NUREG/CR-2612
(ORNL/TM-8099)

cy. 2

Variability in Dose Estimates Associated with the Food Chain Transport and Ingestion of Selected Radionuclides

Prepared by F. O. Hoffman, R. H. Gardner, K. F. Eckerman

Oak Ridge National Laboratory

Prepared for
U.S. Nuclear Regulatory Commission

GPO Printed copy price: $4.75

UNITED STATES
NUCLEAR REGULATORY COMMISSION
WASHINGTON, D. C. 20555

August 4, 1982

ERRATA SHEET

for

NUREG/CR-2612
ORNL/TM-8099

VARIABILITY IN DOSE ESTIMATES ASSOCIATED WITH THE FOOD CHAIN TRANSPORT AND INGESTION OF SELECTED RADIONUCLIDES

Prepared by

Oak Ridge National Laboratory

for the

U.S. Nuclear Regulatory Commission

Please make the following corrections. See attached List of Revisions for insertions and deletions.

DIVISION OF TECHNICAL INFORMATION
AND
DOCUMENT CONTROL

LIST OF REVISIONS

to

Variability in Dose Estimates Associated with the Food Chain Transport and Ingestion of Selected Radionuclides; NUREG/CR-2612; ORNL/TM-8099; by F. O. Hoffman, R. H. Gardner, and K. F. Eckerman

Please make the following changes in your copy of the above manuscript to conform with changes in input data used in the analysis.

Page vii, line 16: Change 24*th* to 21*st*.

Page 16, lines 2 and 8, column 10 of Table 1: Change 0.28 m^2/kg to 0.10 m^2/kg.

Page 17, line 3, column 10 of Table 1: Change footnote i to footnote h.

Page 18, lines 6 and 8, column 10 of Table 2: Change 4×10^{-3} to 1×10^{-2}.

Page 18, line 10, column 10 of Table 2: Change 1.5×10^{-2} to 4×10^{-2}.

Page 19, line 6, column 10 of Table 2: Change 4.4×10^{-3} d/kg to 4.0×10^{-3} d/kg.

Pages 24 and 25: Replace with new Tables 3 and 4 (enclosed).

Page 28, line 9, column 2 of Table 6: Change U_n to U_m.

Page 33, line 12: Change 99.9*th* to 99*th*.

Page 33, line 13: Change 99*th* to 95*th*.

Pages 34 and 35: Replace with new Tables 7 and 8 (enclosed).

Page 36: Replace with new page 36 (enclosed).

Page 47, line 6: Change the words "at least" to "about."

NUREG/CR-2612
ORNL/TM-8099

Variability in Dose Estimates Associated with the Food Chain Transport and Ingestion of Selected Radionuclides

Manuscript Completed: April 1982
Date Published: June 1982

Prepared by
F. O. Hoffman, R. H. Gardner, K. F. Eckerman

Oak Ridge National Laboratory
Oak Ridge, TN 37830

Prepared for
Division of Systems Integration
Office of Nuclear Reactor Regulation
U.S. Nuclear Regulatory Commission
Washington, D.C. 20555
NRC FIN B0748

CONTENTS

LIST OF FIGURES

LIST OF TABLES

ACKNOWLEDGEMENTS

We gratefully acknowledge the careful review of earlier drafts of this manuscript and the helpful advice offered to us by D. C. Kocher, C. F. Baes III, R. V. O'Neill, and G. G. Killough, of the Oak Ridge National Laboratory, and Y. C. Ng from Lawrence Livermore National Laboratory. Estimation of the variability of several key parameters was made possible through the efforts of C. F. Baes III and Y. C. Ng who provided us with prepublished analyses of literature data. Typing of the early drafts of this report was performed by Robin H. Smith. The final manuscript was typed by Wilma C. Minor.

VARIABILITY IN DOSE ESTIMATES ASSOCIATED WITH THE FOOD CHAIN TRANSPORT AND INGESTION OF SELECTED RADIONUCLIDES

F. O. Hoffman, R. H. Gardner, and K. F. Eckerman

ABSTRACT

Dose predictions for the ingestion of ^{90}Sr and ^{137}Cs, using aquatic and terrestrial food chain transport models similar to those in the Nuclear Regulatory Commission's Regulatory Guide 1.109, are evaluated through estimating the variability of model parameters and determining the effect of this variability on model output. The variability in the predicted dose equivalent is determined using analytical and numerical procedures. In addition, a detailed discussion is included on ^{90}Sr dosimetry. The overall estimates of uncertainty are most relevant to conditions where site-specific data is unavailable and when model structure and parameter estimates are unbiased.

Based on the comparisons performed in this report, it is concluded that the use of the generic default parameters in Regulatory Guide 1.109 will usually produce conservative dose estimates that exceed the 90*th* percentile of the predicted distribution of dose equivalents. An exception is the meat pathway for ^{137}Cs, in which use of generic default values results in a dose estimate at the 24*th* percentile. Among the terrestrial pathways of exposure, the non-leafy vegetable pathway is the most important for ^{90}Sr. For ^{90}Sr, the parameters for soil retention, soil-to-plant transfer, and internal dosimetry contribute most significantly to the variability in the predicted dose for the combined exposure to all terrestrial pathways. For ^{137}Cs, the meat transfer coefficient, the mass interception factor for pasture forage, and the ingestion dose factor are the most important parameters. The freshwater finfish bioaccumulation factor is the most important parameter for the dose prediction of ^{90}Sr and ^{137}Cs transported over the water-fish-man pathway.

The conservatism associated with the models and parameters in Regulatory Guide 1.109 is also evaluated for the ingestion of food containing ^{3}H and ^{14}C. Dose predictions to the total body using the Regulatory Guide are compared with upper-limit predictions estimated with specific activity models which assume complete equilibrium between the specific activity in the receptor and the atmosphere. For ^{3}H, the dose predictions of the Regulatory Guide are approximately 33% of an upper-limit estimate. For ^{14}C, the Regulatory Guide predictions are approximately 28% of the maximum.

1. INTRODUCTION

1.1 PURPOSE OF THE REPORT

Use of the models and parameter values provided in Regulatory Guide 1.109 (USNRC, 1977) for predicting doses to individuals from the food chain transport and ingestion of radionuclides is subject to some uncertainty as many of the recommended generic parameter values exhibit considerable variability (Hoffman and Baes, 1979). This report addresses the magnitude of the variability in the predicted dose equivalent due to variability associated with model parameters for the aquatic and terrestrial food chain transport and subsequent ingestion of ^{90}Sr and ^{137}Cs. The extent to which the generic parameter values in the Regulatory Guide result in over- or underestimations of dose equivalent is also evaluated under the assumption that the model structure is an appropriate representation of the real world and that estimates of parameter variability are relevant and unbiased.

Dose equivalents to maximally exposed adults, as the result of a continuous rate of deposition of ^{90}Sr and ^{137}Cs onto agricultural land, are estimated for the following exposure pathways:

deposition → pasture vegetation → meat → man,
deposition → pasture vegetation → milk → man,
deposition → leafy vegetables → man,
deposition → non-leafy vegetables → man.

The water-fish-man exposure pathway for aquatic releases of ^{90}Sr and ^{137}Cs is also considered. In addition, the potential conservatism associated with the Regulatory Guide 1.109 evaluation of dose from atmospheric releases of ^{3}H and ^{14}C is assessed through comparison with specific activity calculations which assume complete equilibrium.

This work represents an extension of the investigations initiated in NUREG/CR-1004 (Hoffman and Baes, 1979). Although a detailed examination of the data available for model parameters is not included in the present study, other sources, including NUREG/CR-1004, are consulted

which document the variability of these data. In certain circumstances, judgment was used to estimate parameter variability because documentation of the variability of parameter values was unavailable. The present study utilizes computerized numerical techniques as well as analytical approaches to estimate the overall influence of combined parameter variability on model output.

1.2 SOURCES OF UNCERTAINTY

The application of mathematical models for environmental radiological assessment results in dose estimates that are always subject to some uncertainty. This uncertainty arises from a number of sources, most of which are related to elements of the real world not accounted for in the mathematical structure of the model or in the quantification of model parameters (Schwarz, 1980; Shaeffer, 1980). The selection of a model structure that is not an exact representation of a particular assessment situation can result in dose predictions that are biased. In addition, predictive bias can occur through systematic errors in the selection of values for model parameters. For example, small amounts of conservatism applied to the selection of every input parameter value, in an otherwise unbiased radiological assessment model, can produce large amounts of conservatism in the final prediction of dose equivalent (Hoffman and Baes, 1979).

A second source of uncertainty is variability in model parameters. This source of uncertainty is especially pertinent when models, such as those employed within Regulatory Guide 1.109, use only a single value to quantify each model parameter and thus produce only a single prediction of dose for a given source term and exposed member of a specified population group. Literature values may range over several orders of magnitude for some model parameters (Hoffman and Baes, 1979; Baes et al., in preparation; Kocher et al., 1980; Ng, in press), although much of this variability may be explained by temporal and spatial environmental factors, differences in experimental techniques, and random experimental error. Nevertheless, some variability will persist even after site-specific experiments have been rigorously designed to obtain the most appropriate estimates of model parameters.

Considerations for these sources of variability are not explicitly included in Regulatory Guide 1.109. Therefore, differences must be expected between actual doses incurred by individuals residing in the vicinity of operating nuclear facilities and Regulatory Guide 1.109 predictions of dose. The expectation, however, is for model predictions to be biased in such a manner as to ensure that actual doses are not substantially underestimated (USNRC, 1975).

1.3 PROCEDURES FOR ESTIMATING PREDICTIVE UNCERTAINTY

The best procedure for estimating the uncertainties associated with model predictions is a process frequently referred to as "model validation" (Mankin et al., 1977; Schwarz, 1980; Shaeffer, 1980). Model validation involves testing predictions against a series of independent data representative of the range of conditions over which application of the model is intended. The primary objective is determining the potential magnitude of predictive error. The "validity" of a model is subsequently determined by deciding what constitutes an acceptable predictive error for a given assessment situation.

Because of the large amount of time and expense required to properly test the models in Regulatory Guide 1.109 under the full range of conditions in which these models may be applied, other methods for addressing predictive uncertainties must be employed. The methodology used in this study estimates a probability distribution of model predictions from estimated distributions of values for each input parameter. Uncertainties associated with the use of the generic parameters in Regulatory Guide 1.109 are estimated by comparing Regulatory Guide 1.109 dose predictions with the probability distribution of dose predictions derived from the combined variability of all model input parameters. This approach has been referred to as an "imprecision analysis" (Schwarz, 1980; Schwarz and Hoffman, 1981; O'Neill et al., 1981). The term "imprecision analysis" is preferred to "uncertainty anaysis" because no direct measure of the departure of model predictions from reality is possible without experimentally testing for the bias in model structure and parameter estimation.

