (dry)}} \times \frac{1}{Q_F}$$

^cValues given are based on sheep, caribou, and reindeer data and the associated intake rates for each.

^dValues are assumed to be the same as those for curium and americium.

3.2.6 Animal feed consumption (Q_F), grazing patterns (f_p , f_s), and effective soil surface density (P)

Values used for the parameters Q_F , f_p , f_s , and P (defined in Section 3.1) are the means of their respective statistical distributions as described by Hoffman and Baes. These means are given in Table 3.6, along with the associated μ and σ for the log-transformed distribution.

3.2.7 Time parameters t_e , t_b , t_h , t_f , and t_s

The parameters t_e , t_h , t_f , and t_s have all been defined in Section 3.1, and the values used at ORNL are listed in Table 3.7. The values listed were taken from the draft Regulatory Guide.⁴ The parameter t_b , representing the period of long-term buildup for activity in soil, was not assigned a specific value. The value of t_b is left to the user's discretion.

Table 3.6. Values of Q_F , f_p , f_s , and P used in terrestrial transport models at ORNL

Parameter	Mean	μ	σ	S.D.	Units	Reference
Q_F^a	15.6			2.6	kg/day (dry)	8
f_p^a	0.40			0.22		8
f_s^a	0.43			0.13		8
ρ^b	215	5.36	0.11		kg/m ²	8

^aParameter normally distributed; therefore, standard deviation (S.D.) is given.

^bDistribution for P derived from a lognormal distribution of soil bulk density ρ (g/cm³).

Table 3.7. Values of t_e , t_h , t_f , and t_s used in terrestrial transport models at ORNL

Parameter	Value	Units	Reference
$t_e^a: t_{e(1)}$	720	hr	4
$t_{e(2)}$	1440	hr	
$t_h^b: t_{h(1)}$	0	hr	4
$t_{h(2)}$	2160	hr	4
$t_{h(3)}$	336	hr	4
$t_{h(4)}$	336	hr	4
t_f	4	day	4
t_s	20	day	4

$t_{e(1)}$ applies to forage grasses; $t_{e(2)}$ applies to crops and leafy vegetables.

$t_{h(1)}$ applies to forage; $t_{h(2)}$ applies to stored feed for animals; $t_{h(3)}$ applies to leafy vegetables; and $t_{h(4)}$ applies to produce.

REFERENCES

1. Fletcher, J. F., and W. L. Dotson (compilers). 1971. *HERMES-A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry*. HEDL-TME-71-168.
2. Hoffman, F. O., C. W. Miller, D. L. Shaeffer, and C. T. Garten, Jr. 1977. Computer codes for the assessment of radionuclides released to the environment. *Nucl. Saf.* 18:343-54.
3. Booth, R. S., S. V. Kaye, and P. S. Rohwer. 1971. A systems analysis methodology for predicting dose to man from a radioactively contaminated terrestrial environment. IN *Proceedings of the Third National Symposium on Radioecology*, Oak Ridge, Tennessee, May 10-12, 1971. CONF-7105C1.
4. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.109. *Calculation of Annual Doses to Man From Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I, Revision 1*.
5. Moore, R. E., C. F. Baes III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller. 1979. *AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides*. ORNL-5532.
6. Ng, Y. C., C. A. Burton, S. E. Thompson, R. K. Tandy, H. K. Kretnner, and M. W. Pratt. 1968. Prediction of the maximum dosage to man from the fallout of nuclear devices. IN *Handbook for Estimating the Maximum Internal Dose from Radionuclides Released to the Environment*. UCRL-50163, pt. IV.

7. Ng, Y. C., C. S. Colsher, D. J. Quinn, and S. E. Thompson. 1977. *Transfer Coefficients for the Prediction of the Dose to Man via the Forage-Cow-Milk Pathway from Radionuclides Released to the Biosphere.* UCRL-51939.
8. Hoffman, F. O., and C. F. Baes (eds.). 1979. *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides. Final Report.* ORNL/NUREG/TM-282.
9. Romney, E. M., J. W. Neel, H. Nishita, J. K. Olafson, and K. H. Larson. 1957. Plant uptake of ^{90}Sr , ^{91}Y , ^{106}Ru , ^{137}Cs , and ^{144}Ce from soils. *Soil Sci.* 83:369-76.
10. Romney, E. M., A. J. Steen, R. A. Wood, and W. A. Rhoads. 1966. Concentration of radionuclides by plants grown on ejecta from the Sedan thermonuclear cratering detonation. pp. 391-98. IN *Radioecological Concentration Processes. Proceedings of International Symposium, Stockholm, Sweden.*
11. Hardy, E. P., B. G. Bennett, and L. T. Alexander. 1977. *Radionuclide Uptake by Cultivated Crops.* HASL-321.
12. Arkhipov, N. P., Y. A. Fedorov, E. F. Bondar, R. M. Aleksakhin, G. N. Romanov, and L. T. Feuraleva. 1974. Predicting ^{90}Sr accumulation in the crop harvest as a result of its uptake from the soil. *Sov. Soil Sci.* 6(4):412-19.
13. Rediske, J. H., J. F. Cline, and A. A. Selders. 1955. *The Absorption of Fission Products by Plants.* HW-36734.
14. Till, J. E., F. O. Hoffman, and D. E. Dunning, Jr. 1978. *Assessment of ^{99}Tc Releases to the Atmosphere - A Plea for Applied Research.* ORNL/TM-6260.

15. Ng, Y. C., W. A. Phillips, Y. E. Ricker, R. K. Tandy, and S. E. Thompson. 1978. *Methodology for Assessing Dose Commitment to Individuals and to the Population from Ingestion of Terrestrial Foods Contaminated by Emissions from a Nuclear Fuel Reprocessing Plant at the Savannah River Plant - Draft Report.* UCID-17743.
16. Barber, D. A. 1964. Influence of soil organic matter on the entry of cesium-137 into plants. *Nature* 204:1326-27.
17. Fredriksson, L., A. Eriksson, and H. Lonsjo. 1966. *Studies on Plant Accumulation of Fission Products under Swedish Conditions. VIII. Uptake of ¹³⁷Cs in Agricultural Crops as Influenced by Soil Characteristics, and Rate of Potassium Fertilization in a Three-Year Micro Plot Experiment.* Research Institute of National Defense. Stockholm, Sweden, FOA 4 Rapport A 4486-4623.
18. Fredriksson, L., H. Lonsjo, and A. Eriksson. 1969. *Studies on Plant Accumulation of Fission Products under Swedish Conditions. XII. Uptake of ¹³⁷Cs by Barley and Peas from 12 Different Top Soils Combined with 2 Subsoils in a Long-Term Micro Plot Experiment.* Research Institute of National Defense. Stockholm, Sweden, FOA 4 Rapport C 4405-4428.
19. Evans, E. J., and A. J. Dekker. 1968. Comparative Cs-137 content of agricultural crops grown in a contaminated soil. *Can. J. Plant Sci.* 48:183-88.
20. Haak, E., and A. Eriksson. 1973. *Studies on Plant Accumulation of Fission Products under Swedish Conditions. VII. Uptake of ¹³⁷Cs by Wheat and Timothy from Six Different Soils as Influenced by Rate of K-Fertilization and by Type and Rate of N-Fertilization in Pot Experiments.* The Research Institute of National Defence, Stockholm, Sweden, FOA 4 Rapport C 4557-A3.

21. Dedolph, R., G. Ter Haar, R. Holtzman, and H. Lucas, Jr. 1970. Sources of Pb in perennial ryegrass and radishes. *Environ. Sci. Technol.* 4(3):217-25.
22. Cox, W. J., and D. W. Rains. 1972. Effect of lime on Pb uptake by five plant species. *J. Environ. Qual.* 1(2):167-71.
23. Rabinowitz, M. 1972. Plant uptake of soil and atmospheric lead in southern California. *Chemosphere* 1(4):175-80.
24. Zimdahl, R. L., D. T. McCreary, and S. M. Gwynn. 1978. Lead uptake by plants — the influence of lead source. *Bull. Environ. Contam. Toxicol.* 19(4):431-35.
25. Watters, R. L., J. E. Johnson, and W. R. Hansen. 1969. *A Study of Unsupported Polonium-210 for Ion Exchange in Soil and Uptake in Vegetation.* C00-1733-3, Colorado State University, Fort Collins, Colo.
26. Kirchmann, R., R. Boulenger, and A. LaFontaine. 1968. Absorption of ^{226}Ra in cultivated plants. pp. 1045-51 IN W. S. Snyder *et al.* (eds.). *Proceedings of the IRPA Congress on Radiation Protection.* Rome, 1966, Vol. II. Pergamon Press, New York.
27. DeBortoli, M., and P. Gaglione. 1972. Radium-226 in environmental materials and foods. *Health Phys.* 22(1):43-48.
28. Taskayev, A. K., V. Ya. Ovchenkov, R. M. Aleksakhin, and I. I. Shirkotomova. 1977. Uptake of ^{226}Ra by plants and changes in its state in the soil-plant tops-litterfall system. *Pochvovedeniye* 2:42-48.
29. Bondfetti, E. A., J. R. Trabalka, C. T. Garten, and G. G. Killough. In press. Biochemistry of actinides: A nuclear fuel cycle perspective. IN *Radioactive Waste in Geologic Storage.* ACS Symposium Service.

30. Adams, W. H., J. R. Buchholz, C. W. Christenson, G. L. Johnson, and E. B. Fowler. 1975. *Studies of Plutonium, Americium, and Uranium in Environmental Matrices*, LA-5661.
31. Romney, E. M., H. M. Mork, and K. H. Larson. 1970. Persistence of plutonium in soil, plants, and small mammals. *Health Phys.* 19: 487-91.
32. Price, K. R. 1972. Uptake of ^{237}Np , ^{239}Pu , ^{241}Am , and ^{244}Cm from Soil by Tumbleweed and Cheatgrass. BNWL-1688.
33. Brown, K. W., and J. C. McFarlane. 1978. Plutonium uptake by plants grown in soil containing plutonium-238 dioxide particles. *Health Phys.* 35:481-85.
34. Cline, J. F. 1968. Uptake of ^{241}Am and ^{239}Pu by plants. pp. 8.24-8.25. IN Thompson, R. C., and Paulette Teal (eds.). *Pacific Northwest Laboratory Annual Report for 1967 to the USAEC Division of Biology and Medicine, Vol. I*. BNWL-714.
35. Essington, E., H. Nishita, and A. Wallace. 1962. Influence of chelates on availability of fission products to plants grown in a contaminated soil. *Soil Sci.* 94:96-105.
36. Schulz, R. K. 1965. Soil chemistry of radionuclides. *Health Phys.* 11:1317-24.
37. Ter Haar, G. 1970. Air as a source of lead in edible crops. *Environ. Sci. Technol.* 4(3):226-30.
38. John, M. K., and C. J. Van Laerhoven. 1972. Lead distribution in plants grown on a contaminated soil. *Environ. Lett.* 3(2):111-16.

39. Vavilov, P. P., O. N. Papova, and R. P. Kodaneva. 1964. The behavior of radium in plants. *Dokl. Akad. Nauk.-S.S.S.R.* 517: 992-94 (translated abstract *Nucl. Sci. Abstr.* 19(2):2016).
40. Mordberg, E. L., V. M. Aliksandruk, G. F. Kovygin, I. I. Shevchenko, V. M. Blyumshtein, and G. F. Yushkevich. 1975. The passage of the isotopes of the uranium-radium series into the grain of certain agricultural crops. *Gig. Sanit.* 40(2):58-61.
41. Bondietti, E. A., C. T. Garten, Jr., C. W. Francis, and L. D. Eyman. In preparation. Uranium and thorium in the biosphere: A review emphasizing behavior applicable to the thorium fuel cycle. *Journal of Environmental Quality.*
42. Bondietti, E. A., and F. H. Sweeton. 1976. Transuranic speciation in the environment. pp. 449-76. IN *Transuranics in Natural Environments*. A Symposium at Gatlinburg, Tennessee, October 1976. NVO-178.
43. Schulz, R. K., G. A. Tompkins, and K. J. Babcock. 1975. Uptake of plutonium and americium by plants from soil: Uptake by wheat from various soils and effect of oxidation of plutonium added to soil. IN *Proceedings of International Symposium on Transuranic Nuclides in the Environment*. San Francisco, 17-21 November, 1975, Vienna, IAEA (1976) as quoted by Bulman, R. A. 1976. *Concentration of Actinides in the Food Chain*. NRPB-R44, Harwell, Didcot, Oxon. OX11 0RQ. June 1976.
44. Dahlgren, Roger C., Ernest A. Bondietti, and L. Dean Eyman. 1976. Biological pathways and chemical behavior of plutonium and other actinides in the environment. pp. 47-80. IN Arnold M. Friedman (ed.). *Actinides in the Environment*, ACS Symposium Series 24. American Chemical Society, Washington, D.C.

45. Squire, H. M., L. J. Middleton, B. F. Sansom, and C. R. Coid. 1958. Experiments on the metabolism of certain fission products in dairy cows. pp. 207-11. IN Extermann, R. C. (ed.). *Radioisotopes in Scientific Research, Vol. IV*, Pergamon Press, New York.
46. Cragle, R. G., and B. J. Demott. 1959. Strontium and calcium uptake and excretion in lactating dairy cows. *J. Dairy Sci.* 42(8): 1367-72.
47. Garner, P. J., H. G. Jones, and B. F. Sansom. 1960. Fission products and the dairy cow: 2. Some aspects of the metabolism of the alkaline earth elements calcium, strontium, and barium. *Biochem. J.* 76:572.
48. Comar, C. L., R. H. Wasserman, and A. R. Twardock. 1961. Secretion of calcium and strontium into milk. *Health Phys.* 7:69-80.
49. Hawthorne, H. A. 1967. "Field studies of the transfers of ^{137}Cs from fallout to milk. pp. 77-85. IN B. Aberg and F. P. Hungate (eds.). *Radiological Contamination Processes, Proceedings of an International Symposium in Stockholm, Sweden, April 1967*.
50. Ward, G. M., J. E. Johnson, and H. F. Stewart. 1965. ^{137}Cs passage from precipitation to milk. pp. 703-10 IN A. W. Klement, Jr. (ed.). *Radioactive Fallout from Nuclear War and Peace*. USAEC Symposium Series #5.
51. Johnson, J. E., G. M. Ward, E. Firestone, and K. L. Knox. 1968. Metabolism of radioactive cesium (^{134}Cs and ^{137}Cs) and potassium by dairy cattle as influenced by high and low forage diets. *J. Anim. Sci.* 94:282-88.
52. Donovan, P. O., D. T. Feeley, and P. P. Canavan. 1969. Lead contamination in mining areas in western Ireland II. Survey of animals, pastures, foods, and waters. *J. Sci. Food Agric.* 20:43-45.

53. Bovay, E. 1971. Accumulation of Pb on vegetation along expressways. Feeding tests on dairy cows with forage contamination with Pb. pp. 101-17. IN Switzerland Federal Committee for Air Hygiene Investigations: The Problem of Leaded Gasoline. *Bull. der Eidg. Gesundheitsamtes* Suppl. B, No. 3.
54. Kerin, D., and Z. Kerin. 1971. Lead contamination of milk and honey caused by industrial lead aerosols. *Prot Vitae* 2:61-62. IN *Kettering Abstracts* 9(240):206-7.
55. Stanley, R. E., A. A. Mullen, and E. W. Bretthauer. 1971. Transfer to milk of ingested radiolead. *Health Phys.* 21:211-15.
56. Lynch, G. P., D. G. Cornell, and D. F. Smith. 1974. Excretion of cadmium and lead into milk. pp. 470-74. IN Hoekstra, W. G., J. W. Suttie, H. E. Ganther, and W. Mertz (eds.). *Trace Element Metabolism in Animals - 2*. University Park Press, Baltimore, Md.
57. Nelmes, A. J., R. St. J. Buxton, F. A. Fairweather, and A. E. Martin. 1974. The implication of the transfer of trace metals from sewage sludge to man. pp. 145-53. IN *Proceedings of the University of Missouri 8th Annual Conference on Trace Substances in Environmental Health*.
58. McInroy, J. F. 1973. *Polonium-210 Metabolism in Ruminants*. Ph.D. Dissertation. Colorado State University. Fort Collins, Colo.
59. Kirchmann, R., A. Lafontaine, J. van der Hoek, and G. Koch. 1972. Comparison of the rate of transfer to cow milk of ^{226}Ra from drinking water and ^{226}Ra incorporated in hay. *C. Soc. Biol.* 166 (11):1557-62.
60. Chapman, T. S., and S. Hammons, Jr. 1963. Some observations concerning uranium content of ingesta and excreta of cattle. *Health Phys.* 9:79-81.

61. Sansom, B. F. 1964. The transfer of plutonium-239 from the diet of a cow to its milk. *Brit. Vet. J.* 120:158-61.
62. Garten, C. T. 1978. A review of parameter values used to assess the transport of plutonium, uranium, and thorium in terrestrial food chains. *Environmental Research* (in press).
63. Ward, G. M., and J. E. Johnson. 1965. The cesium-137 content of beef from dairy and feed-lot cattle. *Health Phys.* 11:95-100.
64. Beasley, T. M., and H. E. Palmer. 1966. Lead-210 and polonium-210 in biological samples from Alaska. *Science* 152:1062-64.
65. Holtzman, R. B. 1966. Natural levels of lead-210, polonium-210 and radium-226 in humans and biota of the Arctic. *Nature* 210:1094-97.
66. Hill, C. R. 1965. Polonium-210 in man. *Nature* 208(5009):423-28.
67. Holtzman, R. B. 1966. ^{226}Ra and the natural airborne nuclides ^{210}Pb and ^{210}Po in Arctic biota. pp. 1087-96. IN Snyder, W. S. (ed.). *Proceedings of the First International Congress of Radiation Protection*. IRPA. Rome, 5-10 September, 1966.

4. SURFACE WATER TRANSPORT

This section will briefly discuss models which we feel are best suited to assess radionuclide releases from breeder reactors to hydrologic (surface water) systems. The models which we consider are two-dimensional, either longitudinal-transverse or longitudinal-vertical. The models also differ in their capability to account for sorptive effects on the pollutant. The Yotsukura-Sayre model (longitudinal-transverse) does not consider sorptions, but the SERATRA (longitudinal-vertical) and FETRA (longitudinal-transverse) models do consider sorption and sediment transport. These models and their parameter needs will be discussed separately.

4.1 Model Without Sorption

The Yotsukura-Sayre model,¹ a continuous release equilibrium model, is basically the same as that of Yotsukura and Cobb.² These models employ an orthogonal curvilinear coordinate system to describe the geometrical configuration of the channel. In its most fundamental form the model can be expressed as follows:³

$$uD \frac{\partial C_i}{\partial x} - \frac{\partial}{\partial y} \left(K_y D \frac{\partial C_i}{\partial y} \right) = (-D)C_i, \quad (4.1)$$

where

C_i = concentration of radionuclide i (Ci m^{-3}),

u = stream velocity (m sec^{-1}),

D = stream depth (m),

x = distance downstream (m),

y = distance across stream (m),

K_y = lateral turbulent diffusion coefficient ($m^2 sec^{-1}$), and

λ = radioactive decay constant = $0.693/\text{half life (sec}^{-1}\text{)}$.

Of these parameters, all but K_y can be measured directly in the stream of concern. However, Yotsukura and Cobb² showed that the variable diffusion factor, $K_y u D^2$, may be replaced by a constant factor, $\overline{K_y u D^2}$ where

$$\overline{K_y u D^2} = \frac{1}{Q} \int_0^Q K_y u D^2 dq, \quad (4.2)$$

Q is the total river discharge and q is the transverse cumulative discharge measured outward from the near shore. Further, Yotsukura and Cobb² suggest that K_y can be determined using the formula of Elder⁴:

$$K_y = \beta D u^*, \quad (4.3)$$

where β is a unitless constant, D is stream depth in m and u^* is the average shear velocity in the y dimension. Therefore, the only constant which may be considered not site specific for the model is the constant β . The value of β in flume studies has been shown to range from about 0.11 to 0.23.² A value of $\beta = 0.72$ was found in the Columbia River,⁵ but Fischer⁶ showed that secondary currents could increase the value of β by as much as a factor of 10. A value of $\beta = 0.6$ has been observed in the Missouri River.⁷

A model user who is interested in predicting the maximum concentration downstream from the same point release should choose a value of $\beta = 0$. A zero β would result in no diffusion and, as seen from Eq. (4.1), would simplify the prediction of activity concentration to a function of decay and downstream distance. This would effectively eliminate the consideration of transverse (or lateral) distance and maximize the activity concentration at any point in midstream.

A model user who is not interested in a conservative estimate will be faced with deciding which value of β to use. From the review of Yotsukura and Cobb² we conclude that β may range from approximately 0.1 to 0.72 (and perhaps higher). We would suggest, then, that a user interested in accurate predictions of pollutant concentrations in water should use a value of β greater than 0.1 and less than 1.0.

4.2 Models with Sorption

If the pollutant being assessed is not in a soluble form or has a marked tendency to adsorb to sediments, as do many actinides, the user may wish to use a model which is capable of predicting sediment or particulate contaminant transport. For such an application, we suggest considering usage of the FETRA model discussed by Onishi and Wise⁸ in spite of the large number of input parameters it requires. FETRA and a similar model, SERATRA,⁹ both estimate sediment and pollutant concentrations downstream from some release site as a function of time after the release. However, SERATRA estimates concentrations longitudinally and vertically, while FETRA considers longitudinal and lateral (transverse) distribution of the pollutant. These models are both much more complex than the Yotsukura model mentioned above, and therefore, SERATRA and FETRA will be discussed in general terms, neglecting specific parameters.

Both SERATRA and FETRA are time-dependent models which consist of three submodels: (1) a sediment transport model, (2) a dissolved contaminant transport model, and (3) a particulate contaminant transport model.

The sediment transport sections of FETRA and SERATRA are capable of accounting for effects of: (1) convection and dispersion, (2) fall

velocity and cohesiveness, (3) deposition on the river bottom, (4) river bed erosion, and (5) tributary flow. Necessary data include: (1) concentration of each of j sediment types per unit volume of water; (2) time after release; (3) longitudinal, lateral, and vertical velocities; (4) particle fall velocity of j sediment types; (5) longitudinal, lateral, and vertical diffusion coefficients of j sediment types; (6) river width; (7) flow depth; (8) sediment deposition and erosion rates for j sediment types; and (9) estimates of several coefficients dealing with erodibility, critical shear stress, and impaction probability on the river bottom for each of j sediment types.

Basically, the same data are required for the particulate contaminant transport submodel. However, this submodel also takes into account the adsorption and desorption of a dissolved pollutant by sediments and removal of the pollutant by chemical or biological means.

The dissolved pollutant submodels of FETRA and SERATRA also account for convection and dispersion of pollutant, adsorption and desorption of pollutant from water, chemical and biological decay of pollutant, and the effects of tributaries. In addition to data required for the two previous submodels, the dissolved contaminant model needs information about the following: (1) distribution coefficient between dissolved and particulate contaminant adsorbed on each of the j sediment types; (2) amount of particulate pollutant per unit mass of j sediments; (3) mass of contaminant dissolved per unit volume of water; (4) longitudinal, lateral, and vertical diffusion coefficients for dissolved contaminant; and (5) chemical and biological decay rate of contaminant.

As the reader has likely noticed, the amount of input data needed for the FETRA and SERATRA models is great. Given the diversity of input

data and the difficulty in measuring some of the input parameters, it is doubtful that either FETRA or SERATRA would be used for a one-time assessment of a pollutant release into a river, such as an accidental release unless prior input data had previously been collected. However, if the model input data were available for a particular site, the models would be well suited to estimate pollutant transfer downstream following some accidental release. We suggest FETRA and SERATRA for those situations in which pollutant sorption to sediments may be important or when the pollutant is a particulate.

REFERENCES

1. Yotsukura, N. and W. W. Sayre. 1976. Transverse mixing in natural channels. *Water Resour. Res.* 12(4):695-704.
2. Yotsukura, N. and E. D. Cobb. 1972. Transverse diffusion of solutes in natural streams. *U.S. Geol. Survey Prof. Paper* 582-C, U.S. Government Printing Office.
3. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.113. *Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix 1.*
4. Elder, J. W. 1959. The dispersion of marked fluid in turbulent shear flow. *J. Fluid Mech.* 5(4):544-60.
5. Glover, R. E. 1964. Dispersion of dissolved or suspended materials in flowing streams. *U.S. Geol. Survey Prof. Paper* 433-B, U.S. Government Printing Office.
6. Fischer, H. B. 1969. The effect of bends on dispersion in streams. *Water Resour. Res.* 5(2):496-506.
7. Yotsukura, N., H. B. Fischer, and W. W. Sayre. 1970. Measurement of mixing characteristics of the Missouri River between Sioux City, Iowa and Plattsmouth, Nebraska, *U.S. Geol. Survey Prof. Paper* 4899-G, U.S. Government Printing Office.
8. Onishi, Y., and S. E. Wise. 1978. *Mechanistic Model of Transport of Sediment and Bacteria in the Lower Missouri River.* PNL-2731.
9. Onishi, Y. 1977. *Batch Transport Model for Sediment and Bacteria Transport in Turbulent Water—Evaluation of the Model and Application to the Upper Missouri.* ONWL-2227.

10. Onishi, Y. 1977. *Mathematical Simulation of Sediment and Radionuclide Transport in the Columbia River.* BNWL-2228.

5. AQUATIC FOOD CHAIN TRANSPORT

Roberta W. Skor

5.1 The Basic Model

Models that describe the transport of radionuclides from liquid effluents to man via aquatic foods generally take the form¹⁻³

$$R_{aijp} = U_{ap} C_i B_{ip} D_{iaj}$$

where

R_{aijp} = dose to organ j of an individual of age group a due to nuclide i via aquatic food p (millirem/year),

U_{ap} = ingestion rate of aquatic food p by an individual of age a (kg/year),

C_i = concentration of radionuclide i in water (pCi/liter),

B_{ip} = the equilibrium ratio of the concentration of radionuclide i in aquatic food p to its concentration in water (liter/kg),

D_{aipj} = dose conversion factor for age group a , radionuclide i , aquatic food p , and organ j (millirem/pCi).

Methods of calculating C_i are considered in Sect. 4 of this report;

U_{ap} is considered in Sect. 6, and D_{aipj} is considered in Sect. 7. The bioaccumulation factor B_{ip} is considered in the remainder of this section.

5.2 The Bioaccumulation Factor

The transfer of radionuclides from contaminated water to human food through various trophic levels of aquatic life is represented by a single parameter, the bioaccumulation factor, B_{ip} . Its value is usually tabulated by nuclide for several trophic levels in freshwater and seawater.⁴⁻⁷

Reported values for B_{ip} vary commonly by an order of magnitude—in some cases by three and four orders of magnitude. Examples of the larger range of values are cesium in freshwater fish and manganese in both freshwater and marine molluscs.⁵⁻⁷

The accumulation of radionuclides in aquatic food occurs by complex interactions of biological, chemical and physical factors including some that can confound the assumption of equilibrium in the definition of B_{ip} (see 5.1). In addition, although nuclides concentrate in different tissues, the tabulations do not always designate the tissue. When different B_{ip} 's of an organism are known, the values given here refer to edible parts.⁵ Stable elements and their congeners markedly affect the accumulation of radionuclide in biota. Extensive studies of ^{89}Sr , ^{90}Sr (and ^{137}Cs) have shown their ecological accumulation to be usually inverse functions of congeners stable element concentration, of calcium (and potassium) in both aquatic and terrestrial biota.⁸⁻¹¹

The B_{ip} 's for Cs and Sr in freshwater fish have been correlated from data from a number of studies in natural waters by Vanderploeg et al.⁷ B_{ip} values may be estimated from site-specific concentrations of potassium and calcium in water according to the following:

1. <50 ppm sediment in unfiltered water
 - piscivorous fish, $B_{ip}(\text{Cs}) = 1.5 \times 10^4 / [\text{K}]_w$
 - nonpiscivorous fish, $B_{ip}(\text{Cs}) = 5 \times 10^3 / [\text{K}]_w$
2. 50 ppm sediment in unfiltered water
 - piscivorous fish, $B_{ip}(\text{Cs}) = 3 \times 10^4 / [\text{K}]_w$
 - nonpiscivorous fish, $B_{ip}(\text{Cs}) = 1 \times 10^3 / [\text{K}]_w$

3. fish muscle

$$B_{ip}(Sr) = \exp \{5.18 \pm 1.11 \text{ (s.e.)} - 1.21 \pm 0.37 \text{ (s.e.)} \ln [Ca]_w\} ,$$

where

$[K]_w$ = equilibrium concentration of K in the water (ppm),

$[Ca]_w$ = equilibrium concentration of Ca in the water (ppm), and

s.e. = standard error of the mean.

Site-specific information should be used in estimating bioaccumulation factors, especially for cesium in freshwater environments, if precision of one order of magnitude is desired.

In the absence of site-specific information, values of B_{ip} for different nuclides have been tabulated. Table 5.1 presents the results of an analysis¹² of B_{ip} values for strontium, iodine, and cesium in freshwater finfish taken from Vanderploeg et al.⁷ Values of B_{ip} currently used at Oak Ridge National Laboratory¹³ for both freshwater and marine species are shown in Table 5.2. The tabulation consists of selected values from Thompson et al.⁵ Most of these values have been calculated from separate analyses of the stable elements in biota and water that may not have originated in the same part of the world. Although these values are reasonable as shown by comparison with experiments in which equilibrium between water and biota was attempted or attained, site specific values may possibly exhibit wide variations from them.

Accidental releases have not been judged important in aqueous food pathway assessments because of the availability of time for counter measures to be taken. A study was made of the accidental consequences of a nuclear submarine collision with a surface vessel in a harbor. The

authors conclude that a single meal of fish or shell fish consumed before the food supply was monitored would result in exposure within the range of "acceptable" dose levels.¹⁴

Table 5.1 Results of a statistical analysis of B_{ip} (liters/kg) for Sr, I, and Cs in freshwater finfish^a

Element	μ	σ	$\bar{X}$
Sr	2.4	1.8	60
I	3.5	0.61	41
Cs	7.2	0.86	1900

^aHoffman, F. O. (ref. 12).

Table 5.2 Values of B_{ip} (liter/kg) for various elements in aquatic foods^a

Element	Freshwater		Marine	
	Invertebrates	Fish	Invertebrates	Fish
Na	1.7E + 01	2.0E + 01	1.0E 00	1.0E 00
P	1.0E + 05	1.0E + 05	1.0E + 04	1.0E + 04
S	1.0E + 02	7.5E + 02	4.0E 00	4.0E 00
Mn ^b	1.0E + 05	3.2E - 01 ^c	5.0E + 04	3.0E + 03
Fe	3.2E + 03	1.0E + 02	2.0E + 04	1.0E + 03
Co ^b	1.0E 04	3.0E 02	1.0E + 04	1.0E + 02
Ni	1.0E + 02	1.0E + 02	1.0E + 02	5.0E + 02
Zn	1.0E + 04	1.0E + 03	5.0E + 04	5.0E + 03
Sr ^b	3.0E + 02		1.0E 00	1.0E 00
Y	1.0E + 03	2.5E + 01	1.0E + 02	3.0E + 01
Zr	6.7E 00	3.3E 00	1.0E + 02	3.0E + 01
Nb	1.0E + 02	3.0E + 04	2.0E + 02	1.0E + 02
Tc	5.0E 00	1.5E + 01	1.0E + 02	1.0E + 01
Ru	3.0E + 02	1.0E + 01	1.0E + 02	3.0E 00
Ag	7.7E + 02	2.3E 00	5.0E + 03	1.0E + 03
Sb	1.0E + 01	1.0E 00	1.0E + 03	1.0E + 03
Te	6.1E + 03	4.0E + 02	1.0E + 02	1.0E + 01
I ^b	4.0E + 02		1.0E + 02	2.0E + 01
Cs ^b	1.0E + 03		5.0E + 01	3.0E + 01
Ce	1.0E + 03	2.5E + 01	1.0E + 02	3.0E + 01
Pm	1.0E + 03	2.5E + 01	1.0E + 03	1.0E + 02
Pb	1.0E + 02	3.0E + 02	1.0E + 03	3.0E + 02
Bi	1.0E + 05	1.5E + 01	1.0E + 05	1.5E + 01
Po	2.0E + 04	5.0E + 01	2.0E + 04	2.0E + 03
Ra	2.5E + 02	5.0E + 01	1.0E + 02	5.0E + 01
Th	5.0E + 02	3.0E + 01	2.0E + 03	1.0E + 04
Pa	1.1E + 02	1.1E + 01	1.0E + 01	1.0E + 01
U	1.0E + 02	1.0E + 01	1.0E + 01	1.0E + 01
Np	4.0E + 02	1.0E + 01	1.0E + 01	1.0E + 01
Pu	1.0E + 02	3.5E + 02	1.0E + 02	3.5E 00
Am	1.0E + 03	2.5E + 01	1.0E + 03	2.5E + 01
Cm	1.0E + 03	2.5E + 01	1.0E + 03	2.5E + 01

^aValues taken from ref. 7 unless otherwise indicated.

^bValues taken from ref. 5 for freshwater; fish muscle.

^cDivided by $[Mn]_w$ water concentration of Mn (ppm).

REFERENCES

1. Soldat, J. K., D. A. Baker, and J. P. Corley. 1973. Applications of a general computational model for composite environmental radiation doses. pp. 483-98. IN *Environmental Behavior of Radionuclides Released in the Nuclear Industry, Symposium Proceedings*. International Atomic Energy Agency, Vienna, Austria.
2. Lyon, R. B. 1976. *RAMM: A System of Computer Programs for Radionuclide Pathway Analysis Calculations*. Atomic Energy of Canada, Limited, Publication AECL-5527.
3. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.109. *Calculation of Annual Doses to Man From Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I. Revision 1*.
4. Polikasov, G. G. 1973. "Radioecology of Marine Plants and Animals," *Radioecology*, Eds. V. M. Klechkostii, G. G. Polikasov, and R. M. Aleksakhin, J. Wiley & Sons, New York.
5. Thomas, S. E., C. A. Burton, D. J. Quinn, Y. C. Ng. 1972. *Concentration Factors of Chemical Elements in Edible Aquatic Organisms*. UCRL-50564 Rev. Lawrence Livermore Laboratory, Livermore, California.
6. Jinks, S. M. and M. Eisenbud. 1972. "Concentration Factors in the Aqueous Environment," *Rad. Data Rep.* 13 243-247.
7. Vanderploeg, H. A., D. C. Parzyck, W. H. Wilcox, J. R. Kercher, and S. V. Kaye. 1975. *Bioaccumulation Factors for Radionuclides in Freshwater Biota*. ORNL-5002.

8. Templeton, W. L. and V. B. Bowen. 1964. "The Relationship Between Concentrations of Calcium, Strontium and Strontium-90 in Wild Brown Trout *SALMOTRUTTA* and the Concentrations of Stable Elements in Some Waters of the United Kingdom and the Implications in Radio-ecological Health Studies," *Int. J. Air Wat Poll.* 8 49-75.
9. Ettenhuber, E. and W. Röhsch. 1975. "The Fish/Water Accumulation Factor—An Important Parameter for Determining the Environmental Capacity of Surface Water," in *Radiological Impacts of Releases from Nuclear Facilities into Aquatic Environments*, STI/PUB/406 IAEA, Vienna.
10. Fleishman, D. G. 1973. "Accumulation of Artificial Radionuclide in Freshwater Fish," in *Radioecology* 347-370.
11. Russell, R. S. 1966. *Radioactivity and Human Diet*, Chaps. 2, 9, 11, 15, 18, Pergamon Press, New York.
12. Hoffman, F. O. 1979. Bioaccumulation factors for freshwater fish. IN Hoffman, F. O., and C. F. Baes III (eds.), *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. ORNL/NUREG/TM-282.
13. Killough, G. G., and L. R. McKay. 1976. *A Methodology for Calculating Radiation Doses from Radioactivity Released to the Environment*. ORNL-4992.
14. Preston, A and D. F. Jeffries. "An Evaluation of the Public Health Aspects of a Marine Accident in Coastal Waters," in *Proceedings of the Fourth International Symposium on the Transport of Hazardous Cargoes by Sea and Inland Waterway*, U566-D-24-76 (also AD-A-023 505) pp. 325-338.

6. ADULT DIETARY INTAKE AND INHALATION RATES

Elizabeth M. Rupp

The internal radiation dose to man as a result of exposure to contaminated air, water, and/or food is determined in part by the amount of material which is incorporated into the human body via food and inhalation pathways.

Dietary parameters and inhalation rates considered in this study are as follows:

1. the consumption of milk, U_{ap}^M (liter/year);
2. the consumption of water, U_{ap}^W (liter/year);
3. the consumption of fruits, leafy vegetables, below-ground vegetables, other vegetables, grain, beef, other meats and poultry, finfish, and shellfish, U_{ap}^F (kg/year);
4. the volumetric inhalation rate, U_{ap}^A (m^3 /year).