The amount of bias inherent with the models and parameter values in Regulatory Guide 1.109, identified through the use of "imprecision analysis," is limited to those circumstances when the structure of the models and the estimates for the distributions of values for each model parameter are relevant and unbiased with respect to a given assessment situation. Nevertheless, the model structure and estimates of parameter distributions may exhibit considerable bias. The relevancy of the results obtained in this study must therefore await confirmation through experimental verification. This cautionary note is not restricted to this specific report. Caution should be exercised whenever interpreting the results of an analysis of any model that has not been subject to extensive validation tests.

2. METHODS FOR DETERMINING THE VARIABILITY IN DOSE ESTIMATION

2.1 SPECIFICATION OF PARAMETER VARIABILITY

No attempt is made in this report to analyze the variability of original data reported in the scientific literature. Information about parameter variability is obtained by consulting other sources, including the reviews of data in NUREG/CR-1004 (Hoffman and Baes, 1979) and similar reviews and analyses currently in progress at Oak Ridge National Laboratory (Baes et al., in preparation) and Lawrence Livermore National Laboratory (Ng et al., in preparation). Where available data are insufficient for specifying a distribution of values for a specific parameter, judgment is used to derive an approximate distribution from a limited amount of information. For example, if a parameter is expected to vary by less than one order of magnitude, minimum, maximum, and expected (most probable or median) values are used to define the limits and mode of a triangular distribution. If, on the other hand, parameters are expected to vary by more than one order of magnitude, a lognormal distribution is assumed. Lognormality is assumed as this distribution most frequently describes data for parameters exhibiting large variability (Hoffman and Baes, 1979; Shaeffer, 1980; Baes et al., in preparation).

The statistical properties of the lognormal distribution are derived by assuming that the maximum observed value approximates the 99th percentile of the distribution and assigning the expected value to the geometric mean. These procedures permit specification of a mean (μ) and variance (σ^2) of logtransformed data, (Hoffman and Baes, 1979) whereby:

$$\mu = \ln X_m, \text{ and} \tag{1}$$

$$\sigma^2 = \frac{\ln X_{99} - \mu}{2.33}, \tag{2}$$

where

X_m = the geometric mean,

X_{99} = the 99*th* cumulative percentile, and

2.33 = the number of standard deviations of a standard normal distribution related to the 99*th* cumulative percentile.

The geometric standard deviation (s_g) can be obtained directly from the variance of logtransformed data, where

$$s_g = \exp \sqrt{\sigma^2} \qquad (3)$$

$$= \exp \sigma \ .$$

For lognormal distributions, the geometric standard deviation is a factor that, when multiplied by the geometric mean to produce an upper value and when divided into the geometric mean to produce a lower value, defines an interval which includes 68% of the distribution.

2.2 PROPAGATION OF PARAMETER VARIABILITY

2.2.1 Analytical methods using lognormal statistics

The simplest procedure for estimating the variability in the prediction of dose equivalent is to reduce the structure of the model to a series of multiplicative parameters and use lognormal statistics. This is the procedure adopted in NUREG/CR-1004 (Hoffman and Baes, 1979) and applied in this study for the water-fish-man pathway. The water-fish-man pathway can be readily described by a series of multiplicative parameters:

$$R_{ij} = C_{iw} \, B_{ip} \, U_F \, D_{ij}, \qquad (4)$$

where

R_{ij} = the annual dose equivalent (mrem) for a specific radionuclide and organ j,

C_{iw} = the concentration of the radionuclide i in water (pCi/l),

B_{ip} = the radionuclide-specific bioaccumulation factor for fish (pCi/kg fish per pCi/l water),

U_F = the annual consumption of fish (kg/yr), and

D_{ij} = the dose equivalent conversion factor (mrem/pCi ingested) for a specific radionuclide i and organ j, or sensitive tissue j.

For such a model, if all input parameters can be described by lognormal distributions, the predicted dose equivalent will also be lognormal (Schubert, Brodsky, and Tyler, 1967; Hoffman and Baes, 1979; Shaeffer, 1980). The geometric mean and geometric standard deviation for the predicted distribution of dose equivalent can be derived from the sum of means and variances of logtransformed data for each model parameter:

$$X_m[R_{ij}] = \exp\left(\mu_{C_{iw}} + \mu_{B_{ip}} + \mu_{U_F} + \mu_{D_{ij}}\right), \text{ and} \tag{5}$$

$$s_g = \exp\left(\sigma^2_{C_{iw}} + \sigma^2_{B_{ip}} + \sigma^2_{U_F} + \sigma^2_{D_{ij}}\right)^{\frac{1}{2}}, \tag{6}$$

where

$X_m[R_{ij}]$ = the geometric mean of the lognormal distribution of the predicted dose equivalent,

s_g = the geometric standard deviation,

μ = the mean of logtransformed data, and

σ^2 = the variance of logtransformed data.

For a lognormal probability distribution, the cumulative percentile (C.P.) associated with any single value (X_a) within that distribution can be obtained as follows:

$$z_a = \frac{\ln X_a - \ln X_m}{\ln s_g} \tag{7}$$

Where z_a is the number of standard deviations by which $\ln X_a$ departs from the mean of a standard normal distribution. The value of z_a is directly related through the use of statistical tables (e.g., Neter et al., 1978) to a specific level of cumulative probability. For example, values of z_a of 1.0, 1.3, 1.6, 2.3, and 3.1 are related to the 84*th*, 90*th*, 95*th*, 99*th*, and 99.9*th* cumulative percentiles, respectively.

The procedure described above is based on the assumption that all parameters in the model are statistically independent and uncorrelated (Hoffman and Baes, 1979).

2.2.2 Numerical methods using Monte Carlo techniques

Numerical methods have been developed to investigate the combined effect of parameter variability on model predictions for models of more complex structure than simple multiplicative chains (McKay, Conover, and Beckman, 1979; Rubenstein, 1981; Iman, Conover, and Campbell, 1980; Gardner, O'Neill, and Hoffman, 1982). These methods select a random set of parameter values from prescribed statistical distributions and obtain a model solution for each parameter set. The results of numerous (500 to 10,000) iterations of model solutions are statistically summarized. The advantage of this procedure is that the variability of model predictions can be calculated based on any number of different theoretical or empirical distributions specified for model parameters. For further details, see O'Neill et al. (1981), Gardner, O'Neill, and Hoffman (1982), Matthies et al. (1981), and Schwarz and Hoffman (1981).

In this report, a numerical Monte Carlo approach (Carney et al., 1981) is applied to predict the distributions of food concentrations and dose equivalents resulting from the transport of ^{90}Sr and ^{137}Cs over the leafy vegetable, non-leafy vegetable, milk, and meat pathways. Direct deposition onto plant surfaces as well as uptake from soil are considered after an assumed continuous rate of deposition over a period of 15 years. Finally, the distribution of dose equivalents is predicted for the case where an individual could be exposed to all four terrestrial food pathways.

The conceptual structure and mathematical algorithms of the models are similar to those employed in Regulatory Guide 1.109 (USNRC, 1977), with minor modifications to account for the potential downward migration of ^{90}Sr and ^{137}Cs out of the assumed 15 cm soil root zone (see Section 3.1.2). Either a triangular, normal or lognormal distribution is specified for each model parameter. Statistical analysis of model predictions produced by 500 computer iterations was performed using the SAS system (Barr, Goodnight, and Sall, 1979).

2.2.3 Criteria for determination of parameter significance

Where lognormal statistics are used to estimate the predictive variability of simple multiplicative chain models, the importance of a given parameter to the predictive variability is indicated by the magnitude of the variance given for the logtransformed data for that parameter. The relative importance of the variability of a parameter to the predictions of the model can be determined by dividing the variance of the logtransformed parameter estimates by the variance of the logtransformed distribution of model predictions.

For more complex models, the effect of parameter variability on model predictions is analyzed using numerical techniques. Parameter significance is determined by correlating randomly selected parameter estimates with the resultant model predictions.

The relationship between parameter variability and model predictions is measured by the simple correlation coefficient, r (Snedecor and Cochran, 1967, Conover, 1971). This statistic indicates the degree of

relationship between two quantities. Values of r may range from -1.0 to 1.0. If r = 0 then no relationship between the two quantities exists. If r is either 1 or -1 there is a perfect positive or negative relationship, respectively. The percent relationship between two quantities (i.e., what percent of the variability in dose is explained by the variability in a particular model parameter) is expressed by the squared value of the correlation coefficient, r^2. In this report, we use r^2 as an index of the relative importance of each parameter with respect to the model prediction. We consider values of r less than 0.2 to be unimportant because they explain less than 4% of the variability of the predictions. We emphasize here that a correlation coefficient is a statistical measure and does not necessarily indicate cause and effect relationships.

3. ESTIMATION OF THE VARIABILITY IN THE PREDICTION OF THE ^{90}Sr BONE-SURFACE DOSE EQUIVALENT AND THE ^{137}Cs WHOLE-BODY DOSE EQUIVALENT DUE TO FOOD CHAIN TRANSPORT AND INGESTION

The radionuclides ^{90}Sr and ^{137}Cs have been selected for analysis because of their importance as fission products, their potential for both aquatic and terrestrial bioaccumulation, and the relative abundance of data that exists for nuclide specific model parameters. The following sections describe the specific models through which the effect of parameter variability is analyzed. The form of the distribution and the amount of variability estimated for each model parameter is also discussed. The final results are expressed in fifty-year committed dose equivalents predicted for an adult residing in the near vicinity of an operating nuclear facility.

3.1 DESCRIPTION OF MODELS FOR SPECIFIC EXPOSURE PATHWAYS

3.1.1 Aquatic foods

The primary pathway considered for the release of radionuclides into the aquatic environment is the transport of ^{90}Sr and ^{137}Cs from a given concentration in water into the edible portions of freshwater fish. Of interest is the variability associated with predictions of the internal dose resulting from the consumption of this fish by adult members of the human population. As discussed previously in Sect. 2.2.1, the mathematical structure of the model is a simple multiplicative chain of parameters [Eq. (4)].

The concentration in water (C_{iw}) is assumed to be a constant 1 pCi/l for both ^{90}Sr and ^{137}Cs. Therefore, the variability in the predicted dose equivalent will be a function of the variability assumed for the freshwater fish bioaccumulation factor (B_{ip}), the annual consumption of fish (U_F), and the dose conversion factor (D_{ij}). The values for the nuclide-specific and non-specific parameters employed in Eq. (4) are discussed in Sect. 3.2.