Table 6.1 presents the annual average values of ingestion and inhalation rates for adult individuals. While these factors are undoubtedly related to age and individual variability, only adult values are given in Table 6.1 because the internal dose conversion factors provided in Sect. 7 are only for adult individuals. A more detailed analysis of human dietary intake and inhalation rates is given elsewhere.¹

Table 6.1. Estimated values of ingestion and inhalation rates for adult individuals

Item	Average value	Reference
	--kg/year--	
Fruits	68	2
Leafy vegetables	18	3
Below ground vegetables	28	2
Other vegetables	45	2
Grains (flour equivalent)	35	2
Beef	32	2
Other meats, poultry	63	2
Finfish	4.4	8, 9
Shellfish	1.3	8, 9
	--liters/year--	
Milk ^a	112	2
Tap water	93	10
Other beverages	400	10
	--m ³ /year--	
Inhalation rate	8030	11

^aInclude all milk drinks, fresh cream, ice cream, and small amounts of cheese.

REFERENCES

1. Rupp, E. M. 1979. Annual dietary intake and respiration rates.
IN Hoffman, F. O., and C. F. Baes III (eds.), *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. ORNL/NUREG/TM-282.
2. U.S. Department of Agriculture. 1965. Food and nutrient intake of individuals in the United States, Spring 1965. *Household Consumption Survey 1965-66*. Report No. 11.
3. Blanchard, R. L. *Criteria for the Estimation of Radiation Dose*. Unpublished. Data averaged by Blanchard from references 4-7, this paper.
4. Soldat, J. K. 1976. Radiation doses from iodine-129 in the environment. *Health Phys.* 30:61.
5. Essig, T. H., and J. P. Corley. 1969. *Criteria Used to Estimate Doses Received by Persons Living in the Vicinity of Hanford: Interim Report No. 2*. BNWL-1019.
6. Soldat, J. K. 1976. Reply to J. C. Thompson, Jr.'s comments on the paper, Radiation doses from iodine-129 in the environment, by J. K. Soldat. *Health Phys.* 31:288.
7. U.S. Department of Agriculture. 1977. *Food Consumption, Prices and Expenditures*. Supplement for 1975 to Agriculture Economic Report AER-138.

8. U.S. National Marine Fisheries Service. 1973-74. Unpublished data.
9. Rupp, E. M., F. L. Miller, and C. F. Baes III. 1979. Some results by age and region of recent surveys of fish and shellfish consumption by U. S. residents. In preparation.
10. Cook, C. B., D. A. Eiler, and O. D. Forker. 1975. Beverage consumption patterns in New York state. *Am. Diet. Assoc.* 67(3):222-27.
11. International Commission on Radiological Protection. 1974. *Report of the Task Group on Reference Man*. ICRP Publications 23, Pergamon Press, New York.

7. DOSE CONVERSION FACTORS

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7.1 Introduction

The radiological impact to man resulting from operation of a nuclear-related facility may be assessed by calculating the dose to individuals and populations residing near the facility. There are a number of potential modes of exposure to man from radioactive effluents released to the environment by breeder reactor facilities. Modes contributing to the vast majority of human exposure are inhalation, ingestion, immersion in contaminated air, and exposure to contaminated ground surfaces. Dose conversion factors are used to estimate the dose resulting from these exposures.

7.2 Internal Dose Conversion Factors

Table 7.1 lists recommended dose conversion factors for inhalation and ingestion of certain radionuclides determined to be of potential significance in breeder reactor fuel cycles. These factors are taken from recently prepared documents by Killough et al.¹ and by Dunning et al.,² and were calculated by the use of dosimetric criteria provided by the International Commission on Radiological Protection³ and other recognized authorities. These factors are computed with the computer code INREM II, which approximates the solutions of differential equations which model intake, translocation and metabolism of a radionuclide and its progeny.

Table 7.1 Fifty-year dose commitments for inhalation or ingestion of certain radionuclides (rem/ μ Ci intake)

	Pathway	Solubility class	Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)
^3H	Inhalation Ingestion		1.25E-4 8.36E-5	1.25E-4 8.30E-5	1.24E-4 8.29E-5	9.85E-5 6.56E-5	1.25E-4 8.30E-5	0.95 0.95
^{14}C	Inhalation Ingestion		6.18E-6 8.94E-4	1.41E-5 1.92E-3	5.29E-6 7.36E-4	5.08E-5 7.05E-3	5.42E-6 7.23E-4	0.95 0.95
^{22}Na	Inhalation Ingestion		9.43E-3 9.13E-3	9.16E-3 1.30E-2	1.02E-2 1.57E-2	1.60E-2 2.41E-2	7.46E-3 1.13E-2	0.95 0.95
^{54}Mn	Inhalation Ingestion	D W	4.18E-3 2.47E-2 8.03E-4	4.10E-3 3.66E-3 1.23E-3	3.79E-3 2.81E-3 3.61E-3	5.86E-3 3.44E-3 1.37E-3	2.27E-3 9.07E-4 7.10E-4	0.1 0.1 0.1
^{55}Fe	Inhalation Ingestion	W Y	3.20E-3 2.51E-2 1.23E-5	6.29E-4 6.09E-4 3.50E-4	1.11E-3 5.16E-4 6.59E-4	7.40E-4 3.43E-4 4.37E-4	1.05E-3 4.86E-4 6.20E-4	0.1 0.1 0.1
^{58}Co	Inhalation Ingestion	Y W	5.92E-2 2.96E-2 2.64E-4	4.17E-3 2.65E-3 7.65E-4	2.36E-3 2.60E-3 3.92E-3	2.58E-3 1.78E-3 4.58E-4	3.64E-4 6.89E-4 5.39E-4	0.05 0.05 0.05
^{60}Co	Inhalation Ingestion	W Y	0.132 1.27 2.68E-3	1.71E-2 8.20E-2 4.37E-3	1.70E-2 1.85E-2 1.24E-2	1.47E-2 5.08E-2 3.99E-3	9.51E-3 6.12E-3 3.82E-3	0.05 0.05 0.05
^{65}Zn	Inhalation Ingestion	D W Y	1.18E-2 2.54E-2 7.65E-2 1.02E-2	1.40E-2 1.05E-2 1.24E-2 1.26E-2	1.74E-2 1.18E-2 9.50E-3 1.65E-2	1.77E-2 1.23E-2 1.21E-2 1.58E-2	1.28E-2 8.30E-3 6.31E-3 1.15E-2	0.5 0.5 0.5 0.5
^{89}Sr	Inhalation	D Y	6.55E-3 0.313	3.89E-3 4.76E-3	2.18E-3 4.35E-5	2.74E-2 5.44E-4	2.18E-3 4.28E-5	0.2 0.01

Table 7.1 (continued)

	Pathway	Solubility class	Lungs	Total body	Ovaries	Total endosteal cells	Testes	Uptake Fraction (SI:B)
⁸⁹ Sr	Ingestion		9.64E-6 1.58E-8	1.80E-3 4.87E-4	8.56E-4 4.33E-5	1.08E-2 5.38E-4	8.55E-4 4.28E-5	0.2 0.01
⁹⁰ Sr	Inhalation	D	9.89E-3	0.241	1.53E-2	2.20	1.53E-2	0.2
		Y	8.49	0.152	3.65E-4	0.233	3.65E-3	0.01
	Ingestion		5.94E-9 2.97E-10	9.45E-2 5.02E-3	5.99E-3 3.00E-4	0.859 4.30E-2	5.99E-3 3.00E-4	0.2 0.01
⁹⁰ Y	Inhalation		3.93E-2	8.90E-4	9.07E-5	9.32E-4	9.07E-5	1.0E-4
	Ingestion		9.80E-9	5.07E-4	1.25E-7	1.22E-6	1.18E-7	1.0E-4
⁹¹ Y	Inhalation	W	0.198	5.67E-3	7.28E-4	1.79E-2	7.21E-4	1.0E-4
	Ingestion		3.30E-7	4.37E-4	1.34E-5	1.97E-5	1.86E-6	1.0E-4
⁹⁵ Zr	Inhalation	W	6.86E-2	4.95E-3	3.14E-3	8.99E-3	1.18E-3	2.0E-3
		Y	0.131	5.56E-3	2.31E-3	3.24E-3	3.29E-4	2.0E-3
	Ingestion		8.29E-5	5.49E-4	3.02E-3	3.55E-4	2.99E-4	2.0E-3
⁹⁵ Nb	Inhalation		3.10E-2	1.94E-3	1.64E-3	1.19E-3	2.46E-4	1.0E-2
	Ingestion		9.81E-5	5.04E-4	2.99E-3	3.19E-4	3.58E-4	1.0E-2
⁹⁹ Tc	Inhalation	D	9.63E-4	1.69E-4	2.41E-4	3.11E-4	2.41E-4	0.80
		W	5.22E-2	8.87E-4	2.12E-4	2.75E-5	2.12E-4	0.80
	Ingestion		0.0	2.15E-4	3.17E-4	4.10E-4	3.17E-4	0.80
¹⁰³ Ru	Inhalation	Y	5.84E-2	1.98E-3	1.20E-3	9.25E-4	2.55E-4	4.0E-2
	Ingestion		1.36E-4	5.29E-4	2.17E-3	3.96E-4	4.27E-4	4.0E-2
¹⁰⁶ Ru	Inhalation	Y	3.85	6.18E-2	7.67E-3	1.00E-2	6.97E-3	4.0E-2
	Ingestion		2.17E-4	5.94E-3	8.96E-3	9.58E-3	8.14E-3	4.0E-2

Table 7.1 (continued)

	Pathway	Solubility class	Target					Uptake Fraction (SI:B)
			Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{127}Te	Inhalation	W	2.48E-3	5.13E-5	5.29E-3	3.06E-5	5.22E-5	0.25
^{127m}Te	Inhalation	W	0.118	2.42E-3	2.53E-3	2.91E-3	2.48E-3	0.25
			2.32E-5	7.60E-4	2.64E-3	3.08E-3	2.55E-3	0.25
^{129m}Te	Inhalation	W	0.153	3.20E-3	4.15E-3	3.18E-3	4.02E-3	0.25
			7.19E-5	1.29E-3	5.23E-3	3.83E-3	4.96E-3	0.25
^{129}I	Inhalation	D	7.88E-4	2.05E-3	3.78E-4	5.65E-4	3.57E-4	0.95
	Ingestion	D		Thyroid: 4.97				
			1.80E-4	3.19E-3	5.92E-4	8.79E-4	5.58E-4	0.95
^{131}I	Inhalation	D		Thyroid: 7.78				
			2.40E-3	6.13E-4	3.98E-5	2.21E-4	2.22E-5	0.95
				Thyroid: 1.13				
^{131}I	Ingestion	D	2.86E-4	9.08E-4	6.11E-5	3.32E-4	3.53E-5	
				Thyroid: 1.81				
^{134}Cs	Inhalation	D	3.38E-2	4.55E-2	6.46E-2	5.89E-2	5.13E-2	0.95
			4.68E-2	6.84E-2	9.74E-2	8.86E-2	7.73E-2	0.95
^{137}Cs	Inhalation	D	1.62E-2	3.26E-2	5.00E-2	5.31E-2	4.44E-2	0.95
			1.99E-2	4.91E-2	7.54E-2	7.99E-2	6.66E-2	0.95
^{144}Ce	Inhalation	Y	2.92	4.68E-2	6.29E-4	1.47E-2	4.58E-4	1.0E-4
		W	0.678	5.11E-2	4.65E-3	1.14E-2	4.40E-3	1.0E-4
		D	1.73E-2	0.175	1.94E-2	1.61E-2	1.88E-2	1.0E-4
	Ingestion	D	4.48E-6	1.01E-3	2.42E-4	4.47E-4	2.29E-5	1.0E-4
^{147}Pm	Inhalation	Y	0.287	5.30E-3	1.24E-4	4.33E-3	1.23E-4	1.0E-4
		W	3.63E-2	6.58E-3	6.34E-4	2.23E-2	6.34E-4	1.0E-4
		D	7.06E-4	2.46E-2	2.58E-3	9.07E-2	2.58E-3	1.0E-4
	Ingestion	D	4.37E-10	5.02E-5	5.55E-7	1.89E-5	5.39E-7	1.0E-4

Table 7.1 (continued)

Isotope	Pathway	Solubility class	Lungs	Target			Uptake Fraction (SI:B)
				Total body	Ovaries	Testes	
^{210}Pb	Inhalation Ingestion	W	6.18 6.92E-5	15.2 7.44	0.669 0.302	0.669 0.302	0.08 0.08
^{211}Pb	Inhalation Ingestion	W	0.140 6.75E-7	2.20E-3 5.77E-5	3.12E-5 8.42E-6	3.12E-5 5.42E-6	0.08 0.08
^{212}Pb	Inhalation Ingestion	W	1.79 5.33E-5	3.65E-2 7.75E-3	1.97E-3 1.87E-3	1.71E-3 6.32E-4	0.08 0.08
^{214}Pb	Inhalation Ingestion	W	0.116 7.59E-6	1.56E-3 8.14E-5	1.09E-4 5.11E-5	1.08E-4 1.32E-5	0.08 0.08
^{210}Bi	Inhalation Ingestion	W	1.02 5.20E-11	2.79E-2 3.04E-3	8.68E-3 2.32E-3	8.68E-3 1.25E-3	0.05 0.05
^{212}Bi	Inhalation Ingestion	W	0.271 1.29E-5	4.32E-3 1.39E-4	3.17E-4 1.15E-4	3.15E-4 5.32E-5	0.05 0.05
^{213}Bi	Inhalation Ingestion	W	9.77E-3 1.79E-5	1.68E-4 5.27E-5	1.62E-5 7.68E-5	1.26E-5 1.14E-5	0.05 0.05
^{214}Bi	Inhalation Ingestion	W	9.39E-2 6.36E-6	1.49E-3 4.00E-5	9.13E-5 2.12E-5	9.06E-5 5.82E-6	0.05 0.05
^{210}Po	Inhalation Ingestion	W D	45.8 1.03 1.14E-8	1.56 2.80 0.562	0.803 2.58 0.521	0.803 2.58 0.521	0.1 0.1 0.1
^{220}Rn	Inhalation Ingestion	gas	2.16E-3 8.47E-8	3.22E-5 2.22E-5	1.21E-7 3.12E-6	9.60E-8 1.31E-6	0 0
^{222}Rn	Inhalation Ingestion	gas	2.34E-3 1.15E-4	3.46E-5 4.59E-3	2.72E-7 5.40E-3	2.66E-7 1.06E-4	0 0
^{223}Ra	Inhalation Ingestion	W	46.4 5.14E-5	1.12 0.561	0.221 0.272	0.221 0.272	0.2 0.2
^{224}Ra	Inhalation Ingestion	W	8.75 9.21E-5	0.267 0.189	6.11E-2 9.27E-2	6.03E-2 8.05E-2	0.2 0.2

Table 7.1 (continued)

Pathway	Solubility class	Target					Uptake Fraction (SI:B)
		Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{225}Ra	W	28.8 3.37E-4	0.942 0.517	0.243 0.260	8.75 8.72	0.226 0.259	0.2 0.2
^{226}Ra	W	56 1.76E-3	19.5 15.8	0.677 0.590	227 202	0.676 0.590	0.2 0.2
^{228}Ra	W	4.81 2.23E-3	11 8.4	1.1 0.778	262 203	1.1 0.778	0.2 0.2
^{225}Ac	Y	49.3 42.5 2.01E-4	0.783 1.54 3.35E-2	4.5E-2 0.25E-2 3.65E-2	1.16 16.9 0.364	3.72E-2 0.25E-2 2.40E-2	1.0E-3 1.0E-3 1.0E-3
^{227}Ac	Y	1030 84.1 3.53E-4	255 581 4.8	35. 93. 0.77	4470 1.22E+4 101	2.40E-2 92. 0.77	1.0E-3 1.0E-3 1.0E-3
^{227}Th	Y	68.7 52.7 1.19E-5	1.18 2.18 2.40E-2	5.93E-2 0.25 6.57E-3	3.14 28.6 0.45	5.90E-2 0.25 6.02E-3	1.0E-3 1.0E-3 1.0E-3
^{228}Th	Y	716 117 6.43E-5	28.6 71.7 0.602	2.2 10. 9.01E-2	2870 1590 13.7	2.1 10 8.95E-2	1.0E-3 1.0E-3 1.0E-3
^{229}Th	Y	1230 116 2.00E-2	387 830 6.82	59. 140. 12	8400 2.06E+4 170	5. 140 1.2	1.0E-3 1.0E-3 1.0E-3
^{230}Th	Y	526 54.2 4.62E-6	155 347 2.85	37 91. 0.75	4280 1.05E+4 87.4	37. 91. 0.75	1.0E-3 1.0E-3 1.0E-3
^{231}Th	Y	4.16E-3 3.91E-3 6.41E-7	1.21E-4 1.77E-4 8.07E-5	3.39E-5 4.96E-5 8.26E-5	7.01E-4 1.98E-3 2.23E-5	7.18E-6 2.69E-5 2.69E-6	1.0E-3 1.0E-3 1.0E-3

Table 7.1 (continued)

Pathway	Solubility class	Target					Uptake Fraction (SI:B)
		Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{232}Th	Inhalation	454	158	32	4870	32	1.0E-3
	Ingestion	46.5 1.04E-4	354 2.90	79 0.65	1.19E+4 98.5	79 0.65	1.0E-3 1.0E-3
^{234}Th	Inhalation	0.243	3.92E-3	1.31E-4	5.18E-4	8.27E-5	1.0E-3
	Ingestion	0.178 1.80E-6	4.70E-3 6.56E-4	1.39E-3 1.15E-4	8.23E-3 1.05E-4	1.34E-3 2.30E-5	1.0E-3 1.0E-3
^{231}Pa	Inhalation	594	413	78	9370	78	1.0E-3
	Ingestion	57.6 4.35E-4	890 7.25	181 1.5	2.17E+4 178	181 1.5	1.0E-3 1.0E-3
^{233}Pa	Inhalation	6.22E-2	1.32E-3	4.89E-4	4.19E-4	7.01E-5	1.0E-3
	Ingestion	4.43E-2 1.45E-5	1.58E-3 2.61E-4	8.53E-4 8.98E-4	2.47E-3 9.41E-5	4.44E-4 7.80E-5	1.0E-3 1.0E-3
^{234}Pa	Inhalation	5.23E-3	2.10E-4	2.44E-4	8.51E-5	1.93E-5	1.0E-3
	Ingestion	4.89E-3 5.62E-5	2.11E-4 3.13E-4	2.15E-4 1.26E-3	1.34E-4 1.03E-4	2.81E-5 9.00E-5	1.0E-3 1.0E-3
^{232}U	Inhalation	887	80	6.4	733	6.4	2.0E-3
	Ingestion	66.5 1.14 9.52E-5 2.38E-3	39 119 0.444 11.1	8.5 27.4 0.112 2.81	649 2030 8.3 208	8.5 27.5 0.113 2.82	5.0E-2 5.0E-2 2.0E-1 5.0E-1
^{233}U	Inhalation	542	19.5	1.17	3.59	1.17	2.0E-3
	Ingestion	55.9 0.950 6.53E-5 1.63E-3	10.9 33.5 0.138 3.43	3.40 11.3 4.64E-2 1.16	10.4 34.8 0.142 3.55	3.40 11.3 4.64E-2 1.16	5.0E-2 5.0E-2 2.0E-3 5.0E-2
^{234}U	Inhalation	536	19.3	1.16	353	1.16	2.0E-3
	Ingestion	55.4 0.941 6.46E-5 1.61E-3	10.6 33.2 0.137 3.40	3.37 11.2 4.59E-2 1.15	10.2 34.0 0.139 3.48	3.37 11.2 4.59E-2 1.15	5.0E-2 5.0E-2 2.0E-3 5.0E-2

Table 7.1 (continued)

	Pathway	Solubility class	Target				Uptake Fraction (SI:B)	
			Lungs	Total body	Ovaries	Total endosteal cells		Testes
²³⁵ U	Inhalation	Y	484	17.4	1.05	3.37	1.06	2.0E-3
		W	49.9	9.77	3.04	8.74	3.03	5.0E-2
	Ingestion	D	0.854	29.9	10.1	29.0	10.1	5.0E-2
²³⁶ U			9.27E-5	0.123	4.24E-2	0.119	4.14E-2	2.0E-3
			2.08E-3	3.07	1.04	2.97	1.03	5.0E-2
	Inhalation	Y	506	18.2	1.09	3.18	1.09	2.0E-3
²³⁸ U		W	52.2	10.2	3.17	9.22	3.17	5.0E-2
		D	0.887	31.3	10.5	30.7	10.5	5.0E-2
	Ingestion		6.09E-5	0.129	4.32E-2	0.126	4.32E-2	2.0E-3
²³⁹ Pu			1.52E-3	3.20	1.08	3.14	1.08	5.0E-2
	Inhalation	Y	480	17.1	1.03	2.89	1.04	2.0E-3
		W	49.0	9.57	2.99	8.36	3.01	5.0E-2
²⁴⁰ Pu		D	0.834	29.3	9.96	27.8	10.0	5.0E-2
	Ingestion		6.01E-5	0.121	4.08E-2	0.114	4.10E-2	2.0E-3
			1.50E-3	3.00	1.02	2.85	1.02	5.0E-2
²³⁷ Np	Inhalation	Y	569	167	40	4240	40.	1.0E-3
		W	58.6	374	38.	1.05E-4	99.	1.0E-3
	Ingestion		8.73E-4	3.08	0.81	86.8	0.81	1.0E-3
²³⁸ Pu	Inhalation	Y	608	149	35.	3270	35.	3.0E-5
		W	63.6	340	89.	8400	89.	3.0E-5
	Ingestion		1.14E-7	8.51E-2	2.21E-2	2.09	2.21E-2	3.0E-5
²³⁹ Pu	Inhalation	Y	580	169	40.	4160	40.	3.0E-5
		W	59.8	381	100.	1.04E+4	100.	3.0E-5
	Ingestion		9.35E-8	9.51E-2	2.48E-2	2.58	2.47E-2	3.0E-5
²⁴⁰ Pu	Inhalation	Y	579	169	40.	4150	40	3.0E-5
		W	59.8	380	100.	1.04E+4	100.	3.0E-5
	Ingestion		1.16E-7	9.49E-2	2.48E-2	2.58	2.47E-2	3.0E-5

Table 7.1 (continued)

	Pathway	Solubility class	Target					Uptake Fraction (SI:B)
			Lungs	Total body	Ovaries	Total endosteal cells	Testes	
^{241}Pu	Inhalation	Y	1.10	3.52	9.00E-1	85.4	0.90	3.0E-5
		W	1.54E-2	7.75	2.0	194	2.1	3.0E-5
	Ingestion		7.18E-8	1.93E-3	5.10E-4	4.83E-2	5.11E-4	3.0E-5
^{242}Pu	Inhalation	Y	550	161	38	4190	38.	3.0E-5
		W	56.8	361	95.	1.05E+4	95	3.0E-5
	Ingestion		1.91E-7	9.02E-2	2.36E-2	2.60	2.35E-2	3.0E-5
^{241}Am	Inhalation	Y	615	175	41	3900	41.	1.0E-3
		W	63.6	392	103.	9730	103.	1.0E-3
	Ingestion		1.28E-4	3.22	0.85	80.2	0.85	1.0E-3
^{243}Am	Inhalation	Y	595	175	41.	4170	42.	1.0E-3
		W	61.4	391	103.	1.03E+4	103.	1.0E-3
	Ingestion		9.56E-4	3.21	0.85	85.2	0.85	1.0E-3
^{242}Cm	Inhalation	Y	172	3.67	2.70E-1	22.5	0.26	1.0E-3
		W	54.8	8.76	2.0	168	2.0	1.0E-3
	Ingestion		2.65E-6	7.34E-2	1.85E-2	1.53	1.85E-2	1.0E-3
^{243}Cm	Inhalation	Y	627	113	25.	2150	25.	1.0E-3
		W	67.5	261	68.	5780	68.	1.0E-3
	Ingestion		3.26E-4	2.15	0.17	47.8	0.56	1.0E-3
^{244}Cm	Inhalation	Y	607	86.5	19.	1620	19.	1.0E-3
		W	66.8	204	53.	4560	53.	1.0E-3
	Ingestion		3.54E-6	1.68	0.44	37.7	0.43	1.0E-3

D = daily, W = weekly, Y = yearly (ref. 3).

SI:B = Gastrointestinal uptake fraction; small intestine: blood (refs. 1,2).

Activity median aerodynamic diameter (AMAD) = 1 μm (ref. 1).

The INREM II computer code utilizes an adaptation of the Task Group Lung Model³ to describe radionuclide deposition and retention dynamics within the respiratory tract. A catenary gastrointestinal (GI) tract model based on the transit times recommended by Eve⁴ simulates retention in the GI tract. Retention in other organs is represented by multicompartment models consisting of series of decaying exponential terms. Detailed discussion of these models and assumptions is available in references 1 and 2.

In the context of radionuclides released during routine breeder reactor fuel cycle operations, it is useful to consider the calculation of 50-year dose commitments in terms of three radionuclide classes, based on effective half-life of radionuclides in the human body. Depending on the organ under consideration, one of several approaches will apply.

For radionuclides that have short biological half-times of residence or short radiological half-lives, the 50-year dose commitment would be approximately the same as the annual dose. For example, all of the dose commitment for ²²⁰Rn exposure (including radon daughters) is received during the first year.

For radionuclides of intermediate longevity in humans, dose would be accrued at a declining rate over the entire 50-year period, based on models and associated parameters specific to the organ and nuclide of concern. For example, ²³²U (physical $T_{1/2} = 71.7$ years) is retained in bone according to the uranium retention function:

$$R_{(\text{bone})}(t) = 0.90 e^{-0.693 t/20} + 0.10 e^{-0.693 t/5000},$$

where t is the time of residence in days. This expression indicates that the initial component of retained uranium is removed rapidly in biological processes with a half-time of 20 days. The second component in the expression contributes the majority of the total 50-year dose commitment from ^{232}U , at a rate which decreases with a half-time of 13.7 years.

For elements which are eliminated very slowly from the body, such as plutonium, and which possess very long physical half-lives [$T_{1/2}$ (^{239}Pu) = 2.4×10^4 years)], an individual will continue to accrue dose after intake of the radioisotope at a relatively constant rate for the entire 50-year period of interest. Under such conditions, the approximate dose received during the year after the radionuclide enters the body is obtained by dividing the dose commitment by 50. Thus, the approximate average annual dose rate is only 1/50 of the dose commitment. If an individual is exposed to effluents for a 20-year operating life of the plant, his annual dose rate during the twentieth year is about 20 times the annual dose rate from one year of exposure.

These generalized observations are approximately correct for the conditions cited. However, a detailed calculation must be made to determine the actual dose received in a given year.

Table 7.1 lists 50-year dose commitments for inhalation and ingestion of those radionuclides of interest in this study. These dose equivalents take into account the contributions of radioactive daughters formed by decay of a parent within the body. Dissimilar migrations of daughter elements are also considered. Where a choice of solubility class exists,

based on a choice of the chemical form of a particular released radionuclide, dose conversion factors for all solubility classes applicable are listed. The reader is referred to Killough et al.,¹ Dunning et al.,² or Morrow et al.³ for detailed discussions of this subject. A quality factor of 20 is used for alpha particles, as suggested by the ICRP.⁵

While dose conversion factors have been calculated elsewhere for 22 target organs,^{1,2} the purposes of this report may be best served by focusing on five organs of major interest from the standpoint of health risk assessment. These organs are lungs, total body, ovaries, testes, and total endosteal cells (sensitive cell layer in bone). Dose conversion factors for lung are calculated as dose to the entire lung; this is commonly identified as the "smeared lung" dose. Gastrointestinal uptake fractions (small intestine: blood) are listed for radionuclides in the final column of Table 7.1. The reader is referred to Killough et al.¹ and Dunning et al.² for detailed discussion of this subject.

7.3 External Dose Conversion Factors

Tables 7.2 and 7.3 list external dose conversion factors for the radionuclides of interest as discussed above. Dose conversion factors for immersion in contaminated air are listed for: (1) β dose rate at the skin surface of an exposed body (body surface), and (2) photon (γ and X-ray) dose rate to various body organs (lungs, ovaries, skeleton, testes, and total body). Similarly, dose conversion factors for exposure to radionuclides deposited on contaminated ground surfaces are calculated for body surface and for the five organs, as above. These dose conversion factors are from the report by Kocher.⁶

Table 7.2 External dose conversion factors for immersion in contaminated air (millirem/yr/ $\mu\text{Ci}/\text{cm}^3$)

	Body surface (8 dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton	Testes	Total body
^3H	5.97E7	0.0	0.0	0.0	0.0	0.0
^{14}C	4.46E8	0.0	0.0	0.0	0.0	0.0
^{22}Na	1.74E9	1.17E10	7.68E9	1.40E10	1.16E10	1.25E10
^{24}Na	4.93E9	2.38E10	7.68E9	1.40E10	1.16E10	1.25E10
^{40}Ar	4.14E9	7.09E9	5.96E9	7.97E9	6.00E9	7.50E9
^{54}Mn	3.49E7	4.25E9	2.41E9	5.14E9	4.35E9	4.51E9
^{55}Fe	3.66E7	6.28E3	3.35E3	1.5E4	5.85E4	7.63E4
^{60}Co	3.01E8	4.97E9	2.66E9	6.09E9	5.24E9	5.28E9
^{60}Co	8.64E8	1.37E10	1.12E10	1.55E10	1.18E10	1.45E10
^{63}Ni	3.97E7	1.05E4	5.59E3	1.91E4	9.31E4	1.27E5
^{65}Zn	5.95E7	3.07E3	2.19E9	3.56E9	2.82E9	3.25E9
^{85}Kr	2.25E9	1.15E7	4.58E6	1.48E7	1.37E7	1.23E7
^{88}Kr	3.11E9	1.12E10	1.14E10	1.24E10	7.29E9	1.18E10
^{88}Rb	1.82E10	3.70E9	3.64E9	4.06E9	2.48E9	3.88E9
^{90}Sr	5.19E9	6.96E5	4.19E5	6.31E5	6.92E5	7.37E5
^{90}Sr	1.76E9	0.0	0.0	0.0	0.0	0.0
^{90}Y	8.25E9	6.89E1	3.97E1	3.75E2	4.64E2	5.55E2
^{91m}V	2.39E8	2.72E9	1.15E9	3.48E9	3.17E9	2.91E9
^{91}Y	5.25E9	1.96E7	1.53E7	2.23E7	1.72E7	2.07E7
^{92}Zr	1.04E9	3.76E9	1.96E9	4.62E9	3.99E9	4.00E9
^{95}Nb	3.91E8	3.90E9	2.07E9	4.77E9	4.10E9	4.14E9
^{99}Tc	7.60E8	0.0	0.0	0.0	0.0	0.0
^{103}Ru	6.72E8	2.41E9	9.56E8	3.13E9	2.90E9	2.58E9
^{106}Ru	9.10E7	0.0	0.0	0.0	0.0	0.0
^{103m}Rh	3.36E8	2.31E5	2.14E5	1.27E6	1.67E6	1.19E6
^{106}Rh	1.25E10	1.06E9	5.00E8	1.33E9	1.19E9	1.13E9
^{110m}Ag	6.01E8	1.44E10	9.19E9	1.72E10	1.42E10	1.53E10
^{110}Ag	1.04E10	1.56E8	7.54E7	1.94E8	1.72E8	1.66E8
^{125}Sb	8.79E8	2.14E9	9.52E8	2.83E9	2.56E9	2.31E9
^{125m}Te	9.57E8	2.53E7	1.92E7	8.39E7	7.78E7	5.72E7
^{127m}Te	7.22E8	6.88E6	5.24E6	2.34E7	2.23E7	1.61E7
^{127}Te	2.00E9	2.48E7	1.09E7	3.44E7	3.13E7	2.69E7
^{129m}Te	2.40E9	1.55E8	7.93E7	2.04E8	1.79E8	1.72E8
^{129}Te	4.85E9	2.87E8	1.35E8	3.81E8	3.44E8	3.12E8
^{129}I	4.84E8	2.52E7	1.90E7	7.63E7	6.34E7	4.85E7
^{131}I	1.71E9	1.95E9	9.03E8	2.73E9	2.46E9	2.12E9
^{133m}Xe	1.71E9	1.39E8	8.20E7	2.47E8	2.17E8	1.69E8
^{133}Xe	1.22E9	1.44E8	8.56E7	3.38E8	1.85E8	1.80E8
^{135m}Xe	8.56E8	2.20E9	8.95E8	2.84E9	2.62E9	2.36E9
^{135}Xe	2.82E9	1.27E9	6.87E8	1.96E9	1.74E9	1.41E9
^{134}Cs	1.45E9	7.69E9	4.08E9	9.83E9	8.58E9	8.47E9
^{135}Cs	5.08E8	0.0	0.0	0.0	0.0	0.0
^{135}Cs	1.53E9	0.0	0.0	0.0	0.0	0.0
^{137m}Ba	5.67E8	3.04E9	1.45E9	3.79E9	3.35E9	3.24E9

Table 7.2 (continued)

	Body surface (E dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton	Testes	Total body
^{144}Ce	8.22E8	8.66E7	4.83E7	1.75E8	1.10E8	1.01E8
^{144}Pr	4.06E8	1.71E7	1.32E7	4.91E7	3.33E7	2.95E7
^{144}Pr	1.07E10	1.77E8	1.59E8	1.98E8	1.31E8	1.85E8
^{147}Pm	5.59E8	1.71E4	8.90E3	3.38E4	2.02E4	1.98E4
^{151}Sm	1.80E8	9.91E2	9.26E2	5.73E3	7.81E3	5.62E3
^{154}Eu	2.52E9	6.43E9	4.44E9	7.79E9	6.15E9	5.85E9
^{155}Eu	5.54E8	2.51E8	1.39E8	5.54E8	2.89E8	3.09E8
^{207}Tl	4.39E9	1.10E7	6.55E6	1.31E7	1.10E7	1.16E7
^{208}Tl	5.30E9	1.91E10	1.85E10	2.13E10	1.31E10	2.02E10
^{209}Tl	6.10E9	1.18E10	1.02E10	1.39E10	1.01E10	1.25E10
^{209}Pb	1.77E9	0.0	0.0	0.0	0.0	0.0
^{210}Pb	3.13E8	4.94E6	3.97E6	1.32E7	6.72E6	7.56E6
^{211}Pb	4.06E9	2.57E8	1.30E8	3.28E8	2.87E8	2.75E8
^{212}Pb	1.55E9	7.25E8	4.06E8	1.23E9	9.78E8	8.22E8
^{214}Pb	2.61E9	1.25E9	6.14E8	1.86E9	1.61E9	1.37E9
^{210}Bi	3.48E9	0.0	0.0	0.0	0.0	0.0
^{211}Bi	8.20E7	2.28E8	1.07E8	3.33E8	2.95E8	2.50E8
^{212}Bi	4.16E9	9.78E8	6.87E8	1.15E9	9.14E8	1.04E9
^{213}Bi	3.99E9	6.78E8	2.94E8	9.20E8	8.30E8	7.33E8
^{214}Bi	5.76E9	8.28E9	6.93E9	9.43E9	6.74E9	8.73E9
^{210}Po	0.0	4.33E4	2.38E4	5.26E4	4.49E4	4.60E4
^{213}Po	0.0	1.55E5	8.32E4	1.89E5	1.62E5	1.64E5
^{214}Po	0.0	5.58E5	3.04E5	6.80E5	5.81E5	5.93E5
^{217}At	0.0	1.52E6	6.14E5	2.01E6	1.86E6	1.64E6
^{219}Rn	5.18E7	2.85E8	1.38E8	4.20E8	3.73E8	3.12E8
^{220}Rn	0.0	2.82E6	1.18E6	3.61E6	3.30E6	3.02E6
^{222}Rn	0.0	2.00E6	7.96E5	2.59E6	2.40E6	2.14E6
^{221}Fr	8.18E7	1.56E8	9.06E7	2.59E8	2.19E8	1.75E8
^{223}Fr	3.33E9	2.39E8	1.43E8	1.12E9	8.06E8	7.29E8
^{223}Ra	6.43E8	6.39E8	3.33E8	1.12E9	8.06E8	7.29E8
^{224}Ra	1.84E7	5.04E7	2.80E7	8.06E7	6.99E7	5.61E7
^{225}Ra	9.42E8	2.11E7	1.65E7	5.92E7	3.63E7	3.49E7
^{226}Ra	3.11E7	3.37E7	2.00E7	5.87E7	4.68E7	3.82E7
^{228}Ra	1.14E8	1.68E-2	8.99E-3	3.08E-2	1.50E-1	2.04E-1
^{225}Ac	1.57E8	7.41E7	4.01E7	1.45E8	9.07E7	8.65E7
^{227}Ac	1.17E8	5.70E5	2.93E5	1.15E6	6.72E5	6.72E5
^{228}Ac	3.82E9	4.82E9	3.20E9	5.85E9	4.73E9	5.12E9
^{227}Th	3.44E8	5.18E8	2.81E8	8.42E8	6.98E8	5.28E8
^{228}Th	1.71E8	9.28E6	5.28E6	1.81E7	1.18E7	1.11E7
^{229}Th	9.42E8	4.05E8	2.24E8	8.15E8	4.99E8	4.81E8
^{230}Th	1.18E8	1.66E6	1.04E6	3.54E6	2.14E6	2.18E6
^{231}Th	1.47E9	6.09E7	3.44E7	1.37E8	7.73E7	8.00E7
^{232}Th	9.08E7	6.72E5	4.31E5	1.54E6	8.78E5	9.85E5
^{234}Th	5.33E8	3.49E7	1.97E7	7.79E7	3.92E7	4.32E7
^{231}Pa	3.40E8	1.50E8	7.64E7	2.36E8	2.02E8	1.69E8
^{233}Pa	1.70E9	9.51E8	4.67E8	1.51E9	1.22E9	1.06E9
^{234}Pa	7.30E9	5.78E7	3.55E7	7.05E7	5.71E7	6.15E7
^{234}Pa	4.07E9	1.02E10	6.31E9	1.28E10	1.03E10	1.09E10