3.1.2 Terrestrial foods

The analyses for the milk, meat, leafy and non-leafy vegetable pathways assume a continuous rate of deposition (d) of 1 $pCi/m^2 \cdot d$. The conceptual structure and algorithms of the models used for these pathways are essentially identical with those in Regulatory Guide 1.109 (USNRC, 1977). An additional term (λ_{is}) is included to account for the downward migration of a deposited radionuclide out of the assumed 15 cm root zone of soil. Loss terms to account for radiological decay of the radioisotopes between harvest and human consumption of vegetation or animal food products are excluded. Because the radiological half-lives of ^{90}Sr and ^{137}Cs are on the order of 30 years, only a negligible amount of these nuclides is lost through radiological decay during the comparatively short time period between harvest and consumption of food products.

The basic equation used for predicting the transfer of ^{90}Sr and ^{137}Cs into vegetation is,

$$C_{iv} = d \frac{r\{1 - \exp - [(\lambda_w + \lambda_{ir})\ t_e]\}}{Y_v(\lambda_w + \lambda_{ir})} + \frac{B_{iv}\{1 - \exp - [(\lambda_{is} + \lambda_{ir})\ t_b]\}}{p(\lambda_{is} + \lambda_{ir})} \quad (8)$$

where

C_{iv} = the concentration of radionuclide i in vegetation (pCi/kg),

d = the deposition rate ($pCi/m^2 \cdot d$),

r = the fraction of the depositing radionuclide that is intercepted by standing vegetation (unitless),

Y_v = the standing crop biomass (kg/m^2),

λ_w = the rate constant for the environmental removal of surface deposited material from vegetation (d^{-1}),

λ_{ir} = the radiological decay constant (d^{-1}),

t_e = the time period vegetation is exposed to depositing radionuclides prior to harvest (d),

B_{iv} = the plant/soil concentration ratio (pCi/kg vegetation per pCi/kg soil),

p = the effective surface density of soil assuming a 15 cm root zone (kg/m^2),

λ_{is} = the rate constant describing the migration of a radionuclide out of the 15 cm root zone of soil (d^{-1}), and

t_b = the assumed midpoint of the lifetime of a light water reactor (d).

In this model, only the parameters λ_{ir}, B_{iv}, and λ_{is} are recognized as dependent on the radionuclide (or radioelement). The parameters r, Y_v, λ_w, t_e, and B_{iv} are assumed to be specific to the type of vegetation being considered: leafy vegetables, non-leafy vegetables, and pasture forage. Parameters specific to vegetation consumed directly by humans are referred to on a fresh or wet weight basis, while parameters specific to herbaceous forage are referred to on a dry weight basis to be consistent with data reported for human and animal diets. Furthermore, the parameters r and Y_v are combined into one single parameter, the mass interception fraction (r/Y_v), because the interception fraction is a function of the standing biomass of vegetation (Miller, 1980; Baes et al., in preparation).

The basic model structure for the transport of radionuclides from herbaceous pasture vegetation into milk and meat is,

$$C_{im} = C_{iv}\, Q_m\, F_{im}, \text{ and} \tag{9}$$

$$C_{if} = C_{iv}\, Q_f\, F_{if}, \tag{10}$$

where

C_{im} = the concentration of the radionuclide in milk (pCi/l),

C_{if} = the concentration of the radionuclide in meat (pCi/kg),

Q_m = the dry matter daily intake of feed derived from forage by dairy cows (kg/d),

Q_f = the dry matter daily intake of feed derived from forage by beef cattle (kg/d),

F_{im} = the equilibrium milk transfer coefficient that relates the concentration in milk at equilibrium to the average daily intake rate of the radionuclide by a dairy cow, (d/l),

F_{if} = the meat transfer coefficient that relates the concentration in meat at the time of slaughter to the average daily intake rate by beef cattle (d/kg), and

C_{iv} = the concentration in animal feed derived from pasture forage per unit dry mass vegetation (pCi/kg), calculated using Eq. (8).

The only nuclide-dependent parameters are F_{im} and F_{if}, in addition to C_{iv}. No differentiation is explicitly made between fresh forage and stored feed consumption because the long radiological half-lives of ^{90}Sr and ^{137}Cs would negate any significant decay during typical time periods assumed for the delay between harvest and consumption of stored feeds. However, an adjustment of the mean values of Q_m and Q_f is made to account for the likelihood that relatively uncontaminated concentrates and grains are imported from outside the immediate region (see Sect. 3.2).

For the consumption and dosimetry of contaminated foods by humans the model structure is the same for all four pathways

$$R_{ijp} = C_{ip}\ U_p\ D_{ij}, \qquad (11)$$

where

R_{ijp} = the annual dose equivalent to a specific organ j resulting from the transport of a radionuclide i over a specific pathway p (mrem/yr),

C_{ip} = the concentration of radionuclide i in the food of pathway p (pCi/kg or pCi/liter),

U_p = the annual rate of consumption of food by humans from pathway p (kg/yr or liter/yr), and

D_{ij} = the ingestion dose conversion factor for a specific nuclide (mrem/pCi).

For the case where all terrestrial food pathways are considered to contribute to the exposure of a given individual, the structure of the model becomes:

$$R_{ij} = D_{ij} \left(U_l C_{il} + U_n C_{in} + U_m C_{im} + U_f C_{if}\right), \qquad (12)$$

where

l = the pathway for leafy vegetables,

n = the pathway for non-leafy vegetables,

m = the milk pathway, and

f = the meat pathway.

3.2 VARIABILITY OF MODEL PARAMETERS

For each model parameter, the prescribed type of statistical distribution, as well as the mean and standard deviation are given in Tables 1 and 2. In addition, estimated minimum and maximum extreme values are included to indicate the level of truncation performed when

Table 1. Variability of nuclide-independent parameters

Pathway	Parameter	Distribution[a]	Mean value[b]	Standard deviation[c]	Level of truncation[d]		Reference	Notes*	NRC[e]
					Minimum	Maximum			
Water-Fish-Man [Eq. (4)]	U_F	L	14 kg/yr	(2.16)	---	---	Rupp et al. (1980)	A	21 kg/yr
Deposition-Leafy Vegetables-Man [(Eqs. (8) and (12)]	r/Y_v^f	L	0.1 m^2/kg	(1.82)	1×10^{-2} m^2/kg	1.0 m^2/kg	Baes et al. (in preparation	B,C	0.28 m^2/kg
	λ_w	L	5.7×10^{-2} d^{-1}	(1.68)	8.7×10^{-3} d^{-1}	3.5×10^{-1} d^{-1}	Miller and Hoffman (1981)	B,C	5×10^{-2} d^{-1}
	t_e	T	75 d	---	40 d	120 d	Shor (private communication)		60 d
	p	L	213 kg/m^2	(1.12)	100 kg/m^2	300 kg/m^2	Baes and Sharp (submitted)		240 kg/m^2
	t_b	C	5475 d	---	---	---			5475 d
	U_l	L	18 kg/yr	(1.62)	0 kg/yr	55 kg/yr	Rupp (1979, 1980)	B,C	64 kg/yr
Deposition-Non-leafy Vegetables-Man [Eqs. (8) and (12)]	r/Y_v^f	L	6.0×10^{-2} m^2/kg	(2.32)	1×10^{-3} m^2/kg	0.8 m^2/kg	Baes et al. (in preparation	B,C	0.28 m^2/kg
	λ_w	L	3.4×10^{-2} d^{-1}	(1.77)	4.6×10^{-3} d^{-1}	3.5×10^{-1} d^{-1}	Miller and Hoffman (1981)	B,C	5×10^{-2} d^{-1}
	t_e	T	100 d	---	60 d	180 d	Shor (private communication)		60 d
	p	L	213 kg/m^2	(1.12)	100 kg/m^2	300 kg/m^2	Baes and Sharp (submitted)		240 kg/m^2
	t_b	C	5475 d	---	---	---	USNRC (1977)		5455 d
	U_n	L	45 kg/yr	(2.16)	0 kg/yr	540 kg/yr	Rupp (1980)	B,C,D	520 kg/yr
Deposition-Pasture-Milk-Man [(Eqs. (8), (9), and (12)]	r/Y_v^g	L	1.8 m^2/kg	(1.55)	0.3 m^2/kg	10 m^2/kg	Miller (1979)		1.1 m^2/kg
	λ_w	L	5.7×10^{-2} d^{-1}	(1.45)	8.7×10^{-3} d^{-1}	3.5×10^{-1} d^{-1}	Miller and Hoffman (1981)		5×10^{-2} d^{-1}
	t_e	T	30 d	---	15 d	200 d		E	30 d
	p	L	213 kg/m^2	(1.12)	100 kg/m^2	300 kg/m^2	Baes and Sharp (submitted)		240 kg/m^2
	t_b	C	5475 d	---	---	---	USNRC (1977)		5475 d
	Q_m	N	11.0 kg/d	2.6 kg/d	4.0 kg/d	25 kg/d	Shor et al. (1982) Shor and Fields (1979)	B,F	12.5 kg/d
				(2.23)	0 L/yr	600 L/yr	Rupp (1979)	A	310 L/yr

Table 1. (continued)

Pathway	Parameter	Distribution[a]	Mean value[b]	Standard deviation[c]	Level of truncation[d]		Reference	Notes*	NRC[e]
					Minimum	Maximum			
Deposition-Pasture-Meat-Man [(Eqs. (8), (10), and (12)]	r/Y_v[g]	L	1.8 m^2/kg	(1.55)	0.3 m^2/kg	10 m^2/kg	Miller (1979)		1.1 m^2/kg
	λ_w	L	5.7×10^{-2} d^{-1}	(1.45)	8.7×10^{-3} d^{-1}	3.5×10^{-1} d^{-1}	Miller and Hoffman		5×10^{-2} d^{-1}
	t_e	T	40 d	---	15 d	200 d		E	40 d[h]
	p	L	213 kg/m^2	(1.12)	100 kg/m^2	300 kg/m^2	Baes and Sharp (submitted)		240 kg/m^2
	t_b	C	5475 d	---	---	---	USNRC (1977)		5475 d
	Q_f	N	8.3 kg/d	2.0 kg/d	1.6 kg/d	18.0 kg/d	Shor et al. (1979) Shor et al. (1982)	B,F	12.5 kg/d
	U_f	L	94 kg/yr	(1.65)	0 kg/yr	300 kg/yr	Rupp (1979, 1980)	B	110 kg/yr

[a] Probability distributions, where L = lognormal, T = triangular, N = normal, and C = constant.

[b] For lognormal distributions, the "mean value" is the geometric mean; for triangular distributions, the "mean value" is the mode.

[c] For lognormal distributions, the "standard deviation" refers to the geometric standard deviation s_g (in parenthesis).

[d] Assumed extreme values for truncating the random selection of parameter values using the Monte Carlo approach for error propagation.

[e] Generic default values recommended in NRC Regulatory Guide 1.109 (USNRC, 1977).

[f] Fresh weight.

[g] Dry weight.

[h] Assumed in the absence of a specific value in Regulatory Guide 1.109.