Table 7.2 (continued)

	Body surface ($\pm$ dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton (γ and x-ray dose rate)	Testes	Total body
^{235}U	1.36E8	1.11E6	6.84E5	2.38E6	1.52E6	1.60E6
^{238}U	4.24E7	1.43E6	7.23E5	2.93E6	1.71E6	1.79E6
^{232}Th	1.02E8	4.68E5	2.96E5	1.08E6	6.85E5	7.94E5
^{235}U	2.73E8	7.58E8	4.45E8	1.33E9	1.04E9	8.59E8
^{238}U	8.58E7	1.51E5	1.15E5	4.13E5	3.05E5	4.01E5
^{232}Th	7.61E7	1.26E5	9.60E4	3.47E5	2.62E5	3.44E5
^{237}Np	5.55E8	1.08E8	5.93E7	2.31E8	1.30E8	1.33E8
^{239}Pu	2.13E9	8.24E8	4.29E8	1.48E9	1.04E9	9.38E8
^{238}Pu	7.45E7	1.21E5	8.25E4	5.53E5	3.17E5	4.30E5
^{239}Pu	4.63E7	2.77E5	1.41E5	5.85E5	3.87E5	4.25E5
^{240}Pu	7.61E7	1.25E5	8.73E4	3.63E5	3.14E5	4.24E5
^{241}Pu	4.78E7	0.0	0.0	0.0	0.0	0.0
^{242}Pu	5.94E7	1.05E5	7.19E4	3.00E5	2.54E5	3.42E5
^{241}Am	1.59E8	7.64E7	5.49E7	1.90E8	8.98E7	1.04E8
^{243}Am	1.39E8	2.10E8	1.28E8	4.68E8	2.40E8	2.68E8
^{244}Cm	6.91E7	1.22E5	7.77E4	3.88E5	3.54E5	4.88E5
^{245}Cm	1.12E9	6.23E8	3.29E8	1.10E9	7.99E8	7.08E8
^{246}Cm	5.99E7	6.27E4	3.72E4	2.36E5	2.74E5	3.32E5

Table 7.3 External dose conversion factors for exposure to contaminated ground surface (millirem/year/ $\mu\text{Ci}/\text{cm}^2$)

	Body surface (β dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton	Testes	Total body
^1H	0.0	0.0	0.0	0.0	0.0	0.0
^{14}C	0.0	0.0	0.0	0.0	0.0	0.0
^{23}Na	2.22E5	2.20E6	1.40E6	2.64E6	2.21E6	2.33E6
^{24}Na	7.87E6	3.56E6	3.63E6	3.87E6	2.38E6	3.76E6
^{41}Ar	6.00E6	1.23E6	1.04E6	1.39E6	1.04E6	1.30E6
^{54}Mn	0.0	8.19E5	4.63E5	9.89E5	8.37E5	8.69E5
^{55}Fe	0.0	1.91E1	6.33E9	2.17E1	1.05E2	1.44E2
^{58}Co	8.52E3	9.70E5	5.16E5	1.19E6	1.03E6	1.03E6
^{60}Co	6.68E3	2.40E6	1.96E6	2.72E6	2.07E6	2.54E6
^{63}Ni	0.0	2.23E1	1.19E1	4.07E1	1.98E2	2.71E2
^{65}Zn	6.01E-4	5.56E5	3.96E5	6.44E5	5.12E5	5.88E5
^{84}Kr	1.26E6	2.36E3	9.42E2	3.05E3	2.83E3	2.53E3
^{86}Kr	2.84E6	1.76E6	1.76E6	1.95E6	1.13E6	1.84E6
^{87}Rb	1.95E7	5.96E5	5.76E5	6.56E5	4.07E5	6.24E5
^{90}Sr	8.57E6	1.32E2	7.92E1	1.57E2	1.31E2	1.40E2
^{90}Sr	2.74E5	0.0	0.0	0.0	0.0	0.0
^{90}Y	1.31E7	1.19E-1	6.73E-2	6.44E-1	7.92E-1	9.65E-1
^{91}Y	4.41E5	5.54E5	2.35E5	7.09E5	6.46E5	5.93E5
^{91}Y	8.88E6	3.46E3	2.71E3	3.95E3	3.04E3	3.66E3
^{95}Zr	2.29E4	7.38E5	3.84E5	9.06E5	7.84E5	7.84E5
^{95}Nb	0.0	7.61E5	4.05E5	9.31E5	8.01E5	8.09E5
^{99}Tc	0.0	0.0	0.0	0.0	0.0	0.0
^{101}Ru	8.98E4	4.97E5	1.97E5	6.45E5	5.98E5	5.32E5
^{101}Ru	0.0	0.0	0.0	0.0	0.0	0.0
^{101}Rh	0.0	2.32E2	2.22E2	1.41E3	1.94E3	1.36E3
^{106}Rh	1.70E7	2.11E5	9.87E4	2.66E5	2.39E5	2.25E5
^{110}Ag	3.39E4	2.71E6	1.70E6	3.24E6	2.70E6	2.88E6
^{110}Ag	1.54E7	3.10E4	1.50E4	3.87E4	3.42E4	3.31E4
^{123}Sb	1.05E5	4.42E5	1.98E5	5.91E5	5.34E5	4.80E5
^{125}Te	0.0	1.11E4	8.45E3	3.68E4	3.42E4	2.50E4
^{127}Te	3.66E4	3.10E3	2.37E3	1.05E4	9.96E3	7.21E3
^{127}Te	8.24E5	5.25E3	2.30E3	7.28E3	6.62E3	5.69E3
^{129}Te	3.05E6	3.20E4	1.67E4	4.48E4	3.97E4	3.71E4
^{129}Te	7.22E6	5.90E4	2.77E4	7.98E4	7.24E4	6.50E4
^{131}I	0.0	1.32E4	9.89E3	4.02E4	3.41E4	2.57E4
^{131}I	3.25E5	4.12E5	1.91E5	5.78E5	5.22E5	4.49E5
^{133}Xe	0.0	3.58E4	2.19E4	7.05E4	6.23E4	4.77E4
^{133}Xe	1.56E0	4.01E4	2.45E4	9.70E4	5.79E4	5.57E4
^{135}Xe	1.31E6	4.53E5	1.85E5	5.87E5	5.41E5	4.87E5
^{135}Xe	2.48E6	2.74E5	1.49E5	4.25E5	3.77E5	3.04E5
^{137}Cs	5.17E5	1.57E6	7.99E5	1.94E6	1.69E6	1.67E6
^{137}Cs	0.0	0.0	0.0	0.0	0.0	0.0
^{137}Cs	3.99E5	0.0	0.0	0.0	0.0	0.0
^{137}Ba	1.19E6	6.06E5	2.91E5	7.59E5	6.70E5	6.47E5
^{140}Ce	0.0	2.04E4	1.16E4	4.19E4	2.65E4	2.50E4
^{144}Pr	0.0	6.47E3	4.98E3	1.35E4	1.26E4	1.12E4
^{144}Pr	1.55E7	2.94E4	2.55E4	3.34E4	2.27E4	3.10E4

Table 7.3 (continued)

	Body surface (β dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton	Testes	Total body
¹⁴⁷ Pm	0.0	3.82E0	2.00E0	7.57E0	4.54E0	4.43E0
¹⁵¹ Sm	0.0	1.08E0	9.98E-1	6.14E0	8.50E0	6.24E0
¹⁵⁴ Eu	1.77E6	1.20E6	8.15E5	1.47E6	1.15E6	1.28E6
¹⁵⁵ Eu	0.0	6.19E4	3.50E4	1.38E5	7.22E4	7.73E4
²⁰⁷ Tl	6.69E6	2.09E3	1.24E3	2.49E3	2.08E3	2.21E3
²⁰⁸ Tl	8.10E6	2.94E6	2.72E6	3.32E6	2.13E6	3.11E6
²⁰⁹ Tl	9.62E6	2.08E6	1.74E6	2.49E6	1.82E6	2.22E6
²⁰⁹ Pb	4.80E5	0.0	0.0	0.0	0.0	0.0
²¹⁰ Pb	0.0	1.66E3	1.32E3	4.42E3	2.62E3	3.05E3
²¹¹ Pb	5.90E6	5.14E4	2.59E4	6.60E4	5.77E4	5.51E4
²¹² Pb	2.01E4	1.61E5	9.02E4	2.75E5	2.16E5	1.83E5
²¹⁴ Pb	9.16E5	2.68E5	1.32E5	4.02E5	3.45E5	2.96E5
²¹⁰ Bi	4.51E6	0.0	0.0	0.0	0.0	0.0
²¹¹ Bi	6.53E2	4.90E4	2.30E4	7.16E4	6.32E4	5.37E4
²¹² Bi	6.37E6	1.80E5	1.23E5	2.14E5	1.71E5	1.92E5
²¹³ Bi	5.36E6	1.42E5	6.14E4	1.93E5	1.74E5	1.53E5
²¹⁴ Bi	8.50E6	1.44E6	1.17E6	1.65E6	1.20E6	1.51E6
²¹⁰ Po	0.0	8.40E0	4.62E0	1.02E1	8.70E0	8.92E0
²¹³ Po	0.0	3.01E1	1.62E1	3.68E1	3.15E1	3.20E1
²¹⁴ Po	0.0	1.08E2	5.91E1	1.32E2	1.13E2	1.15E2
²¹⁷ At	0.0	3.17E2	1.28E2	4.18E2	3.88E2	3.41E2
²¹⁹ Rn	5.46E2	6.11E4	2.96E4	9.02E4	8.00E4	6.71E4
²²⁰ Rn	0.0	5.76E2	2.41E2	7.38E2	6.75E2	6.16E2
²²¹ Fr	0.0	3.42E4	1.99E4	5.70E4	4.80E4	3.85E4
²²³ Fr	3.53E6	5.91E4	3.69E4	1.20E5	7.30E4	7.39E4
²²³ Ra	2.74E1	1.43E5	7.47E4	2.53E5	1.80E5	1.64E5
²²⁴ Ra	0.0	1.0E4	6.11E3	1.76E4	1.52E4	1.22E4
²²⁵ Ra	7.03E0	8.94E3	6.96E3	2.51E4	1.57E4	1.53E4
²²⁶ Ra	0.0	7.43E3	4.41E3	1.30E4	1.03E4	8.45E3
²²⁸ Ra	0.0	3.45E-5	1.84E-5	6.31E-5	3.07E-4	4.19E-4
²²⁸ Ac	0.0	1.70E4	9.17E3	3.34E4	2.11E4	2.04E4
²²⁷ Ac	0.0	1.31E2	6.75E1	2.69E2	1.70E2	1.77E2
²²⁸ Ac	4.15E6	9.04E5	5.90E5	1.11E6	9.00E5	9.64E5
²²⁷ Th	0.0	1.15E5	6.27E4	1.89E5	1.55E5	1.31E5
²²⁸ Th	0.0	2.17E3	1.23E3	4.32E3	2.98E3	2.95E3
²²⁹ Th	0.0	9.46E4	5.25E4	1.93E5	1.20E5	1.17E5
²³⁰ Th	0.0	4.38E2	2.70E2	9.77E2	7.69E2	8.92E2
²³¹ Th	0.0	1.55E4	8.85E3	3.65E4	2.37E4	2.55E4
²³² Th	0.0	2.09E2	1.31E2	5.11E2	4.77E2	6.10E2
²³⁴ Th	0.0	8.57E3	4.87E3	1.93E4	9.95E3	1.11E4
²³¹ Pa	0.0	3.31E4	1.70E4	5.33E4	4.64E4	3.96E4
²³³ Pa	2.00E4	2.08E5	1.02E5	3.31E5	2.68E5	2.34E5
^{234m} Pa	1.17E7	1.09E4	6.70E3	1.34E4	1.08E4	1.16E4
²³⁴ Pa	1.70E6	1.96E6	1.19E6	2.47E6	2.00E6	2.10E6
²³² U	0.0	3.40E2	2.04E2	8.33E2	8.24E2	1.05E3
²³³ U	0.0	3.58E2	1.81E2	7.86E2	6.18E2	7.33E2

Table 7.3 (continued)

	Body surface (β dose rate)	Lungs	Ovaries (γ and x-ray dose rate)	Skeleton	Testes	Total body
^{238}U	0.0	1.77E2	1.07E2	4.82E2	5.53E2	7.43E2
^{235}U	0.0	1.67E5	9.80E4	2.93E5	2.31E5	1.91E5
^{234}U	0.0	9.96E1	6.17E1	3.12E2	4.43E2	6.21E2
^{232}U	0.0	8.59E1	5.32E1	2.71E2	3.89E2	5.46E2
^{230}Th	0.0	2.64E4	1.46E4	5.78E4	3.49E4	3.60E4
^{234}Th	3.58E4	1.84E5	9.58E4	3.33E5	2.33E5	2.13E5
^{238}Pu	0.0	1.08E2	5.90E1	3.88E2	5.59E2	7.97E2
^{239}Pu	0.0	9.11E1	4.56E1	2.44E2	2.71E2	3.60E2
^{240}Pu	0.0	1.06E2	5.91E1	3.78E2	5.36E2	7.64E2
^{241}Pu	0.0	0.0	0.0	0.0	0.0	0.0
^{242}Pu	0.0	8.47E1	4.69E1	3.00E2	4.24E2	6.03E2
^{243}Am	0.0	2.20E4	1.58E4	5.50E4	2.69E4	3.16E4
^{244}Am	0.0	5.23E4	3.21E4	1.22E5	6.05E4	6.79E4
^{245}Cm	0.0	1.21E2	6.29E1	4.94E2	6.28E2	9.05E2
^{246}Cm	0.0	1.39E5	7.33E4	2.48E5	1.80E5	1.62E5
^{247}Cm	0.0	9.80E1	4.91E1	4.21E2	5.59E2	8.07E2

It is important to emphasize that contributions from daughter nuclides, which may be present in a given source term or which will build up due to decay of the parent radionuclide, are not included in the parent nuclide's dose conversion factor. Care must be taken, when preparing a source term to be used in the calculation of dose from a fuel cycle facility, to include such daughter nuclides as will build up during the exposure period in question, either during atmospheric transport or following deposition. External dose conversion factors for all significant daughters of breeder cycle radionuclides of interest have also been included in Tables 7.2 and 7.3, to be used in such calculations. The reader is referred to Kocher⁶ for a detailed discussion of this methodology, and for diagrams of half-lives and branching fractions of radionuclides of interest.

REFERENCES

1. Killough, G. G., D. E. Dunning, Jr., S. R. Bernard, and J. C. Pleasant. 1978. *Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities.* ORNL/NUREG/TM-190, Vol. 1.
2. Dunning, D. E., S. R. Bernard, P. J. Walsh, G. G. Killough, and J. C. Pleasant. 1979. *Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities.* ORNL/NUREG/TM-190, Vol. 2.
3. Morrow, P. E., D. V. Bates, B. R. Fish, T. F. Hatch, and T. T. Mercer. 1966. Deposition and retention models for internal dosimetry of the human respiratory tract. *Health Phys.* 12:173-207.
4. Eve, I. S. 1966. A review of the physiology of the gastrointestinal tract in relation to radiation doses from radioactive materials. *Health Phys.* 12:131-62.
5. International Commission on Radiological Protection. 1977. *Publication 26, Annals of the ICRP* 1, 1.
6. Kocher, D. C. 1979. *Dose-Rate Conversion Factors for External Exposure to Photon and Electron Radiation from Radionuclides Occurring in Routine Releases from Nuclear Fuel Cycle Facilities.* ORNL/NUREG/TM-283.

8. DOSES DUE TO ATMOSPHERIC RELEASES OF ^3H AND ^{14}C

Elizabeth L. Stulen

8.1 Tritium (^3H)

8.1.1 Introduction

Various methodologies exist for the estimation of ingestion dose from continuous atmospheric releases of tritium.¹⁻⁷ Most of these methodologies are based on the specific activity model (according to which the tritium content of the plant or body water, related to the content of stable hydrogen, is the same as that of the water in air), but the resulting dose estimates may vary by up to a factor of 10 depending on the equilibrium assumptions employed for vegetation, beef, and milk.

Bush,¹ Evans,² Killough and McKay,³ and Moore,⁴ base their dose calculations on the assumption that body hydrogen or body water is uniformly labeled with tritium (an equilibrium ratio of 1.0). However, their approaches are somewhat different in that Bush¹ and Evans² estimate dose based on the average energy delivered by HTO to body water that contains 1 $\mu\text{Ci/liter}$. Killough and McKay,³ and Moore,⁴ follow the food and water ingestion pathway.

The U.S. Nuclear Regulatory Commission,⁵ and Ng et al.⁶ assume various uptake fractions for tritium in plants, beef, and milk, resulting in doses 1/4-1/2, respectively, those estimated by the Killough and McKay,³ and Moore⁴ methodologies.

Vogt⁷ recommends modifying the basic specific activity model so as to consider the water supply of plants from the soil water and precipitation deposited on the leaves, as well as that from humidity.

A review of the literature available for estimating dose from atmospheric releases of tritium, as well as the range of reported values for the input parameters is being undertaken at the Oak Ridge National Laboratory (ORNL). Parameters being considered include: quality factor, half-life, air-to-water dilution, and organic binding of tritium.

At present, the methodology outlined in the AIRDOS-EPA computer code⁸ is being employed at ORNL for estimating tritium doses. This methodology has been modified slightly from that reported by Moore⁴ and may be modified further to reflect the changes suggested by the literature review in progress.

8.1.2 The AIRDOS-EPA methodology for ^3H

The AIRDOS-EPA code⁸ treats tritium in the following manner: if tritium (T) is released to the atmosphere as HT or T_2 , atoms of T may exchange with hydrogen atoms in water molecules in the air, and the plume is treated as though it contained HTO initially. The tritium may then be assumed to follow water almost precisely through the environment. For this reason, doses from drinking water are included for tritium. Rather than attempting to relate the doses to the ground deposition rate, it is assumed that doses from ingestion of food and drinking water at an environmental location are proportional to the tritium concentration in air.³

The total ingestion dose from tritium if the source of all of an individual's food and drinking water is assumed to be at his specific environmental location is:

$$D_t = C_f X + C_w X, \quad (8.1)$$

where

D_t = tritium ingestion dose from food and water (rem/year),

C_f = dose conversion factor for food ($\text{rem-cm}^3/\text{pCi-year}$),

C_w = dose conversion factor for drinking water ($\text{rem-cm}^3/\text{pCi-year}$), and

χ = groundlevel concentration of tritium in air at the environmental location (pCi/cm^3).

The total-body dose conversion factor for ingestion is $8.3 \times 10^{-5} \text{ rem/pCi}$.¹⁰ This number is used to derive the value of C_f and C_w , based on the specific activity of tritium in atmospheric moisture with an average specific humidity of $8 \text{ g H}_2\text{O/m}^3$ of air (ref. 3). If tritium in food is in equilibrium with atmospheric tritium and man consumes 1638 g of water daily in his food, C_f is $6.18 \text{ rem-cm}^3/\text{pCi-year}$. The C_w value for an assumed daily drinking water intake of 1512 g is $5.70 \text{ rem-cm}^3/\text{pCi-year}$. This value is used, however, only if the source of each individual's drinking water is assumed to be at his specific environmental location. For all other cases, C_w is reduced to account for dilution by distant sources.

The code artificially breaks down the ingestion dose from tritium into percentage contributions of 50.5% from vegetables (including fruits and grains), 17.5% from meat and 31.0% from milk. The percentages are based on approximate water contents of foods: 82.4% for vegetables, 62.3% for meat, and 87.5% for milk, for daily intakes of 0.532 kg vegetables, 0.258 kg meat, and 0.307 kg milk (Sect. 6).

Tritium doses via inhalation of air and skin absorption are estimated by the code and added to the dose estimated to be received via ingestion. The dose conversion factor for inhalation¹⁰ of air containing tritium includes a 50% contribution for skin absorption from air.

8.2 Carbon-14 (^{14}C)

8.2.1 Introduction

Methods for calculating dose to man from atmospheric releases of ^{14}C are also being reviewed. Basically, these methods involve a specific activity approach, although they differ in their treatment of pathways. The NRC (ref. 5) assumes that the ratio in vegetation of ^{14}C to natural carbon is the same as the ratio in the atmosphere of ^{14}C to natural carbon. The ingestion of ^{14}C in foods, including various transfer coefficients for vegetation-to-beef and vegetation-to-milk, is then used to calculate the ultimate dose to man. For intermittent releases, an account is made for achievement of a fractional equilibrium ratio by relating total annual releases time to the total annual time during which photosynthesis occurs.

The methodology currently in use at ORNL, and described in the AIRDOS-EPA code,⁸ assumes that the ^{14}C in body tissues is in equilibrium with the ^{14}C in the atmosphere during continuous releases. Since about 99% of the dose from ^{14}C is via the ingestion pathway,⁹ a breakdown based on percent contribution to diet of the three major food pathways, as well as the carbon content of each food, is utilized in the AIRDOS-EPA code to facilitate a pathway analysis.

8.2.2 The AIRDOS-EPA methodology for ^{14}C

The description of the treatment of ^{14}C in the AIRDOS-EPA code⁸ is as follows: if ^{14}C is released in the form of $^{14}\text{CO}_2$, it will mix with atmospheric CO_2 and become available for plant photosynthesis. Cattle grazing on pasture will take in ^{14}C in grass, and then man will receive

it in milk and beef. Factors in a data statement are used in the code to multiply by the concentration of ^{14}C in air to obtain an ingestion dose for each reference organ. These dose conversion factors, listed in Table 8.1 are based on specific activity calculations for ^{14}C in body tissues in equilibrium with ^{14}C in the atmosphere.⁹

Nearly all the ^{14}C dose comes from ingestion. An artificial breakdown by the three food pathways is accomplished by estimating carbon intakes for vegetables, milk, and meat, based on a carbon content of approximately 8% for vegetables, 24% for meat, and 7% for milk.

Table 8.1. Dose conversion factors for ^{14}C

Organ	Dose conversion factors ^a (rem-cm ³ /pCi-year)
Whole body	1.16×10^3
Red marrow	2.03×10^3
Lungs	5.07×10^2
Endosteal cells	1.85×10^3
Stomach wall	7.43×10^2
Lower large intestine wall	8.92×10^2
Thyroid	5.27×10^2
Liver	7.30×10^2
Kidneys	6.49×10^2
Testes	4.46×10^2
Ovaries	4.46×10^2

^aThese factors are taken from ref. 8 and are based on the assumption that the specific activity in human tissue is equal to the average steady state value in the atmosphere (ref. 9).

REFERENCES

1. Bush, W. R. 1972. *Assessing and Controlling the Hazard from Tritiated Water*. AECL-4150.
2. Evans, A. G. 1969. New dose estimates from chronic tritium exposures. *Health Phys.* 16:57-63
3. Killough, G. G. and L. R. McKay. 1976. *A Methodology for Calculating Radiation Doses from Radioactivity Released to the Environment*. ORNL-4992.
4. Moore, R. E. 1977. *The AIRDOS-II Computer Code for Estimating Radiation Dose to Man from Airborne Radionuclides in Areas Surrounding Nuclear Facilities*. ORNL-5245.
5. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.109, Calculation of annual doses to man from routine releases of reactor effluents for the purpose of evaluating compliance with 10 CFR Part 50, Revision 1.
6. Ng, Y. C., W. A. Phillips, Y. E. Richer, R. K. Tandy, and S. E. Thompson. 1978. *Methodology for Assessing Dose Commitment to Individuals and to the Population from Ingestion of Terrestrial Foods Contaminated by Emissions from a Nuclear Fuel Reprocessing Plant at the Savannah River Plant*. UCID-17743.
7. Vogt, K. J. 1978. *Models for the Assessment of the Environmental Exposure by Tritium Released from Nuclear Installations*. Proc. of International Symposium on Behavior of Tritium in the Environment. IAEA-SM-232/15. In press.
8. Moore, R. E., C. F. Baes III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller. 1979. *AIRDOF-III: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides*. ORNL-5532.

9. Killough, G. G. and P. S. Rohwer. 1978. A new look at the dosimetry of ^{14}C released to the atmosphere as carbon dioxide. *Health Phys.* 34: 141-59.
10. Killough, G. G., D. E. Dunning, Jr., S. R. Bernard, and J. C. Pleasant. 1978. *Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities.* Vol. 1, NUREG/CR-0150, ORNL/NUREG/TM-190.

Appendix A
A COMPARISON OF EXPOSURE PATHWAYS

TABLE A-1

Releases of radionuclides to the environment from nuclear facilities, including breeder reactors, are generally either airborne or waterborne. In either case, there are several pathways by which humans may be exposed to radioactivity from either type of release. The main body of this report presents information on the models and parameter values which may be used to estimate the dose to humans as a result of these exposures. The purpose of this appendix is to illustrate for both release types the relative contribution of various pathways of exposure to the total dose due to radionuclide releases from various nuclear facilities.

This comparison is accomplished through a review of pathway comparisons performed for various facilities of the light water reactor (LWR) fuel cycle.

The primary reference for the discussion of atmospheric releases was published by Hoffman and Kaye.¹ Most of the data mentioned here were taken from tables published in that paper. A primary reference for the importance of various pathways following an aquatic release is a paper by Soldat² from the same symposium volume. While these examples do not include breeder reactor facilities, the models used in these comparisons are basically the same as those discussed in this report. As such, they should serve the illustrative purposes desired for this appendix. Further, no single comparison or sets of comparison can define the critical pathway or pathways to man from a given type of facility since any such comparison depends on the radionuclide make-up of the release, the rate of release, the dietary and behavioral habits

of the human receptor, and the existence of a particular pathway at a given site.

A.1 Atmospheric Release

Hoffman and Kaye¹ calculated the maximum individual exposure from the major terrestrial pathways for atmospheric releases from model LWR fuel cycle facilities. Each model facility resulted in a dose to the organs of some limiting individual (adult, child or both) after assuming a generic location and respiratory parameters designed to maximize the exposure to humans via that pathway. These doses were then compared to applicable radiation standards to assess the significance of the particular pathway. For each facility, the principal nuclides and principal exposure pathways for organs of the limiting child or adult were listed. Rather than reproduce those seven tables here, we have summarized the information in Table A.1. A reader desiring the detail of the original report should see Tables 8-14 of ref. 1.

There are several observations about the pathways which may be made from the information in Table A.1. One conclusion is that for the considered release and human behavioral conditions milk is an extremely important pathway of exposure, especially for children. Of the 31 entries in the rightmost column in Table A.1, "principal pathways for principal nuclides," 16 are milk ingestion. Milk ingestion was often the most important pathway of radiation exposure when children alone were considered.

For adults, milk ingestion was slightly less important, but milk, vegetable, and beef ingestion still accounted for one-half of the principal pathway entries. Immersion in contaminated air was an important

Table A.1 Principal pathways and nuclides for maximally exposed individuals from model facilities in the uranium fuel cycle

Source	Limiting receptor	Principal nuclides	Reference organ	Principal pathway for principal nuclides
Uranium mill	Child	^{226}Ra	Bone, lung, thyroid, total body	Milk (93-94%)
	Adult	^{226}Ra	GI, liver	Milk (58-59%)
		^{238}U	GI	Vegetables (58-80%)
		^{210}Po	Kidney, liver	Vegetables (67%)
		^{230}Th ^{210}Po	Kidney, liver Liver	Inhalation (97%) Vegetable (80%)
Conversion facility	Child	^{226}Ra	Lung, thyroid, total body	Milk (93%)
	Adult	^{234}U	Lung	Inhalation (93%)
		^{230}Th	Bone, kidney	Inhalation (97-98%)
		^{226}Ra	Bone	Milk (60%)
Uranium enrichment facility	Child	^{234}U	Bone	Milk (56%)
		^{234}U	Lung	Inhalation (92%)
Fuel fabrication plant	Child	^{234}U	Bone	Milk (56%)
Boiling water reactor	Child	^{131}I	Thyroid	Milk (99%)
	Adult	^{85}Kr	Thyroid	Immersion in air (100%)
		^{86}Rb	GI	Inhalation (72%)
		^{85}Kr	GI	Immersion in air (100%)
	Child/Adult	^{85}Kr	Bone, skin, total body, lung, liver, kidney	Immersion in air (100%)
		^{14}C	Bone, total body	Milk (88%)
		^{133}Xe	Skin	Immersion in air (100%)
		^{86}Rb	Lung, liver, kidney	Immersion in air (59-76%)
		^{131}I	Thyroid	Milk (96%)
		^{129}I	Thyroid, total body, kidney	Milk (59-75%)
Fuel reprocessing	Child	^{90}Sr	Bone, lung	Milk (83-93%)
		^{90}Zr	Bone, liver, total body, kidney, lung	Milk (54-96%)
		^{134}Cs	Liver	Milk (84%)
		^{137}Cs	Bone, total body	Milk (88%)
		^{129}I	Thyroid	Milk (46%)
	Adult	^{131}I	Thyroid	Milk (77%)
		^{106}Ru	GI	Beef (99%)
		^{85}Kr	Skin	Immersion in air (100%)
	Child/Adult	^{85}Kr	Skin	Immersion in air (100%)

Data taken from Tables 8-14 of ref. 1.

Range of percentages indicates the contribution by the principal nuclide to the organs of reference over the listed pathway.

pathway for the boiling water reactor and the fuel reprocessing plant. Inhalation was a principal pathway for adults subject to releases from uranium mill and the uranium conversion facilities and for children near uranium enrichment facilities. Exposure to contaminated surfaces was found to be relatively unimportant for the releases considered in this paper.

As might be expected, the principal radionuclides released were not the same for each facility type. Important releases early in the fuel cycle tend to be composed of naturally occurring radionuclides, while the principal radionuclides released by the reactor and reprocessing plant are more frequently fission products.

It is apparent from this brief discussion that the importance of the various pathways of exposure is very much a function of the nuclides released. Therefore, it follows that the importance of any pathway varies between types of facilities and within types of facilities, depending upon the particular radionuclides present and the emission control equipment installed.

Studies similar to the one of Hoffman and Kaye¹ were performed by various authors, but for specific types of facilities and for population dose commitments rather than maximum individual doses (Table A.2). The data in Table A.2 from these studies are not identified by radionuclide, but the conclusions drawn from Table A.1 hold for these other studies; namely, ingestion and inhalation appear to be the most important pathways, but the relative rankings of the pathways varies between and within facility types due to differences in assumptions about radionuclide releases and environmental transport factors.

Table A.2 Fraction of the 50-year dose commitment to the population within 50 miles of each fuel cycle facility from gaseous releases per year of operation of the facility

Facility	Fraction of 50 mile population dose commitment				References
	Immersion	Contaminated ground surface	Inhalation	Ingestion	
Mill ^a	<1.0E-3	2.0E-3	4.9E-2	9.5E-1	3
Mill ^a	<1.0E-3	1.1E-1	1.5E-1	7.4E-1	4
Fuel fabri- cation					
UO ₂	<1.0E-3	3.2E-1	1.9E-1	4.9E-1	3
Mixed oxide	<1.0E-3	1.0E-2	9.8E-1	8.0E-3	3
Fuel fabri- cation	<1.0E-3	7.4E-1	2.1E-1	4.4E-2	4
Reprocessing ^a	2.2E-1	1.2E-1	5.7E-1	9.9E-2	3
Reprocessing ^a	3.7E-2	2.2E-1	1.3E-1	6.2E-1	4

^aBoth references 3 and 4 considered both a mill and a reprocessing plant. The results differ because of differences in source term and assumptions about the facility.

A.2 Aquatic Releases

We are aware of no paper examining the importance of various aquatic pathways which is directly comparable to the paper by Hoffman and Kaye.¹ An adjoining paper by Soldat² in the same volume discusses "only the potential aquatic exposure pathways, including drinking water, aquatic foods, swimming and boating, contaminated sediments, and crops and animal products from irrigated farms." Soldat² goes on to say that the importance of the various pathways cannot be evaluated without knowing the release rates and physicochemical form of the radionuclides, the type of water receiving effluent and its use by humans.

Given these precautions, Soldat² discusses several studies of population doses from aquatic releases and attempts to compare the importance of the various pathways. Results from the studies cited by Soldat² and two additional studies are summarized in Table A.3. Different assumptions were used in each of these so care must be exercised when comparing them.

However, these four studies do reaffirm the point made in the atmospheric section that the importance of the pathway varies greatly depending on the organ of concern, the type of release, and the assumptions made when calculating doses, as well as the relationship between the calculated dose and a dose limit. The only apparent exception to this rule is the swimming pathway which, for the studies referenced, never contributed more than 3% of the total calculated dose.

Aside from the constant small contribution of the swimming pathway, the relative importance of the various pathways changes frequently due to

Table A.3 Fraction of the calculated total dose from one or more radionuclides listed by aquatic pathway in several studies

Organ	Radionuclide	Source of release	Fraction of total dose contribution from radionuclide given by				Reference
			Drinking water	Fish ingestion	Irrigation	Swimming	
Total body	Mixed	LWR	0.54	0.46	NC ²	NC ²	2
	Mixed	LWR	0.27	0.54	0.19	NC	2
	Mixed	LMFBR	0.89	0.03	0.06	0.03	4
	³ H	Waste burial	0.71	0.01	0.27		8
	⁹⁰ Sr	Waste burial	0.07		0.93		8
	¹⁰⁶ Ru	Waste burial	0.62	0.11	0.27		8
	¹³⁷ Cs	Waste burial	0.11	0.75	0.14		8
Thyroid	Mixed	LWR	0.97	0.03	NC	NC	2
	Mixed	LWR	0.63	0.13	0.25	NC	2
	³ H	Waste burial	0.71	0.01	0.27		8
	⁹⁰ Sr	Waste burial	0.92	0.08			8
	¹⁰⁶ Ru	Waste burial	0.22	0.04	0.75		8
	¹³⁷ Cs	Waste burial	0.13	0.87			8
Bone	Mixed	LWR	0.52	0.48	NC	NC	2
	Mixed	LWR	0.03	0.71	0.21	NC	2
	Mixed	LMFBR	0.97		0.03		4
	³ H	Waste burial	0.71	0.01	0.27		8
	⁹⁰ Sr	Waste burial	0.45	0.04	0.52		8
	¹⁰⁶ Ru	Waste burial	0.62	0.10	0.28		8
	¹³⁷ Cs	Waste burial	0.11	0.80	0.09		8

¹No calculation (NC) pathway not considered.

²Robbins and Martin⁶ as cited by Soldat.²

³Schuckler, Kalckbrenner, and Bayer⁷ as cited by Soldat.²

⁴Contribution of this pathway is less than 1% of the total

changes in the assumptions used in the calculations. The drinking water pathway contributes from 8 to 97% of the total dose. The fish ingestion pathway varies from negligible (less than 1%) to as much as 87%. Finally, the irrigation pathway contributed from less than 1% to as much as 93%. It is important to remember that the range of contributions of each pathway may not be completely specified in Table A.3; site specific considerations for almost any parameter or variable in the dose equations may significantly alter the contribution to dose of any given parameter.

Likely the most reasonable conclusion of this short review is that no single aquatic pathway to man dominates the dose calculations. Rather, the only pathway which is not very important, primarily as a result of low exposure possibility, is the swimming pathway. The only time this pathway would likely become important is if the other pathways were nonexistent at the site being considered.

A.3 Relative Contribution of Atmospheric vs Aquatic Releases

In general, an examination of the references cited in this appendix as well as other reference sources indicates that, for both individual and population doses, atmospheric releases from nuclear facilities are expected to contribute a larger fraction of the dose to humans than are aquatic releases. One reason for this expectation is that the source term for atmospheric releases is generally higher than the source term for aquatic releases. Also, humans tend to have more interaction with airborne than with waterborne radionuclides. However, variations between sites can be large enough that conditions prevailing at a specific site

should be considered when attempting to determine the relative importance of aquatic or atmospheric exposures in the calculation of a specific maximally exposed individual or population dose.