*Explanation of Notes:

A Values derived from a percent frequency distribution in the referenced report; U_F is derived from a survey of 77 Lake Michigan sport fishermen; U_m is based on a daily recall survey of 1,980 milk consuming individuals.

B Cited references consulted along with the use of judgment to estimate an expected value and probable range of values.

C Judgment applied in determining values considered appropriate for this study, variability estimated by relating the maximum observed value to the 99*th* percentile of a lognormal distribution.

D Mean value estimated assuming 75% of the total non-leafy vegetable diet is obtained outside the region of contamination; 99*th* percentile value is assumed to be three times the average with 100% of the total non-leafy vegetable diet being derived from the site of residence.

E Range estimated assuming rotational grazing for establishing a minimum value and unrestricted grazing for establishing a maximum value.

F Dry-matter concentrates and grains assumed to be obtained from outside the region of contamination for estimation of the mean value. Maximum extreme value is equivalent to maximum reported value and assumes a minimum intake of concentrates and grains.

G 99*th* percentile assumed to be three times the average; meat consumption includes poultry.

Table 2. Variability of nuclide-dependent parameters

Pathway	Parameter	Distribution[a]	Mean value[b]	Standard deviation[c]	Level of truncation[d]		Reference	Notes*	NRC[e]
					Minimum	Maximum			
Ingestion Dosimetry [Eqs. (4), (11), and (12)]	D_{ij} (^{90}Sr)	L	1.6×10^{-3} mrem/pCi	(1.4)	---	---	This report (Appendix A)	A	7.58×10^{-3} mrem/pCi
	D_{ij} (^{137}Cs)	L	3.7×10^{-5} mrem/pCi	(1.32)	---	---	Schwarz and Dunning (in press)	A	7.97×10^{-5} mrem/pCi
Fish Bioaccumulation [Eq. (4)]	B_{ip} (^{90}Sr)	L	11 L/kg	(6.0)	---	---	Hoffman and Baes (1979)		30 L/kg
	B_{ip} (^{137}Cs)	L	1300 L/kg	(2.36)	---	---	Hoffman and Baes (1979)		2000 L/kg
Deposition-Leafy Vegetables [Eq. (8)]	B_{iv} (^{90}Sr)[f]	L	0.33	(3.3)	7×10^{-3}	2.4	Ng et al. (in preparation); Baes et al. (in preparation)	B	1.7×10^{-2}
	B_{iv} (^{137}Cs)[f]	L	5.5×10^{-3}	(4.5)	1×10^{-4}	8×10^{-2}	Ng et al. (in preparation); Baes et al. (in preparation)	B	~~4×10^{-5}~~ 1×10^{-2}
Deposition-Non-Leafy-Vegetables [Eq. (8)]	B_{iv} (^{90}Sr)[f]	L	8.5×10^{-2}	(3.9)	8×10^{-4}	3.4	Ng et al. (in preparation); Baes et al. (in preparation)	B	1.7×10^{-2}
	B_{iv} (^{137}Cs)[f]	L	5×10^{-3}	(4.5)	1×10^{-5}	1×10^{-1}	Ng et al. (in preparation); Baes et al. (in preparation)	B	4×10^{-3}
Deposition-Pasture Vegetation [Eq. (8)]	B_{iv} (^{90}Sr)[g]	L	1.4	(3.42)	6×10^{-2}	46	Ng et al. (in preparation); Baes et al. (in preparation)	B	6.8×10^{-2}
	B_{iv} (^{137}Cs)[g]	L	4.4×10^{-2}	(3.82)	7×10^{-4}	1.2	Ng et al. (in preparation); Baes et al. (in preparation)	B	1.5×10^{-2}

Table 2. (continued)

Pathway	Parameter	Distribution[a]	Mean value[b]	Standard deviation[c]	Level of truncation[d]		Reference	Notes*	NRC[e]
					Minimum	Maximum			
Rate Constant for Loss from Soil Root Zone [Eq. (8)] - all pathways	λ_{is} (^{90}Sr)	L	6.7×10^{-5} d^{-1}	(7.4)	1.1×10^{-7} d^{-1}	1.2×10^{-2} d^{-1}	Baes and Sharp (submitted)	C	0 d^{-1}
	λ_{is} (^{137}Cs)	L	1.7×10^{-6} d^{-1}	(6.7)	4×10^{-7} d^{-1}	8×10^{-4} d^{-1}	Baes and Sharp (submitted)	C	0 d^{-1}
Pasture-Milk [Eq. (9)]	F_{im} (^{90}Sr)	L	1.2×10^{-3} d/L	(1.62)	2×10^{-4} d/L	8×10^{-2} d/L	Hoffman and Baes (1979); Ng (1981)		8×10^{-4} d/L
	F_{im} (^{137}Cs)	L	6.7×10^{-3} d/L	(1.79)	1.2×10^{-3} d/L	3×10^{-2} d/L	Hoffman and Baes (1979)		1.2×10^{-2} d/L
Pasture-Meat [Eq. (10)]	F_{if} (^{90}Sr)	L	5.8×10^{-4} d/kg	(3.3)	4×10^{-5} d/kg	4×10^{-3} d/kg	Ng et al. (in preparation		6×10^{-4} d/L
	F_{if} (^{137}Cs)	L	2.1×10^{-2} d/kg	(2.0)	3×10^{-3} d/kg	2×10^{-1} d/kg	Ng et al. (in preparation		4.4×10^{-3} d/kg
Radiological Decay Constant - all Pathways [Eq. (8)]	λ_{ir} (^{90}Sr)	C	6.64×10^{-5} d^{-1}	---	---	---	Kocher (1981)		6.64×10^{-5} d^{-1}
	λ_{ir} (^{137}Cs)	C	6.29×10^{-5} d^{-1}	---	---	---	Kocher (1981)		6.29×10^{-5} d^{-1}

[a] Probability distribution, where L = lognormal and C = constant.

[b] For lognormal distributions, the "mean value" is the geometric mean X_m.

[c] For lognormal distributions, the "standard deviation" refers to the geometric standard deviation s_g (in parenthesis).

[d] Assumed extreme values for truncating the random selection of parameter values using the Monte Carlo approach for error propagation.

[e] Generic default values recommended in NRC Regulatory Guide 1.109 (USNRC, 1977).

[f] Fresh weight vegetation, dry weight soil.

[g] Dry weight vegetation, dry weight soil.

*Explanation of notes:

A The D_{ij} for ^{137}Cs is specific to the whole-body; the D_{ij} for ^{90}Sr is specific to the bone surface (endosteal cells); the NRC default D_{ij} for ^{90}Sr is specific to the total bone.

B Judgment exercised in the review of the cited references to derive a mean value and standard deviation; B_{iv} for pasture vegetation has been determined for the dry weight of vegetation; B_{iv} for leafy vegetables and non-leafy vegetables has been determined on a fresh weight basis.

C Values are estimated assuming a constant rate of moisture infiltration into soil of 60 cm/yr; the variability in λ_{is} is dominated by the variability in K_d (soil/water distribution coefficient).

using the Monte Carlo approach, thus restricting the selection of parameter values from their prescribed distribution. In general, these minimum and maximum values have been set for a range that is four times larger than the observed range. Although this choice is arbitrary, past investigations have not shown the selection of minimum and maximum extremes to have a profound effect on the numerical generation of distributions of dose predictions (Schwarz and Hoffman, 1981).

Table 1 lists parameters that are nuclide independent, and Table 2 lists parameters that are dependent on the nuclide (or element) of interest. As mentioned previously (Sect. 2.1), statistical properties for each parameter are based on a careful review of original references by other investigators (cited in Tables 1 and 2) and on the use of judgment to estimate means, standard deviations, and minimum and maximum extreme values for parameters associated with limited data. Notes in the tables identify situations in which additional interpretation and judgment was necessary. The parameters for which judgment formed the primary basis for specification of values in Tables 1 and 2 are:

r/Y_v for leafy and non-leafy vegetables,

λ_w for non-leafy vegetables,

Q_f for beef cattle, and

U_f for members of the exposed human population group.

Adjustments were made in estimating the mean values of:

Q_m for dairy cattle,

Q_f for beef cattle, and

U_n for members of the exposed human population group,

to account for the likely consumption of animal feeds and food crops produced outside the immediate region of contamination (assumed to be not greater than a few kilometers from the release source). For dairy and beef cattle, respectively, the mean values of Q_m and Q_f in Table 1

were derived assuming all grains and concentrates are imported from outside the immediate region. The geometric mean value for U_n assumes only 25% of the total ingested amount of non-leafy vegetables is home grown. Maximum values for these parameters were set equal to maximum reported values with no importation of feed or food from outside the immediate region.

Estimates given for

λ_w for leafy vegetables,

t_e for all terrestrial pathways,

U_l for leafy vegetable consumption, and

U_n for non-leafy vegetables,

were derived from reported median and maximum values using the approaches described in Sect. 2.1.

Estimates for

U_m for milk consumption, and

U_F for fish consumption,

are based on published percent frequency distributions that summarize the results of specific surveys. The variability expressed for U_m should be conservatively biased because the survey is based on daily recall responses to a single interview, although the number of milk consuming persons questioned was substantial (1,980 individuals). The variability in daily patterns of milk consumption is expected to be greater than for consumption rates averaged over time spans similar to those addressed by the model (on the order of one year).

The large variability estimated for λ_{is} for both ^{90}Sr and ^{137}Cs reflects the wide spectrum in values reported for the soil (sediment)-to-solution distribution coefficient (K_d) for the elements strontium and cesium. This variability is the consequence of a large diversity in soil chemistry and experimental procedures used to determine K_d

(Baes and Sharp, submitted). In addition, strong correlations are expected between B_{iv} and K_d (Baes et al., in preparation). Thus, because of the inverse relationship between K_d and λ_{is}, high values of B_{iv} should correspond to high values of λ_{is}. No attempt is made in this study to explicitly account for the effect of correlations between B_{iv} and λ_{is}.

Estimates for the ^{90}Sr dose factors D_{ij} for the endosteal region and active red marrow of the bone are derived from measured variability in ^{90}Sr bone concentrations (Schubert et al., 1967) and equating the ingestion dose factor for ^{90}Sr derived from ICRP-30 (1979) to the geometric mean of a lognormal distribution (see Appendix A). In Table 2, only the ^{90}Sr D_{ij} for the endosteal region of the bone is listed. For the ingestion of ^{137}Cs, whole body D_{ij} estimates are taken directly from a detailed statistical analysis performed by Schwarz and Dunning (in press). These ^{137}Cs D_{ij} estimates include the effect of observed correlations between age, body mass, and whole body retention of ^{137}Cs.

Further considerations for other parameters not discussed in the text of this report are noted in Tables 1 and 2.