REFERENCES

1. Hoffman, F. O. and S. V. Kaye. 1976. Terrestrial exposure pathways: potential exposures of man from the environmental transport of waste nuclides. pp. 524-538. IN *Proceedings of the International Symposium on the Management of Wastes from the LWR Fuel Cycle*. Denver, Colorado, July 11-16, 1976. CONF-76-0701.
2. Soldat, J. K. 1976. Aquatic exposure pathways—potential exposure of man from environmental transport of waste radionuclides. pp. 539-553 IN *Proceedings of the International Symposium on the Management of Wastes from the LWR Fuel Cycle*. Denver, Colorado, July 11-16, 1976. CONF-76-0701.
3. Witherspoon, J. P. 1974. Population exposure estimates as derived from an environmental assessment of LWR fuel cycle facilities. Part I: Population doses from operating facilities. pp. 191-198. IN *Symposium on Population Exposures, Proceedings of the Eight Mid-year Topical Symposium of the Health Physics Society*. Knoxville, Tennessee, October 21-24, 1974. CONF-741018.
4. Rohwer, P. S., D. C. Parzyck, and S. V. Kaye. 1976. An application of the cumulative exposure index (CUEx) as an operational radiological protection guide. pp. 22-30. IN *Proceedings of the Ninth Midyear Topical Symposium of the Health Physics Society on Operational Health Physics*. Denver, Colorado, February 9-12, 1976.
5. U.S. Atomic Energy Commission. 1974. *Environmental Statement, Liquid Metal Fast Breeder Reactor Program*. Vol. II. WASH-1535.

6. Robbins C. and J. A. Martin, Jr. 1976. Population doses from liquid effluents into the Mississippi Basin. IN *Twenty-First Annual Meeting of the Health Physics Society*. San Francisco, California, June 27-July 2, 1976. (abstract).
7. Schuckler, M., R. Kalckbrenner, and A. Bayer. 1976. *Zukünftige radiologische Belastung durch kerntechnische Anlagen im Einzugsgebiet des Oberrheins, Teil II, Belastung über den Wasserweg*. pp. 724-726. Institut für Neutronenphysik und Reaktortechnik, Kernforschungszentrum Karlsruhe.
8. Little, C. A. and S. J. Cotter. 1980. *Estimated Radiological Doses to the Maximum Individual and Downstream Populations from Releases of Tritium, Strontium-90, Ruthenium-106, and Cesium-137 from White Oak Dam*. ORNL/TM-7039.

Appendix B

EXAMPLE CALCULATIONS

Charles W. Miller

The body of this report presents models and parameter values which may be used to assess the impact on man of radionuclides released to the environment by breeder reactor and other nuclear facilities. The purpose of this appendix is to illustrate how these models and parameter values are used in numerical calculations. Example problems are solved for environmental concentrations and subsequent doses to humans as a result of releases to the atmosphere and surface waters for a limited number of nuclides. While the examples considered are far from exhaustive when compared to those that might be encountered in a real assessment situation, it is hoped that these cases will be of assistance to persons attempting such calculations for the first time or to those who simply need to understand how such calculations are made. Also, these problems could be used as a partial check of computer implementations of material contained in this report.

When solving these example problems, reference is made to the location in the main body of the report where the equation, methodology, or parameter value being used is originally presented. The reader should consult the referenced locations for a complete definition of all of the terms used in solving the sample problem.

In the main body of this report, information is presented on the statistical distribution associated with the parameters whenever information on the distribution could be obtained. In this appendix, mean values of these parameters are used in the example calculations. It must be emphasized that the selection of the mean value was an arbitrary one for illustrative purposes only, and such use here does

not constitute a recommendation that mean values be used for all assessment calculations. In addition, the release rates used in these example problems were also chosen arbitrarily, and no relationship between them and actual release rates from breeder reactors or any other nuclear facilities was intended.

B.1 Environmental Concentrations

B.1.1 Releases to the Atmosphere

B.1.1.1 Air concentration from a ground-level release. Assume an annual release of 1×10^{-2} Ci of $^{131}\text{I}_2$ from a ground level ($H = 0$ m) source under Pasquill atmospheric stability category D. Because of the release height, the Pasquill-Gifford values of σ_y and σ_z should be used in the Gaussian plume model, Eq. (2.1). From Table 2.1, for a downwind distance of $x = 2000$ m,

$$\begin{aligned}\sigma_y &= (a_1 \ln x + a_2)x \\ &= (-0.0059[\ln 2000] + 0.11)(2000) \\ &= 130 \text{ m},\end{aligned}\tag{B.1}$$

and

$$\begin{aligned}\sigma_z &= \frac{1}{2.15} \exp(b_1 + b_2 \ln x + b_3 \ln^2 x) \\ &= \frac{1}{2.15} \exp(-1.35 + 0.793[\ln 2000] + 0.0022[\ln 2000]^2) \\ &= 57 \text{ m}.\end{aligned}\tag{B.2}$$

Equation (2.8) for plume depletion cannot be solved analytically. However, values of the ratio $\frac{Q'}{Q}$, the plume depletion fraction, for $v_d = 1$ cm/s and $u = 1$ m/s can be obtained from Fig. 5.5 of Meteorology

and Atomic Energy - 1968 (ref. 1). For a downwind distance of 2000 m, a release height of 0 m, and Pasquill D stability, Fig. 5.5 gives $\frac{Q'}{Q} = 0.35$. Assume that the actual wind speed for the $^{131}\text{I}_2$ release being considered here is 3 m/s. Further assume that $v_d(\text{total}) = 3.5 \text{ cm/s}$ (Sect. 2.2.1) and $v_w = 1.3 \text{ cm/s}$ (ref. 2). This results in a total deposition velocity of 4.8 cm/s. The value of $\frac{Q'}{Q}$ noted above can now be changed to reflect the assumed release conditions by

$$\begin{aligned} \left(\frac{Q'}{Q}\right)_2 &= \left(\frac{Q'}{Q}\right)_1 \frac{u_1 v_2}{u_2 v_1} \\ &= (0.35) \frac{(1 \text{ m/s})(4.8 \text{ cm/s})}{(3 \text{ m/s})(1 \text{ cm/s})} \\ &= 0.19 \end{aligned} \quad (\text{B.3})$$

This indicates that the deposition processes considered here result in an air concentration at 2000 m that is 19% of the air concentration that would be expected if these deposition processes were not considered.

The half-life, $t_{1/2}$, of ^{131}I is 8.05 days (ref. 3). Therefore, the radiological decay constant, λ , is $9.96 \times 10^{-7} \text{ s}^{-1}$.

Equation (2.1) may be used to calculate the ground-level, center-line air concentration. Accounting for decay and plume depletion, Eq. (2.1) becomes

$$x = \left[\frac{Q}{\pi u \sigma_y \sigma_z} \exp\left(-\lambda \frac{x}{u}\right) \right] \left(\frac{Q'}{Q}\right)_2 \quad (\text{B.4})$$

$$Q = 1 \times 10^{-2} \text{ Ci/year} = 3.2 \times 10^{-10} \text{ Ci/s}$$

$$\begin{aligned} x &= \left[\frac{(3.2 \times 10^{-10} \frac{\text{Ci}}{\text{s}})}{\pi(3 \text{ m/s})(130 \text{ m})(57 \text{ m})} \exp(-[9.96 \times 10^{-7}][2000/3]) \right] (0.19) \\ &= 8.7 \times 10^{-16} \text{ Ci/m}^3 \end{aligned}$$

The 22.5° sector average air concentration, based on Eq. (2.2), is given by

$$\begin{aligned} x &= \left[\frac{2.032 Q}{x u \sigma_z} \exp(-\lambda \frac{x}{u}) \right] \left(\frac{Q'}{Q} \right)_2 \quad (B.5) \\ &= \left[\frac{(2.032)(3.2 \times 10^{-10} \frac{\text{Ci}}{\text{s}})}{(2000 \text{ m})(3 \text{ m/s})(57 \text{ m})} \exp(-[9.96 \times 10^{-7}][2000/3]) \right] (0.19) \\ &= 3.6 \times 10^{-16} \frac{\text{Ci}}{\text{m}^3} \end{aligned}$$

B.1.1.2 Air concentration from an elevated release. Assume an annual release of 100 Ci of ^{85}Kr from a 100-m tall stack. Since Kr is a nonreactive gas and it has a half-life of 10.76 years (ref. 3), radiological decay and plume depletion need not be considered in this calculation. Using the Brookhaven dispersion parameters for this elevated release and assuming class B₁ stability and a downwind distance of 2 km, Table 2.2 gives

$$\begin{aligned} \sigma_z &= p_z x^{q_z} \quad (B.6) \\ &= (0.33)[(2000 \text{ m})^{0.86}] \\ &= 230 \text{ m} \end{aligned}$$

If $u = 4 \text{ m/s}$, the 22.5° sector-average air concentration for this release is, from Eq. (2.2),

$$\begin{aligned} x &= \frac{2.032Q}{xu\sigma_z} \exp \left\{ -1/2 \left(\frac{H}{\sigma_z} \right)^2 \right\} \quad (B.7) \\ &= \frac{(2.032)(3.2 \times 10^{-6} \frac{\text{Ci}}{\text{s}})}{(2000 \text{ m})(4 \text{ m/s})(230 \text{ m})} \exp \left\{ -1/2 \left(\frac{100 \text{ m}}{230 \text{ m}} \right)^2 \right\} \\ &= 3.2 \times 10^{-12} \text{ Ci/m}^3 \end{aligned}$$

B.1.1.3 Ground deposition. The rate of dry deposition onto the earth's surface is given by Eq. (2.4) as

$$d = xv_d \quad (B.8)$$

Using the information contained in Sect. B.1.1.1 for a ground-level atmospheric release of $^{131}\text{I}_2$,

$$\begin{aligned} v_d &= 3.5 \text{ cm/s} \\ x &= 3.6 \times 10^{-16} \text{ Ci/m}^3 \end{aligned}$$

and

$$\begin{aligned} d &= xv_d \\ &= (3.6 \times 10^{-16} \text{ Ci/m}^3)(3.5 \times 10^{-2} \text{ m/s}) \\ &= 1.3 \times 10^{-17} \text{ Ci/m}^2\text{-s} \\ &= 4.7 \times 10^{-2} \text{ pCi/m}^2\text{-h} \end{aligned}$$

B.1.1.3.1 Concentration on grass. The concentration in and on grass as a result of the deposition rate calculated above is given by Eq. (3.1) as

$$C_i^v = d_i \left\{ \frac{R[1 - \exp(-\lambda_{Ei} t_e)]}{Y_v \lambda_{Ei}} + \frac{B_{iv}[1 - \exp(-\lambda_i t_b)]}{P \lambda_i} \right\} \exp(-\lambda_i t_h) \quad (B.9)$$

The input parameter values for this case are

$$d_i = 4.7 \times 10^{-2} \text{ pCi/m}^2\text{-h (from above),}$$

$$\frac{R}{Y_v} = 2 \text{ m}^2/\text{kg (Table 3.1),}$$

$$\lambda_{Ei} = 6.4 \times 10^{-3} \text{ h}^{-1} \text{ (ref. 4) ,}$$

$$t_e = 720 \text{ h (Table 3.7),}$$

$$B_{iv} = 2.0 \times 10^{-1} \text{ (Table 3.2),}$$

$$\lambda_i = 3.6 \times 10^{-3} \text{ h}^{-1} \text{ (ref. 3),}$$

$$P = 215 \text{ kg/m}^2 \text{ (Table 3.6),}$$

$$t_b = 8.76 \times 10^3 \text{ h (assumed),}$$

$$t_h = 0 \text{ (Table 3.7) .}$$

The resulting concentration of ^{131}I in or on the grass is

$$C_i^v = 15 \text{ pCi/kg .}$$

B.1.1.3.2 Concentration on fresh produce. Equation (B.9) can also be used to estimate the concentrations in and on fresh produce as a result of the deposition of ^{131}I considered above. Assuming this fresh produce is leafy vegetables, the input parameter values for this case are

$$d_i = 4.7 \times 10^{-2} \text{ pCi/m}^2\text{-h (from above),}$$

$$R = 0.2 \text{ (Table 3.1),}$$

$$Y_v = 1.9 \text{ kg/m}^2 \text{ (Table 3.1),}$$

$$\lambda_{Ei} = 6.4 \times 10^{-3} \text{ h}^{-1} \text{ (ref. 4),}$$

$$\begin{aligned}
 t_e &= 1440 \text{ h (Table 3.7),} \\
 B_{iv} &= 5.5 \times 10^{-2} \text{ (Table 3.3),} \\
 \lambda_i &= 3.6 \times 10^{-3} \text{ h}^{-1} \text{ (ref. 3),} \\
 P &= 215 \text{ kg/m}^2 \text{ (Table 3.6),} \\
 t_b &= 8.76 \times 10^3 \text{ h (assumed),} \\
 t_h &= 336 \text{ h (Table 3.7).}
 \end{aligned}$$

Entering these values into Eq. (B.9) results in a concentration of ^{131}I in or on leafy vegetables of

$$C_i^v = 2.3 \times 10^{-1} \text{ pCi/kg}$$

B.1.1.3.3 Concentration in milk. Assume that both beef and dairy cattle are getting 100% of their food from the grass considered in Sect. B.1.1.3.1. The resulting concentration of ^{131}I in milk produced by these dairy cattle is, from Eq. (3.2),

$$C_i^M = F_m C_i^v Q_F \exp(-\lambda_i t_f) \quad (\text{B.10})$$

The input parameter values are

$$\begin{aligned}
 F_m &= 1.0 \times 10^{-2} \text{ day/liter (Table 3.4),} \\
 C_i^v &= 15 \text{ pCi/kg (see above),} \\
 Q_F &= 15.6 \text{ kg/day (Table 3.6),} \\
 t_f &= 4 \text{ days (Table 3.7),} \\
 \lambda_i &= 8.6 \times 10^{-2} \text{ day}^{-1} \text{ (ref. 3),}
 \end{aligned}$$

resulting in

$$C_i^M = 1.6 \text{ pCi/liter.}$$

B.1.1.3.4 Concentration in beef. Similarly, the concentration of ^{131}I in beef may be found using Eq. (3.4),

$$C_i^F = F_f C_i^V Q_F \exp(-\lambda_i t_s) \quad (\text{B.11})$$

and the new input values of

$$F_f = 7 \times 10^{-3} \text{ day/kg (Table 3.5),}$$

and

$$t_s = 20 \text{ days (Table 3.7).}$$

This calculation results in

$$C_i^F = 2.9 \times 10^{-1} \text{ pCi/kg.}$$

B.1.2 Release to Surface Water

B.1.2.1 Surface water concentrations. The fundamental form of the model for surface water concentration without sorption is given by Eq. (4.1). This model cannot be solved in general, however, without resorting to computerized numerical techniques. If, however, we assume a vertical line source emitting a constant W Ci/s, a closed form solution of Eq. (4.2) can be obtained (ref. 5). If we further assume a uniform, straight, rectangular channel in which the water flows, the solution to Eq. (4.2) is given by (ref. 5)

$$C_i = \frac{W}{Q} \left[1 + 2 \sum_{n=1}^{\infty} \exp \left(- \frac{n^2 \pi^2 x k_y}{u y^2} \right) \left\{ \cos^2 \left(\frac{n \pi b}{y} \right) \right\} \right] \quad (\text{B.12})$$

where

Q = total river flow (m^3/s),

b = cross-stream location of sampling point (m), and

the other parameters are as defined for Eq. (4.2). Using flow parameters based on Hudson River data (ref. 6) and a discharge rate of 1 Ci/s of ^{137}Cs , the input parameter values are

$$W = 1 \times 10^{-6} \text{ Ci/s},$$

$$Q = 1600 \text{ m}^3/\text{s},$$

$$K_y = 5 \text{ m}^2/\text{s},$$

$$u = 0.6 \text{ m/s},$$

$$y = 600 \text{ m},$$

$$x = 2000 \text{ m},$$

$$b = 200 \text{ m}.$$

Considering only the first three terms of the series expansion,

$$\begin{aligned} C_i &= \frac{(1 \times 10^{-6} \text{ Ci/s})}{(1600 \text{ m}^3/\text{s})} \left[1 + 2 \sum_{n=1}^3 \exp \left(- \frac{n^2 \pi^2 [2000 \text{ m}] \left[\frac{5 \text{ m}}{\text{s}} \right]^2}{[0.6 \text{ m/s}] [600 \text{ m}]^2} \right) \right. \\ &\quad \left. \cdot \left\{ \cos^2 \left(\frac{n \pi [200 \text{ m}]}{[600 \text{ m}]} \right) \right\} \right] \\ &= (6.2 \times 10^{-10}) [1 + 2(0.16 + 0.04 + 0.02)] \\ &= 9 \times 10^{-10} \text{ Ci/m}^3 \\ &= 9 \times 10^{-13} \text{ Ci/liter} \end{aligned}$$

B.1.2.2 Concentration in fish. Assume that freshwater finfish inhabit the stream considered above. The concentration CF_i (Ci/kg) of ^{137}Cs in these fish is given by

$$CF_i = B_{ip} C_i \quad (\text{B.13})$$

If we further assume that no site-specific information is available on the concentration of K in this stream, a mean value of B_{ip} can be obtained from Table 5.1. This results in

$$\begin{aligned} CF_i &= (1900 \text{ liter/kg})(9.1 \times 10^{-13} \text{ Ci/liter}) \\ &= 1.7 \times 10^{-9} \text{ Ci/kg} \end{aligned}$$

B.2 Dose Calculations

B.2.1 External Doses

B.2.1.1 Immersion in contaminated air. The external dose rate to an individual immersed in contaminated air is given by (ref. 7)

$$R_{ij}^{ea} = D_{ij} x_i \quad (\text{B.14})$$

where

R_{ij}^{ea} = dose rate to organ j due to immersion in air contaminated with radionuclide i (millirem/year),

D_{ij}^{ea} = dose conversion factor due to immersion in air for radionuclide i and organ j (millirem/year per $\mu\text{Ci}/\text{cm}^3$),

x_i^a = concentration of radionuclide i in air ($\mu\text{Ci}/\text{cm}^3$).

From Sect. B.1.1.1 above, a 22.5° sector-averaged air concentration for a ground-level release of $^{131}\text{I}_2$ was found to be

$$\begin{aligned} x_j^a &= 3.6 \times 10^{-16} \text{ Ci/m}^3 \\ &= 3.6 \times 10^{-16} \text{ } \mu\text{Ci/cm}^3 \end{aligned}$$

From Table 7.2 the dose conversion factor for the β dose rate to the body surface from $^{131}\text{I}_2$ is found to be

$$D_{ij}^{ea} = 1.71 \times 10^9 \text{ millirem/year per } \mu\text{Ci/cm}^3$$

Therefore, for this release the annual β dose rate to the body surface is

$$\begin{aligned} R_{ij}^{ea} &= (1.71 \times 10^9 \frac{\text{millirem}}{\text{year}} \frac{\text{cm}^3}{\mu\text{Ci}}) (3.6 \times 10^{-16} \frac{\mu\text{Ci}}{\text{cm}^3}) \\ &= 6.2 \times 10^{-7} \text{ millirem/year} \end{aligned}$$

Similar calculations can be performed for other organs listed in Table 7.2. The resulting γ and X-ray dose rates for all organs considered are as follows:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Body surface	6.2×10^{-7}
Lungs	7×10^{-7}
Ovaries	3.3×10^{-7}
Skeleton	9.8×10^{-7}
Testes	8.9×10^{-7}
Total body	7.6×10^{-7}

Section B.1.1.2 gives an air concentration for an elevated release of ^{85}Kr ,

$$\begin{aligned} \chi_i^a &= 3.2 \times 10^{-12} \text{ Ci/m}^3 \\ &= 3.2 \times 10^{-12} \text{ } \mu\text{Ci/cm}^3 \end{aligned}$$

Using Eq. (B.14) and Table 7.2, the annual β and photon dose rates for this release condition are found to be the following:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Body surface	7.2×10^{-3}
Lungs	3.7×10^{-5}
Ovaries	1.5×10^{-5}
Skeleton	4.8×10^{-5}
Testes	1.4×10^{-5}
Total body	3.9×10^{-5}

B.2.1.2 Exposure to contaminated ground. The external dose rate to an individual as a result of exposure to a radionuclide deposited on the ground is found from (ref. 7)

$$R_{ij}^{cg} = d_i \frac{1 - \exp(-\lambda_T t)}{\lambda_T} D_{ij}^{cg} (8.64 \times 10^4), \quad (\text{B.15})$$

where

R_{ij}^{cg} = dose rate to organ j due to exposure to ground contaminated with radionuclide i (millirem/year),

d_i = rate of deposition onto ground of radionuclide i ($\mu\text{Ci/cm}^2 \cdot \text{s}$),

λ_T = radioactive decay constant λ_r + environmental decay constant λ_w (day^{-1}),

t = time allotted for surface buildup (days),

D_{ij}^{cg} = dose conversion factor for radionuclide i and organ j due to surface exposure to an infinite plane at a point 1 m above ground (millirem/year per $\mu\text{Ci}/\text{cm}^2$),

$$8.64 \times 10^4 = \text{s/day.}$$

Section B.1.1.3 gives a dry deposition rate onto the earth's surface resulting from a ground-level release of $^{131}\text{I}_2$,

$$\begin{aligned} d_i &= 1.3 \times 10^{-17} \text{ Ci/m}^2\text{-s} \\ &= 1.3 \times 10^{-15} \mu\text{Ci/cm}^2\text{-s} \end{aligned}$$

If a one year surface buildup time and no losses from the surface due to weathering effects are assumed, then

$$\begin{aligned} \lambda_T &= \lambda_r = 8.61 \times 10^{-2} \text{ d}^{-1} \\ t &= 365 \text{ d} \end{aligned}$$

and

$$\begin{aligned} R_{ij}^{cg} &= (1.3 \times 10^{-15} \frac{\mu\text{Ci}}{\text{cm}^2\text{-s}}) \frac{(1 - \exp[-\{8.61 \times 10^{-2} \text{ d}^{-1}\} \{365 \text{ d}\}])}{(8.61 \times 10^{-2} \text{ d}^{-1})} \\ &\quad \cdot (8.64 \times 10^4 \frac{\text{s}}{\text{d}}) D_{ij}^{cg} \\ &= (1.3 \times 10^{-9} \frac{\mu\text{Ci}}{\text{cm}^2}) D_{ij}^{cg} \end{aligned}$$

Values of D_{ij}^{cg} for various organs may be found in Table 7.3 For the β dose rate to the body surface from $^{131}\text{I}_2$,

$$D_{ij}^{cg} = 3.25 \times 10^5 \frac{\text{millirem}}{\text{year}} \cdot \frac{\text{cm}^2}{\mu\text{Ci}}$$

Therefore, for this tissue,

$$R_{ij}^{cg} = (1.3 \times 10^{-9} \frac{\mu Ci}{cm^2})(3.25 \times 10^5 \frac{\text{millirem}}{\text{year}} \cdot \frac{cm^2}{\mu Ci})$$

$$= 4.2 \times 10^{-4} \text{ millirem/year.}$$

Similar calculations for the remaining organs listed in Table 7.3 complete the following table of dose rates:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Body surface	4.2×10^{-4}
Lungs	5.4×10^{-4}
Ovaries	2.5×10^{-4}
Skeleton	7.5×10^{-4}
Testes	6.8×10^{-4}
Total body	5.8×10^{-4}

B.2.2 Internal Doses

B.2.2.1 Inhalation dose. The dose rate to various organs of the body as a result of inhaling contaminated air is given by (ref. 7)

$$R_{ij}^{br} = x_i^a B_r D_{ij}^{br} \quad , \quad (B.16)$$

where

R_{ij}^{br} = dose rate to organ j due to breathing air contaminated with radionuclide i (rem/year),

x_i^a = concentration of radionuclide i in air ($\frac{\mu Ci}{m^3}$),

B_r = breathing rate ($\frac{m^3}{\text{year}}$),

D_{ij}^{br} = dose conversion factor for radionuclide i and organ j due to breathing of contaminated air (rem/ μCi intake).

From Table 6.1,

$$B_r = 8.03 \times 10^3 \text{ m}^3/\text{year}.$$

From Sect. B.1.1.1 for $^{131}\text{I}_2$,

$$\begin{aligned} x_i^a &= 3.6 \times 10^{-16} \text{ Ci/m}^3 \\ &= 3.6 \times 10^{-10} \text{ } \mu\text{Ci/m}^3. \end{aligned}$$

From Table 7.1 for the thyroid,

$$D_{ij}^{br} = 1.13 \frac{\text{rem}}{\mu\text{Ci}}.$$

The resulting annual dose rate to the thyroid for this release of $^{131}\text{I}_2$ is then

$$\begin{aligned} R_{ij}^{br} &= (3.6 \times 10^{-10} \frac{\mu\text{Ci}}{\text{m}^3})(8.03 \times 10^3 \frac{\text{m}^3}{\text{year}})(1.13 \frac{\text{rem}}{\mu\text{Ci}}) \\ &= 3.3 \times 10^{-6} \text{ rem/year} \\ &= 3.3 \times 10^{-3} \text{ millirem/year.} \end{aligned}$$

The doses to the remaining organs listed in Table 7.1 are also estimated from the application of Eq. (B.16) and included in the following:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Thyroid	3.3×10^{-3}
Lungs	6.9×10^{-6}
Total body	1.8×10^{-6}
Ovaries	1.2×10^{-7}
Total endosteal cells	6.4×10^{-7}
Testes	6.4×10^{-8}

Note that inhalation calculations are not made for ^{85}Kr since it is assumed that the inhalation doses from noble gases are insignificant when compared to air immersion doses.

B.2.2.2 Ingestion doses. The calculation of dose to humans as a result of ingestion of contaminated food is analogous to the calculation of inhalation doses as expressed by Eq. (B.16) (ref. 7),

$$R_{ij}^f = C_i U_i^f D_{ij}^f \quad (B.17)$$

where

R_{ij}^f = dose rate to organ j as a result of eating food contaminated with radionuclide i (rem/year),

C_i = concentration of radionuclide i in the food of interest ($\mu\text{Ci/kg}$ or $\mu\text{Ci/liter}$),

U_i^f = rate of intake of the food of interest containing radionuclide i (kg/year or liter/year),

D_{ij}^f = dose conversion factor for radionuclide i and organ j due to ingestion of contaminated food ($\text{rem}/\mu\text{Ci intake}$).

B.2.2.2.1 Ingestion of fresh produce. The concentration of ^{131}I in leafy vegetables from the ground-level release of $^{131}\text{I}_2$ being considered in this appendix was found in Sect. B.1.1.3.2 to be

$$\begin{aligned} C_i &= C_i^v = 2.3 \times 10^{-1} \text{ pCi/kg} \\ &= 2.3 \times 10^{-7} \text{ } \mu\text{Ci/kg.} \end{aligned}$$

Table 6.1 gives an annual average adult intake rate for leafy vegetables of

$$U_i^f = 18 \text{ kg/year ,}$$

assuming all of the leafy vegetables a person consumes are grown in the area effected by the release being considered. From Table 7.1 the ingestion dose conversion for $i = {}^{131}\text{I}$ and $j = \text{thyroid}$ is found to be

$$D_{ij}^f = 1.81 \text{ rem}/\mu\text{Ci}$$

It follows from Eq. (B.17) that the dose rate to the thyroid in this example is

$$\begin{aligned} R_{ij}^f &= (2.3 \times 10^{-7} \mu\text{Ci}/\text{kg})(18 \frac{\text{kg}}{\text{year}})(1.81 \frac{\text{rem}}{\mu\text{Ci}}) \\ &= 7.5 \times 10^{-6} \text{ rem/year} \\ &= 7.5 \times 10^{-3} \text{ millirem/year.} \end{aligned}$$

The dose rates to other organs from the ingestion of ${}^{131}\text{I}$ in leafy vegetables are also calculated from Eq. (B.17):

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Thyroid	7.5×10^{-3}
Lungs	1.2×10^{-6}
Total body	3.7×10^{-6}
Ovaries	2.5×10^{-7}
Total endosteal cells	1.4×10^{-6}
Testes	1.5×10^{-7}

B.2.2.2.2 Ingestion of milk. From Sect. B.1.1.3.3 the concentration of ${}^{131}\text{I}$ in milk for the release being considered here is

$$\begin{aligned} C_i &= C_i^m = 1.6 \text{ pCi/liter} \\ &= 1.6 \times 10^{-6} \mu\text{Ci/liter} \end{aligned}$$

Once again assuming all of the milk a person consumes comes from the area surrounding the release being considered, Table 6.1 gives an annual average adult intake rate for milk

$$U_i^f = 112 \text{ liter/year.}$$

The appropriate dose conversion factors may again be found in Table 7.1.

For $i = {}^{131}\text{I}$ and $j = \text{thyroid}$, it is again found that

$$D_{ij}^f = 1.81 \text{ rem/}\mu\text{Ci}$$

Applying Eq. (B.17), the dose rate to the thyroid is

$$\begin{aligned} R_{ij}^f &= (1.6 \times 10^{-6} \mu\text{Ci/l})(112 \text{ liter/year})(1.81 \text{ rem/}\mu\text{Ci}) \\ &= 3.2 \times 10^{-4} \text{ rem/year} \\ &= 3.2 \times 10^{-1} \text{ millirem/year.} \end{aligned}$$

Similar calculations for the dose rate to other organs due to the ingestion of ${}^{131}\text{I}$ in milk are included in the following:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Thyroid	3.2×10^{-1}
Lungs	5.1×10^{-5}
Total body	1.6×10^{-4}
Ovaries	1.1×10^{-5}
Total endosteal cells	6.0×10^{-5}
Testes	6.2×10^{-6}

B.2.2.2.3 Ingestion of meat. The dose to various organs as a result of the ingestion of ${}^{131}\text{I}$ in meat may also be found using Eq. (B.17) and the following input parameters:

$$\begin{aligned} C_i &= C_i^f = 2.9 \times 10^{-1} \text{ pCi/kg (Sect. B.1.1.3.4),} \\ &= 2.9 \times 10^{-7} \mu\text{Ci/kg} \end{aligned}$$

$$U_i^f = 2 \text{ kg/year (Table 6.1),}$$

$$D_{ij}^f = \text{values for various organs found in Table 7.1.}$$

If we again assume that the person consumes only beef raised in the area influenced by the release under consideration, the dose rates that result are as follows:

<u>Organ</u>	<u>Dose rate, rem/year</u>
Lungs	2.6×10^{-7}
Total body	8.2×10^{-9}
Ovaries	5.7×10^{-10}
Total endosteal cells	3.1×10^{-9}
Testes	3.3×10^{-10}
Thyroid	1.7×10^{-5}

B.2.2.2.4 Ingestion of fish. Equation (B.17) may also be used to calculate the dose due to the ingestion of fish containing ^{137}Cs as calculated in Sect. B.1.2.2. From that section,

$$\begin{aligned} C_i &= CF_i = 1.7 \times 10^{-9} \text{ Ci/kg} \\ &= 1.7 \times 10^{-3} \text{ } \mu\text{Ci/kg} . \end{aligned}$$

Table 6.1 gives, for freshwater finfish,

$$U_i^f = 4.4 \text{ kg/year} .$$

The dose conversion factors for $i = ^{137}\text{Cs}$ are also found in Table 7.1.

For $j = \text{total body}$,

$$D_{ij}^f = 4.91 \times 10^{-2} .$$

Assuming all of the fish consumed by this person contains C_i , the resulting annual dose rate to the total body is

$$\begin{aligned} R_{ij}^f &= (1.7 \times 10^{-3} \frac{\mu\text{Ci}}{\text{kg}})(4.4 \frac{\text{kg}}{\text{year}})(4.91 \times 10^{-2} \frac{\text{rem}}{\mu\text{Ci}}) \\ &= 3.7 \times 10^{-4} \text{ rem/year} \\ &= 3.7 \times 10^{-1} \text{ millirem/year.} \end{aligned}$$

Similar calculations give the additional results included below for the other organs listed in Table 7.1:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Total body	3.7×10^{-1}
Lungs	1.5×10^{-1}
Ovaries	5.6×10^{-1}
Total endosteal cells	6.0×10^{-1}
Testes	5.0×10^{-1}

B.2.3 Tritium and Carbon-14 Dose Calculations

B.2.3.1 Tritium doses. Assume ^3H is being released from an elevated source under the meteorological and location conditions considered for the ^{85}Kr release in Sect. B.1.1.2. From this section,

$$\frac{\lambda}{Q} = 1.0 \times 10^{-6} \text{ s/m}^3$$

If we assume for this ^3H release that $Q = 1 \times 10^{-6} \text{ Ci/s}$, then the air concentration is

$$\begin{aligned} \chi &= 1.0 \times 10^{-12} \text{ Ci/m}^3 \\ &= 1.0 \times 10^{-6} \text{ pCi/cm}^3 \end{aligned}$$

The total ingestion dose from ^3H for food and water can be calculated using Eq. (8.1),

$$D_t = C_f \lambda + C_w \lambda \quad (B.18)$$

Using the values of C_f and C_w found in Sect. 8.1.2, the ^3H ingestion dose for the release considered here is

$$\begin{aligned} D_t &= (6.18 \frac{\text{rem-cm}^3}{\text{pCi-year}}) (1.0 \times 10^{-6} \frac{\text{pCi}}{\text{cm}^3}) \\ &\quad + (5.70 \frac{\text{rem-cm}^3}{\text{pCi-year}}) (1.0 \times 10^{-6} \frac{\text{pCi}}{\text{cm}^3}) \\ &= 1.2 \times 10^{-5} \text{ rem/year} \\ &= 1.2 \times 10^{-2} \text{ millirem/year.} \end{aligned}$$

The dose due to the inhalation and skin absorption of ^3H is found from Eq. (B.16). For the total endosteal cells,

$$\begin{aligned} x_i^a &= 1.0 \times 10^{-6} \text{ pCi/m}^3, \\ E_r &= 8.03 \times 10^3 \text{ m}^3/\text{year (Table 6.1),} \\ D_{ij}^{\text{br}} &= 9.85 \times 10^{-5} \text{ rem/pCi (Table 7.1),} \end{aligned}$$

and

$$\begin{aligned} R_{ij}^{\text{br}} &= (1.0 \times 10^{-6} \frac{\text{pCi}}{\text{m}^3}) (8.03 \times 10^3 \frac{\text{m}^3}{\text{year}}) (9.85 \times 10^{-5} \frac{\text{rem}}{\text{pCi}}) \\ &= 7.9 \times 10^{-7} \text{ rem/year} \\ &= 7.9 \times 10^{-4} \text{ millirem/year.} \end{aligned}$$

For all other organs listed in Table 7.1, $D_{ij}^{\text{br}} = 1.25 \times 10^{-4}$ resulting in $R_{ij}^{\text{br}} = 1.0 \times 10^{-3}$ millirem/year.

B.2.3.2 Carbon-14 doses. Assume that $^{14}\text{CO}_2$ is released with the ^3H considered in Sect. B.2.3.1. As a result,

$$\frac{\chi}{Q} = 1.0 \times 10^{-6} \text{ s/m}^3$$

for this release situation also. If $Q = 5.0 \times 10^{-7} \text{ Ci/s}$,

$$\begin{aligned}\chi &= 5.0 \times 10^{-13} \text{ Ci/m}^3 \\ &= 5.0 \times 10^{-7} \text{ pCi/cm}^3\end{aligned}$$

According to Sect. 8.2 the primary dose to man from ^{14}C is via the ingestion pathway. This dose rate for various organs is equal to the product of the concentration of ^{14}C in air and the appropriate dose conversion factor listed in Table 8.1. The resulting dose rates from the air concentration noted above are as follows:

<u>Organ</u>	<u>Dose rate, millirem/year</u>
Whole body	6.0×10^{-1}
Red marrow	1.0
Lungs	2.5×10^{-1}
Endosteal cells	9.0×10^{-1}
Stomach wall	3.8×10^{-1}
Lower large intestine wall	4.6×10^{-1}
Thyroid	2.7×10^{-1}
Liver	3.6×10^{-1}
kidneys	3.3×10^{-1}
Testes	2.2×10^{-1}
Ovaries	2.2×10^{-1}

REFERENCES

1. Van der Hoven, I. 1968. Deposition of particles and gases. pp. 202-8. IN Slade, D. (ed.), *Meteorology and Atomic Energy - 1968*. USAEC TID-24190.
2. Miller, C. W., and F. O. Hoffman. A critique of methods for estimating plume depletion and deposition of airborne radionuclides. IN *Proceedings of the 12th Annual Symposium of the German-Swiss Fachverband für Strahlenschutz*. Norderney, Federal Republic of Germany, October 2-6, 1978.
3. U.S. Department of Health, Education, and Welfare, Public Health Service. 1979. *Radiological Health Handbook*. Publication No. 2016.
4. Hoffman, F. O., and C. F. Baes III (eds.). 1979. *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. ORNL/NUREG/TM-282.
5. U.S. Nuclear Regulatory Commission. 1977. Regulatory Guide 1.113. *Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix I*.
6. Yeh, G. 1979. Personal communication.
7. Moore, R. E., C. F. Baes III, L. M. McDowell-Boyer, A. P. Watson, F. O. Hoffman, J. C. Pleasant, and C. W. Miller. 1979. *AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides*. ORNL-5532.