The applicability of these estimates of parameter variability (and their underlying assumptions) to conditions prevailing at a specific site is obviously subject to question. Nevertheless, we expect the estimates presented in Tables 1 and 2 to encompass site-specific conditions averaged for time periods of comparable length to that considered by the model. Parameter variability estimated from a review of data reported in the available literature will likely exceed the variability of parameter values for a specific site.

Interpretation of the results of this study should be restricted to situations where site-specific information is not available. However, site-specific data, when available, should always have priority over the values listed in Tables 1 and 2. In the absence of such data, ranking of the importance of the present estimates of model parameters (with respect to the variability in the prediction of dose equivalent) should provide direction for the acquisition of improved parameter estimates.

3.3 RESULTS OF COMBINED PARAMETER VARIABILITY

The distribution of predicted dose equivalents arising from the various food chain pathways is represented in Tables 3 and 4 through values of the geometric mean (X_m) and the geometric standard deviation (s_g). In these tables, the dose equivalents calculated using the generic parameter values in Regulatory Guide 1.109 (USNRC, 1977) are listed and compared with the distribution of predicted dose equivalents derived from estimates of parameter variability. The degree of potential conservatism associated with the Regulatory Guide 1.109 predictions is given by the cumulative probability. The cumulative probability indicates the chance that predicted dose equivalents derived from estimates of parameter variability will be equal to or less than the Regulatory Guide 1.109 prediction. For example, cumulative probabilities on the order of 0.90 indicate only one of 10 predicted doses are expected to exceed the regulatory guide dose estimate, while a cumulative probability of 0.99 indicates that the regulatory guide dose estimate will only be exceeded one out of 100 times.

The relative importance of model parameters and individual pathways to the variability of the predicted dose equivalent is given in Tables 5 and 6. The values listed under "importance index" represent the fraction of the total variability in the predicted dose accounted for by variability in the model parameter or pathway. Correlations among the predicted doses for each individual exposure pathway occur in direct proportion to the number of parameters shared (i.e., all pathways share the parameter D_{ij}). All terrestrial pathways share the parameters λ_{is}, p, λ_{ir}, and t_b. In addition, the meat and milk pathways also share the parameters r/Y_v, λ_w, and B_{iv}. A higher importance rank is attained by shared parameters in the analysis of the dose received through combined pathways of exposure than is attained by these parameters in the analysis of individual exposure pathways (Tables 5 and 6).

Table 3. Variability in predicted dose equivalents to the bone surface for ^{90}Sr ingested via selected food chain pathways

Pathway	X_m[a]	s_g[b]	NRC[c] (percentile)[d]	Method[e]
Water-Fish-Man (mrem per pCi/l)	0.25	7.2	4.8 (0.93)	A
Deposition-Leafy Vegetables-Man (mrem per pCi/m^2·d)	0.27	3.0	1.1 (0.90)	B
Deposition-Non-leafy Vegetables-Man (mrem per pCi/m^2·d)	0.34	3.7	8.8 (>0.99)	B
Deposition-Pasture-Milk-Man (mrem per pCi/m^2·d)	0.15	3.7	0.43 (0.79)	B
Deposition-Pasture-Meat-Man (mrem per pCi/m^2·d)	0.055	4.2	0.13 (0.73)	B
Deposition-All Terrestrial Pathways-Man (mrem per pCi/m^2·d)	1.2	2.4	10.3 (>0.99)	B

[a] Geometric mean.

[b] Geometric standard deviation, unitless.

[c] Predicted dose equivalent for total bone using values in Regulatory Guide 1.109 (USNRC, 1977).

[d] Cumulative probability associated with NRC prediction.

[e] Explanation of method used for error propagation of model parameters: A = lognormal statistics (Section 2.2.1); B = Monte Carlo computer techniques (Section 2.2.2).

Table 4. Variability in predicted dose equivalents to the whole body for ^{137}Cs ingested via selected food chain pathways

Pathway	X_m [a]	s_g [b]	NRC[c] (percentile)[d]	Method[e]
Water-Fish-Man (mrem per pCi/L)	0.67	3.3	3.3 (0.91)	A
Deposition-Leafy Vegetables-Man (mrem per pCi/m^2·d)	1.4×10^{-3}	2.4	1.1×10^{-2} (>0.99)	B
Deposition-Non-leafy Vegetables-Man (mrem per pCi/m^2·d)	3.4×10^{-3}	3.3	8.7×10^{-2} (>0.99)	B
Deposition-Pasture-Milk-Man (mrem per pCi/m^2·d)	8.2×10^{-3}	3.2	6.4×10^{-2} (0.96)	B
Deposition-Pasture-Meat-Man (mrem per pCi/m^2·d)	2.1×10^{-2}	2.9	9.0×10^{-3} (0.21)	B
Deposition-All Terrestrial Pathways-Man (mrem per pCi/m^2·d)	4.4×10^{-2}	2.2	0.17 (0.96)	B

[a] Geometric mean.

[b] Geometric standard deviation, unitless.

[c] Predicted dose equivalent for total bone using values in Regulatory Guide 1.109 (USNRC, 1977).

[d] Cumulative probability associated with NRC prediction.

[e] Explanation of method used for error propagation of model parameters: A = lognormal statistics (Section 2.2.1); B = Monte Carlo computer techniques (Section 2.2.2).

Table 5. Ranking of parameter and pathway importance to variability in the dose prediction for ^{90}Sr

Pathway	Parameter	Importance rank		Importance index	
		(Pathway)	Parameter	(Pathway)	Parameter
Water-Fish-Man	B_{ip}		1		0.82[a]
	U_F		2		0.15[a]
	D_{ij}		3		0.03[a]
Deposition-All Terrestrial Pathways-Man	λ_{is}		1		0.18[b]
	D_{ij}		2		0.10[b]
	U_n		2		0.10[b]
	B_{iv}[c]		3		0.08[b]
	B_{iv}[d]		3		0.08[b]
	B_{iv}[e]		4		0.06[b]
Deposition-Non-leafy Vegetables-Man		(1)		(0.50[b])	
	U_n		1		0.32[b]
	B_{iv}		2		0.26[b]
	r/Y_v		3		0.12[b]
	λ_{is}		4		0.07[b]
	D_{ij}		5		0.05[b]

Table 5. (continued)

Pathway	Parameter	Importance rank		Importance index	
		(Pathway)	Parameter	(Pathway)	Parameter
Deposition-Leafy Vegetables-Man		(2)		(0.34[b])	
	B_{iv}		1		0.45[b]
	U_l		2		0.17[b]
	λ_{is}		3		0.15[b]
	D_{ij}		4		0.07[b]
Deposition-Pasture-Milk-Man		(3)		(0.28[b])	
	U_m		1		0.35[b]
	B_{iv}		2		0.21[b]
	F_{im}		3		0.16[b]
	D_{ij}		4		0.05[b]
Deposition-Pasture-Meat-Man		(4)		(0.03[b])	
	F_{if}		1		0.52[b]
	B_{iv}		2		0.15[b]
	U_f		3		0.11[b]

[a]Determined by dividing the variance of logtransformed parameter distribution by the variance of the logtransformed distribution of dose prediction.

[b]Determined by squaring the coefficient for the rank order correlation between parameter values and the predicted dose; only values above 0.04 are included in this table.

Table 6. Ranking of parameter and pathway importance to variability in the dose prediction for ^{137}Cs

Pathway	Parameter	Importance rank		Importance index	
		(Pathway)	Parameter	(Pathway)	Parameter
Water-Fish-Man	B_{ip}		1		0.52[a]
	U_F		2		0.42[a]
	D_{ij}		3		0.06[a]
Deposition-All Terrestrial Pathways-Man	F_{if}		1		0.25[b]
	r/Y_v(pasture)		2		0.15[b]
	D_{ij}		3		0.12[b]
	λ_w (pasture)		4		0.09[b]
	U_f		4		0.09[b]
	U_m		5		0.07[b]
Deposition-Pasture-Meat-Man		(1)		(0.73)[b]	
	F_{if}		1		0.43[b]
	U_f		2		0.19[b]
	r/Y_v		3		0.14[b]
	λ_w		4		0.08[b]
	D_{ij}		5		0.07[b]
Deposition-Pasture-Milk-Man		(2)		(0.34)[b]	
	U_m		1		0.46[b]
	F_{im}		2		0.22[b]

Table 6. (continued)

Pathway	Parameter	Importance rank		Importance index	
		(Pathway)	Parameter	(Pathway)	Parameter
Deposition-Pasture-Milk-Man (continued)	r/Y_v		3		0.10[b]
	λ_w		4		0.06[b]
	D_{ij}		5		0.05[b]
Deposition-Non-leafy-Vegetables-Man		(3)		(0.05)[b]	
	U_n		1		0.40[b]
	r/Y_v		2		0.31[b]
	λ_w		3		0.13[b]
	D_{ij}		4		0.06[b]
Deposition-Leafy-Vegetables-Man		(4)		(0.03)[b]	
	r/Y_v		1		0.30[b]
	U_l		2		0.25[b]
	λ_w		3		0.18[b]
	D_{ij}		4		0.11[b]
	B_{iv}		5		0.05[b]

[a]Determined by dividing the variance of logtransformed parameter distribution by the variance of the logtransformed distribution of dose prediction.

[b]Determined by squaring the coefficient for the rank order correlation between parameter values and the predicted dose; only values above 0.04 are included in this table.

3.3.1 Water-fish-man pathway

For both ^{90}Sr (Table 3) and ^{137}Cs (Table 4), the Regulatory Guide prediction of dose equivalents exceeds the 90*th* percentile. The freshwater fish bioaccumulation factor is the most important parameter for both nuclides, although this parameter is of predominant importance for ^{90}Sr (Tables 5 and 6). Much of this variability in the dose prediction for this pathway might be reduced if the bioaccumulation factor for ^{90}Sr and ^{137}Cs were derived on a site-specific basis using data on water quality, the trophic level of fish consumed, and the exchangeable calcium and potassium content in water (Hoffman and Baes, 1979; Vanderploeg et al., 1975).

3.3.2 Deposition-terrestrial food pathways - man

The prediction of dose equivalent for the ingestion of ^{90}Sr and ^{137}Cs ingested via terrestrial food pathways appears to be more variable for ^{90}Sr than ^{137}Cs (Tables 3 and 4). The most variable pathway is the meat pathway for ^{90}Sr, with a geometric standard deviation (s_g) of 4.2. and the non-leafy vegetable pathway for ^{137}Cs with a s_g of 3.3. The non-leafy vegetable pathway is the most important contributor (50%) to the combined exposure via all pathways for the ingestion of ^{90}Sr (Table 5), while the meat pathway is dominant (73%) for the combined pathway exposure to ^{137}Cs (Table 6).

Frequency distributions of the predicted dose equivalent due to the exposure to all terrestrial pathways have been generated through 500 computer iterations using the Monte Carlo approach. The distributions are skewed and are approximately lognormal (Figs. 1 and 2). Errors associated with the 95*th* percentile estimates of these distributions are about ±15%, whereas the 99*th* percentiles may vary by ±30%.

The dose from exposure to all terrestrial pathways has lower variability associated with it than for any single pathway (Tables 3 and 4). This effect is attributed to the process of adding independent distributions, which should result in a decreasing relative error (Bendat and Piersol, 1966). The s_g for combined pathway exposure is 2.4 for ^{90}Sr and 2.2 for ^{137}Cs. The lowest values of s_g for a single exposure pathway (leafy vegetables) are 3.0 and 2.4 for ^{90}Sr and ^{137}Cs, respectively.

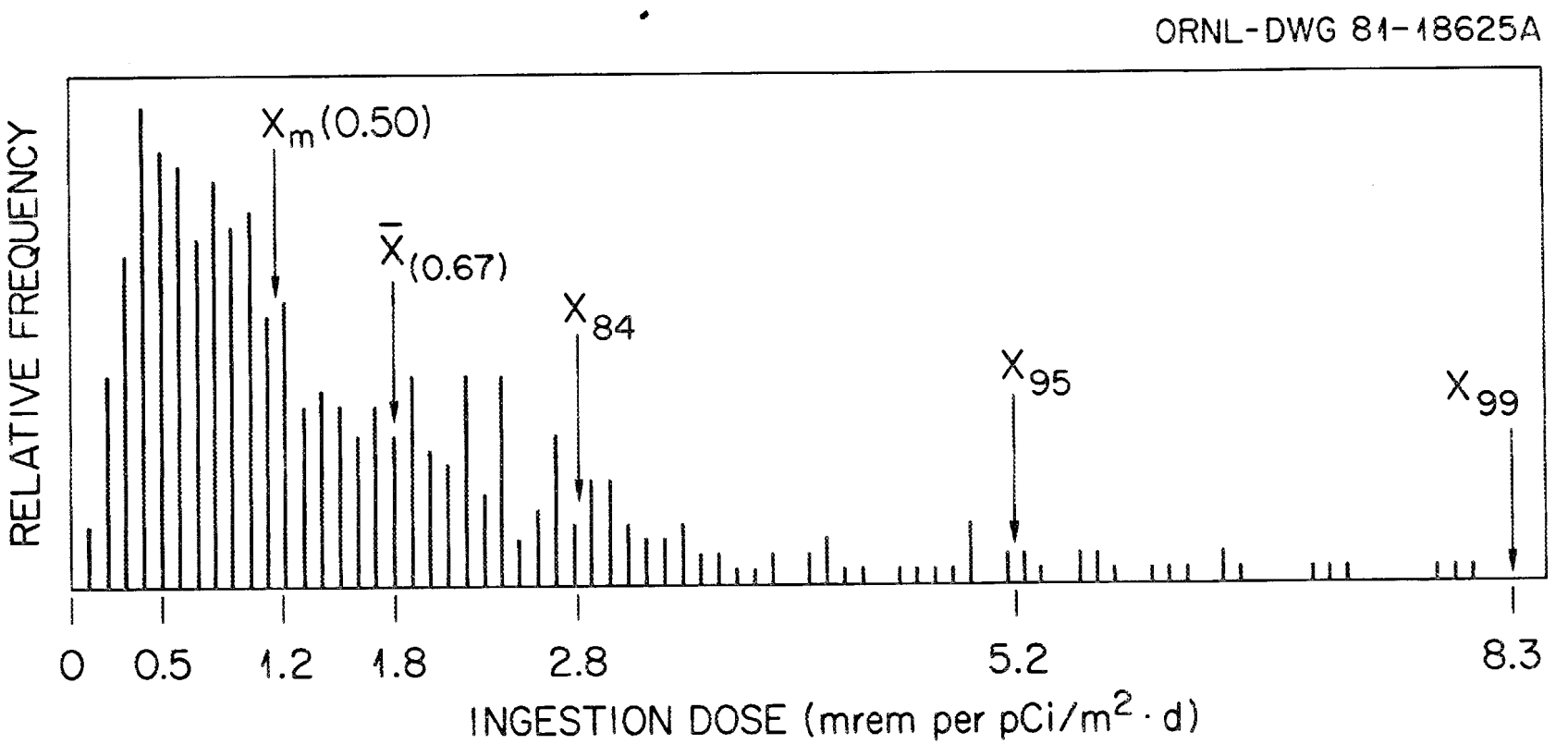

Fig. 1. A frequency distribution of the predicted Strontium-90 ingestion dose to bone surface via terrestrial food pathways subsequent to a continuous rate of deposition of 1 pCi/m^2·d for 15 years. X_m, X, X_{84}, X_{95}, and X_{99} are the median, mean, 84*th*, 95th, and 99*th* percentiles of the distribution, respectively.

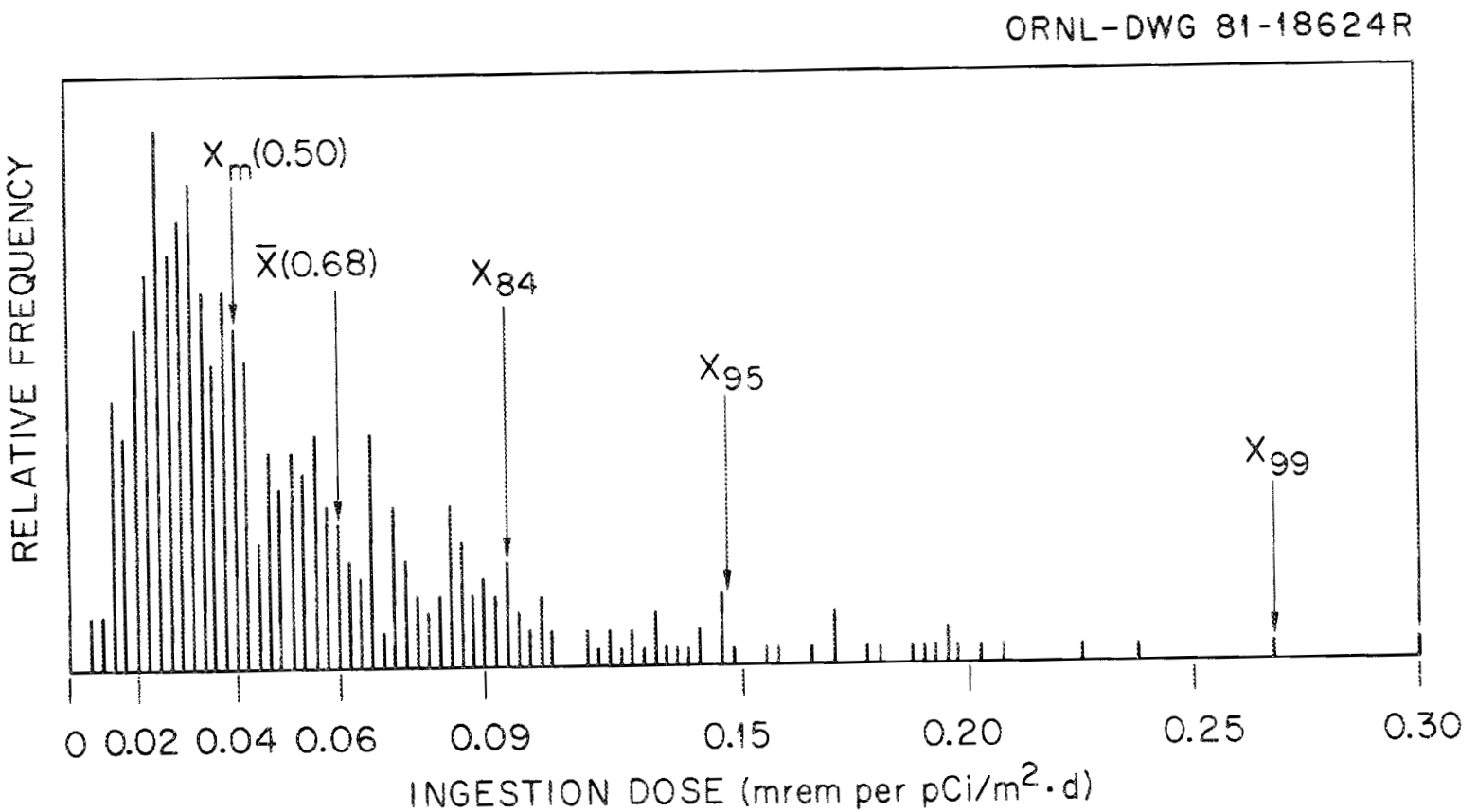

Fig. 2. A frequency distribution of the predicted Cesium-137 ingestion dose to total body via terrestrial food pathways subsequent to a continuous rate of deposition of 1 pCi/m^2·d for 15 years. X_m, $\bar{X}$, X_{84}, X_{95}, and X_{99} are the median, mean, 84*th*, 95*th*, and 99*th* percentiles of the distribution, respectively.

For the combined pathway exposure to ^{90}Sr, the parameters λ_{is}, D_{ij}, and U_n are ranked highest in importance (Table 5). The importance of λ_{is} is due to the significance of the soil as a secondary source of ^{90}Sr after 15 years of continuous deposition and the fact that λ_{is} is shared by all pathways. For combined pathway exposure to ^{137}Cs, the parameter F_{if} is ranked highest in importance, followed by the interception (r/Y_v) and retention (λ_w) parameters for pasture vegetation (Table 6). The parameters r/Y_v and λ_w directly affect the variability of the concentration of deposited ^{137}Cs in vegetation consumed by both milk and meat producing animals.

The Regulatory Guide 1.109 (USNRC, 1977) prediction of dose equivalent due to exposure to all four terrestrial pathways exceeds the ~~99th~~ 77th percentile for ^{90}Sr (Table 3) and the ~~99th~~ 95th percentile for ^{137}Cs (Table 4). Much of this apparent conservatism is associated with the dietary habits U_p assumed in Regulatory Guide 1.109 for maximally exposed individuals (Table 1). However, conservative bias is also associated with the NRC selection of ingestion dose factors D_{ij} for ^{90}Sr and ^{137}Cs (Table 2). Nevertheless, we note that the NRC dose estimates for ^{90}Sr are not directly comparable to the predicted distribution of doses in Table 3 because of differences in the assumed target tissue of the bone (Appendix A). The NRC ^{90}Sr D_{ij} is specific to the total bone while the values of D_{ij} used to analyze the variability of model predictions is specific for the endosteal cells of the bone surface. Substituting the ICRP-30 (1979) ^{90}Sr endosteal cell D_{ij} for the total bone D_{ij} used in Regulatory Guide 1.109 would reduce the NRC estimates in Table 3 by a factor of 4.7.

The only pathway in which the NRC estimates appear to be less than conservative is the ^{137}Cs meat pathway (Table 4). This is attributable to the generic value for F_{if} given in Regulatory Guide 1.109 which is near the lower end of the range of values reported for this parameter (Ng, Colsher, and Thompson, in preparation; Hoffman and Baes, 1979). The prediction of the concentration of ^{90}Sr and ^{137}Cs in terrestrial foods (Tables 7 and 8) is less variable than the prediction of dose, due to the influence of fewer parameters. Less conservatism is also evident with Regulatory Guide 1.109 predictions of food concentrations than for

Table 7. Variability in predicted ^{90}Sr food concentrations for selected terrestrial pathways

Pathway	X_m [a]	s_g [b]	NRC [c] (percentile) [d]
Deposition-Leafy Vegetables [Eq. (8)] [e]	9.0 pCi/kg	2.6	2.2 pCi/kg (0.07)
Deposition-Non-leafy Vegetables [Eq. (8)]	4.5 pCi/kg	2.7	2.2 pCi/kg (0.24)
Deposition-Pasture-Milk [Eq. (8) and (9)]	0.96 pCi/L	2.7	0.18 pCi/L (0.05)
Deposition-Pasture-Meat [Eq. (8) and (10)]	0.35 pCi/kg	3.7	0.15 pCi/kg (0.26)

[a] Geometric mean.

[b] Geometric standard deviation, unitless.

[c] Predicted food concentrations using values in Regulatory Guide 1.109 (USNRC, 1977).

[d] Cumulative probability associated with NRC prediction.

[e] Food concentrations resulting from a constant deposition of 1 $pCi/m^2 \cdot d$ over a period of 15 years; error propagation performed using Monte Carlo computer techniques (Section 2.2.2).

Table 8. Variability in predicted ^{137}Sr food concentrations for selected terrestrial pathways

Pathway	X_m [a]	s_g [b]	NRC [c] (percentile) [d]
Deposition-Leafy Vegetables [Eq. (8)] [e]	2.1 pCi/kg	2.0	2.1 pCi/kg (0.50)
Deposition-Non-leafy Vegetables [Eq. (8)]	2.1 pCi/kg	2.4	2.1 pCi/kg (0.50)
Deposition-Pasture-Milk [Eq. (8) and (9)]	2.4 pCi/L	2.2	2.6 pCi/L (0.54)
Deposition-Pasture-Meat [Eq. (8) and (10)	6.2 pCi/kg	2.4	1.0 pCi/kg (0.02)

[a] Geometric mean.

[b] Geometric standard deviation, unitless.

[c] Predicted food concentrations using values in Regulatory Guide 1.109 (USNRC, 1977).

[d] Cumulative probability associated with NRC prediction.

[e] Food concentrations resulting from a constant deposition of 1 pCi/m^2·d over a period of 15 years; error propagation performed using Monte Carlo computer techniques (Section 2.2.2).

predictions of dose. The most pronounced indication of potential NRC underestimation of food concentrations is for the prediction of ^{90}Sr in milk and leafy vegetables, and for ^{137}Cs in meat. The predicted NRC concentrations for ^{90}Sr in leafy vegetables and milk, and for ^{137}Cs in meat are factors of 4.1, 5.3, and 6.2, respectively, lower than the predicted geometric mean. These concentrations occur respectively at the *7th*, *5th*, and *2nd* cumulative percentile of the distributions of food concentrations produced using Monte Carlo computer techniques.

4. TRITIUM AND CARBON-14 SPECIFIC ACTIVITY MODELS

The "specific activity" approach is commonly employed for predicting the environmental transport and dosimetry of ^{3}H and ^{14}C, assuming the predominant chemical form of these radionuclides in the atmosphere is tritiated water vapor (HTO) and $^{14}CO_2$, respectively. The specific activity approach assumes that the environmental behavior and fate of a radionuclide is approximately the same as for stable isotopes of the same element. The specific activity approach for tritium is useful because most of the hydrogen in the body is associated with body water and most of the body water can be accounted for through ingestion of food and beverages, and absorption of atmospheric water through the skin or the respiratory system (Till et al., 1980a). Ultimately, the source of this water is assumed to be the atmospheric humidity (which also forms rain, the primary source of water for animal and plant food products and beverages). The specific activity approach for ^{14}C is useful because most of the carbon in the body is derived from carbon in food, beverages, and the atmosphere. The carbon in food and drinks ultimately originates from CO_2 in the atmosphere via either direct absorption by green plants or fixation through photosynthesis (Killough and Rohwer, 1978).

4.1 MODEL STRUCTURE AND ASSUMPTIONS

The basic structure of "specific activity" models is a multiplicative chain of parameters which relate the atmospheric specific activity (activity of the radionuclide per mass of stable element) to the specific activity in the body. Dose equivalent is predicted using a dose-rate factor appropriate for steady-state conditions when the body specific activity has approached equilibrium. Under the assumption of complete equilibrium:

$$\text{Annual Dose}_{ij} = (\text{SpA})_i \cdot (\text{DRF})_{ij}, \qquad (13)$$

where

$(SpA)_i$ = the atmosphere specific activity relative to the radionuclide i of concern and its stable element, and

$(DRF)_{ij}$ = the conversion factor that relates the body specific activity of radionuclide i and its stable element to a dose equivalent rate at equilibrium for a given organ, tissue, or the whole body (j).

For tritium,

$$SpA = \chi/H, \tag{14}$$

where

χ = the ground-level air concentration of ^{3}H (pCi/m^3), and

H = the absolute humidity of the ground-level atmosphere ($g[H_2O]/m^3$)

For Carbon-14,

$$SpA = \chi/C, \tag{15}$$

where

χ = the ground-level air concentration of ^{14}C (pCi/m^3), and

C = the concentration of airborne carbon (g/m^3).

4.2 SOURCES OF UNCERTAINTY

The primary sources of uncertainty associated with the specific activity approach are related to:

(1) the chemical form of the radionuclide and the chemical form of the stable isotope of the same element,
(2) the estimation of the concentration of the radionuclide in air,
(3) the estimation of the concentration of the stable isotope in air, and
(4) the assumption that, at complete equilibrium, all of the radionuclide and the stable isotope in the body is derived from the atmospheric specific activity at a given location.

For tritium and ^{14}C, chemical forms other than HTO or $^{14}CO_2$ in the atmosphere, and conditions other than complete equilibrium between a receptor and the environment, will result in a reduction in the estimated radiological impact. These conditions are likely to occur. Releases of chemical forms of tritium other than HTO have been reported, such as HT, TT, and tritiated hydrocarbons (Kahn et al., 1971; Blanchard et al., 1976; Murphy and Pendergast, 1979) as well as chemical forms of ^{14}C other than $^{14}CO_2$, such as ^{14}CO, and $^{14}CH_4$ and other hydrocarbons containing ^{14}C (Schwibach, Riedel, and Bretschneider, 1978). These chemical forms are less available for incorporation into biological systems than H_2O and CO_2. Furthermore, complete equilibrium assumes that all of the hydrogen and carbon in the body is derived from the carbon and hydrogen in the atmosphere at a given location. The intake of carbon and hydrogen from locations with substantially lower specific activities would lead to a reduction in the body specific activity from that assumed at complete equilibrium.

Aside from the obvious, i.e., import of food products and beverages from outside the locality of concern, a variety of conditions can lead to an effective intake of carbon or hydrogen from locations with lower specific activities. For example, at the location of the maximum atmospheric specific activity of tritium, irrigation with water from

streams or deep aquifers will provide vegetation with a source of water of potentially lower tritium specific activity. A similar condition is expected when rain is formed above a plume containing tritium. The opposite effect may occur when food crops are grown so close to an elevated source of tritium that the plume does not effectively reach ground level (Vogt, 1979). In this special case, the specific activity of rain will be greater than the specific activity of the ground-level atmospheric humidity. For ^{14}C, the importation of feeds for local domestic animals used as a food source, and the ingestion of local aquatic foods whose carbon source originates outside the region of contamination, will also result in a reduction in the ^{14}C specific activity in the human body from that assumed at complete equilibrium.

The specific activity approach under the assumption of complete equilibrium, should produce an upper limit estimate of dose, provided that:

(1) the atmospheric specific activity has been correctly estimated, and
(2) the atmospheric specific activity at the location of concern is not less than at other locations (including locations above the plume) from which the human body could receive carbon or hydrogen.

For the case where complete equilibrium is assumed, specific activity calculations are independent of the age of the receptor. The estimation of the body specific activity and the dose rate factor do not require information on body mass, physiological uptake and retention of the radionuclides, and rates of inhalation and ingestion. This information is of importance for the ^{3}H and ^{14}C dose calculations performed with the models assumed in Regulatory Guide 1.109 (USNRC, 1977), as complete equilibrium is not assumed between the human receptor and the atmospheric specific activity.

Errors in the calculation of the annual average air concentrations from point source releases are expected to be factors of about 2 to 4 for distances less than 10 km under non-complex terrain conditions (Little and Miller, 1979; Crawford, 1978). Seasonal and diurnal

meteorological conditions may result in annual average atmospheric dispersion calculations for elevated releases underestimating plant fixation of CO_2. Unstable weather categories are most prevalent during the daylight hours of the growing season when photosynthesis is active (Killough and Rohwer, 1978). These categories tend to increase the vertical dispersion of pollutants in the atmosphere, leading to higher ground-level air concentrations near the source of an elevated release than during more stable weather conditions. An example given by Killough and Rohwer (1978) indicates that, for relatively short downwind distances from a 100 m high source, calculated annual 24-hr. average air concentrations of ^{14}C were a factor of 3 lower than calculated air concentrations using daytime meteorology averaged over a growing season.

The annual average variability of atmospheric hydrogen (in the form of H_2O vapor) is relatively small, but larger than the annual average variability of atmospheric carbon (in the form of CO_2). The annual average absolute humidity throughout the United States ranges from 3 to 17 g/m^3 (Etnier, 1980), whereas in non-industrial locations the annual average atmospheric carbon ranges from about 0.16 to 0.20 g/m^3 (Reiter, 1971). The atmospheric carbon concentration assumed by the NRC is 0.16 g/m^3 (USNRC, 1977), and the absolute humidity is 8 g/m^3 (Eckerman et al., 1980).

For tritium, additional sources of uncertainty are primarily related to internal dosimetry. The value of the appropriate "quality factor" (QF) has recently come into question. Based on a review of the literature, Till, Etnier, and Meyer, (1980b) recommend a QF for internally deposited tritium of 2 rather than the value of 1.0 recommended by the NCRP (1979). The influence of organically bound tritium on the magnitude of the dose rate factor also merits further investigation (Till et al., 1980a). In general, it is assumed that dose estimates based on only the body water tritium contribution may be increased by 20% to adequately include the dose contribution from organically bound tritium (Rohwer, 1976; NRCP, 1979). The validity of this assumption is currently being re-assessed at Oak Ridge National Laboratory (Killough, 1982).

4.3 COMPARISON OF NRC TERRESTRIAL PATHWAY MODEL PREDICTIONS WITH SPECIFIC ACTIVITY MODEL PREDICTIONS FOR ^{3}H AND ^{14}C

A comparison between total-body dose equivalent estimates for ^{3}H and ^{14}C using the specific activity approach and the terrestrial pathway models in NRC Regulatory Guide 1.109 (USNRC, 1977) is of interest to determine the relationship of the NRC pathway predictions to values considered to be upper limits. For the purposes of this comparison, ground-level air concentrations of ^{3}H and ^{14}C are assumed to be exact. The specific activity approach is used assuming complete equilibrium between the specific activity in the receptor and the atmosphere, with no hydrogen or carbon imported from outside the region of concern.

Assuming a concentration of 1 pCi/m^3 of ^{3}H or ^{14}C in the ground-level atmosphere and the water vapor and carbon content in the atmosphere to be 8 g/m^3 and 0.16 g/m^3, respectively, the ground-level atmospheric specific activities for these radionuclides would be:

0.125 pCi of ^{3}H/g H_2O

and

6.25 pCi of ^{14}C/g carbon.

The maximum total-body dose equivalent rate for an individual is calculated (Eq. 12) through multiplication of the above values times the appropriate dose-equivalent rate factors $(DRF)_{ij}$.

For the estimation of the total-body dose due to ingestion of tritium, a $(DRF)_{ij}$ of 9.5×10^{-2} mrem/yr per pCi/g H_2O is given by Till et al. (1980a). This value should be raised by a factor of 2 if a tritium quality factor (QF) of 2.0 is considered more appropriate than a QF of 1.0 (Till, Etnier, and Meyer, 1980b).

For ^{14}C, the total-body ingestion $(DRF)_{ij}$ is 2.1×10^{-1} mrem/yr per pCi/g C (Killough and Rohwer, 1978). The maximum annual dose equivalent using the specific activity approach (Eq. 11) is:

1.2×10^{-2} mrem for tritium

(or 2.4×10^{-2} mrem assuming a QF of 2.0), and

1.3 mrem for ^{14}C.

In Regulatory Guide 1.109 (USNRC, 1977), the specific activity approach is used to estimate the transfer of ^{3}H and ^{14}C from air to vegetation. The concentration of ^{3}H or ^{14}C per mass vegetation (C_{iv}) is estimated in Reg. Guide 1.109 by multiplying the vegetation specific activity by the fraction of the total mass of the edible tissue of vegetation assumed to be composed of water or carbon. For vegetation, 75% of the total fresh mass of edible tissue is assumed to be water, while 11% of the total fresh mass of edible tissue is assumed to be carbon. In addition, the specific activity of tritium in the water content of vegetation is assumed to be only 50% of the specific activity of atmospheric tritiated water vapor. Using the Regulatory Guide 1.109 approach (USNRC, 1977), a 1 pCi/m^3 concentration of ^{3}H or ^{14}C in the above-ground atmosphere would result in a calculated vegetation concentration of:

$$C_v^{3H} = (1 \text{ pCi m}^3)(1/8 \text{ g m}^{-3})(0.50)(0.75)(1000 \text{ g kg}^{-1})$$

$$= 47 \text{ pCi/kg fresh weight,}$$

$$C_v^{14C} = (1 \text{ pCi/m}^3)(1/0.16 \text{ g m}^{-3})(0.11)(1000 \text{ g/kg}^{-1})$$

$$= 690 \text{ pCi kg}^{-1} \text{ fresh weight.}$$

The NRC terrestrial pathway models for ^{3}H and ^{14}C are similar to those used for ^{90}Sr and ^{137}Cs [Eqs. (8), (9), (10), and (11)]. The default values for nuclide-dependent parameters recommended in Regulatory Guide 1.109 for ^{3}H and ^{14}C in the absence of site-specific data are listed in Table 9. Nuclide-independent values are given in Table 1.

Table 9. Nuclide-dependent parameters used in Regulatory Guide 1.109 for the terrestrial transport and dosimetry of ^{3}H and ^{14}C

Parameter	Nuclide	Parameter value[a]
Milk transfer coefficient		
F_m (d/l)	^{3}H	1.0×10^{-2}
	^{14}C	1.2×10^{-2}
Meat transfer coefficient		
F_f (d/kg)	^{3}H	1.2×10^{-2}
	^{14}C	3.1×10^{-2}
Ingestion dose conversion factor[b]		
D_{ij} (mrem/pCi)	^{3}H	1.05×10^{-7}[c]
	^{14}C	5.68×10^{-7}

[a] All values have been derived from the specific activity approach.

[b] Dose factors are specific for the total body.

[c] Value based on QF of 1.7.

Because the concentration of ^{3}H and ^{14}C in vegetation is calculated on a fresh weight basis, the NRC values for the daily dry matter consumption of grazing animals (Q_m and Q_f) listed in Table 1 should be multiplied by four to convert back to the fresh weight values (50 kg/d) recommended in Regulatory Guide 1.109 (USNRC, 1977).

For both ^{3}H and ^{14}C, the ingestion dose calculated with the pathway models in Regulatory Guide 1.109 is dominated by the consumption of vegetables, assuming all vegetables are grown at the location of maximum exposure (Table 10). The comparisons of model predictions (Table 10) indicate that the calculated annual total-body ^{3}H dose equivalent for all terrestrial pathways using the Regulatory Guide 1.109 approach is approximately 33% of the maximum upper limit, whereby the NRC dose estimate for ^{14}C is approximately 28% of the maximum. A lower percentage (16%) would be obtained by the NRC pathway calculations for tritium if the specific activity dose rate factor $(DRF)_{ij}$ were to be based on a quality factor (QF) of 2.0 instead of 1.0. By comparison, the QF associated with the tritium D_{ij} used in the pathway analysis of NRC is 1.7 (Till et al., 1980a).

Table 10. A comparison between Regulatory Guide 1.109 and specific activity calculations of annual dose equivalent to a maximally exposed individual for a given concentration of ^{3}H and ^{14}C

Calculational approach	Pathway	Annual dose equivalent[a]	
		^{3}H	^{14}C
Regulatory Guide 1.109	air-vegetables-man	2.9×10^{-3}	2.3×10^{-1}
Regulatory Guide 1.109	air-vegetation-milk-man	7.7×10^{-4}	7.3×10^{-2}
Regulatory Guide 1.109	air-vegetation-meat-man	3.3×10^{-4}	6.7×10^{-2}
Regulatory Guide 1.109	air-all terrestrial pathways-man	4.0×10^{-3}	3.7×10^{-1}
Specific activity[b]	all pathways	1.2×10^{-2} [c]	1.3

[a] Annual dose equivalent (mrem/yr) resulting from 1 pCi/m^3 in above-ground air.

[b] Assuming complete equilibrium between the atmospheric specific activity and the human body; dose equivalents therefore represent maximum upper-limit estimates.

[c] For a quality factor of 2.0 (Till et al., 1980a; Till, Etnier, and Meyer, 1980b) this value should be raised by a factor of 2.

5. SUMMARY AND CONCLUSIONS

The comparison of Regulatory Guide 1.109 estimated doses with predicted statistical distributions of dose equivalents indicates a high level of conservatism when radiological assessments are performed using the site-independent default values recommended in the Regulatory Guide. Usually, dose predictions made with Regulatory Guide 1.109 are ~~at least~~ about one order of magnitude greater than the predicted geometric mean. An exception is the deposition-pasture-meat-man pathway for ^{137}Cs. The Regulatory Guide 1.109 estimate is less than the predicted geometric mean for this pathway by approximately a factor of 2, which is the result of a low value selected in the Regulatory Guide for the ^{137}Cs meat transfer coefficient (F_{if}).

For ^{90}Sr, much of the conservatism associated with the Regulatory Guide estimates is due to the NRC selection of the dose conversion factor (D_{ij}) for the total bone rather than the bone surface. The differences in the dose conversion factors for bone surface and total bone accounts for a factor of approximately 5, with the highest values being for total bone. Additional sources of conservatism can be attributed to the high values assumed for the consumption of non-leafy vegetables in the Regulatory Guide. In our analysis, one in one hundred individuals ingest locally produced non-leafy vegetables at a rate comparable to that assumed in the Regulatory Guide (520 kg/yr).

If values other than the recommended default parameters in Regulatory Guide 1.109 are employed, the conservatism associated with the Regulatory Guide predictions may be reduced. For example, not all of the radionuclide-dependent parameters (Table 2) recommended in the Regulatory Guide are conservative. Most of the conservative Regulatory Guide parameters are those listed as radionuclide-independent (Table 1), and many of these can be easily revised with readily available site-specific information on agricultural practices and dietary habits. Thus, the use of site-specific data for these parameters, while retaining the recommended Regulatory Guide generic default values for parameters which are less readily amended on a site-specific basis, may lead to

results that are substantially less conservative than indicated by the present analysis.

Ultimately, the final evaluation of the models and parameters recommended in Regulatory Guide 1.109 should come from the results of experimental validation. Nevertheless, priorities for experimental validation can be established through identification of the exposure pathways and parameters that contribute most to the uncertainty in the overall dose equivalent. In this study, the parameters for soil retention, soil-to-plant transfer, and ingestion dosimetry contribute most significantly to the variability in the predicted ^{90}Sr dose for the combined exposure to all terrestrial pathways. For ^{137}Cs, the meat transfer coefficient, the mass interception factor for pasture forage, and the ingestion dose factor are the most important parameters. The freshwater finfish bioaccumulation factor is the most important parameter for the dose prediction of ^{90}Sr and ^{137}Cs transported over the water-fish-man pathway.

The variability indicated by our analysis for exposure to all terrestrial food pathways exceeds the variability observed for large population groups exposed to ^{90}Sr and ^{137}Cs in weapons fallout. For example, Schubert et al. (1967) report a geometric standard deviation for ^{90}Sr per gram bone from 224 individuals less than 25 years old of $s_g = 1.55$. Schwarz and Dunning (submitted) in their review of the literature suggest that the overall variability for ^{137}Cs per gram of body tissue from multiple pathways of exposure is $s_g = 1.4$. The variability indicated by our analysis is $s_g = 2.4$ for the ^{90}Sr bone surface dose equivalent and $s_g = 2.2$ for the total body ^{137}Cs dose equivalent.

In addition to the numerous sources of bias associated with the estimation of parameter variability, unaccounted correlations between model parameters might also explain the differences between observed and predicted geometric standard deviations. The assumption of independent and uncorrelated model parameters was used in our analysis in the absence of data contradicting this assumption. However, the effect of correlations between parameters on the variability of model predictions could be substantial. For example, Schwarz and Dunning (submitted) and O'Neill, Gardner, and Mankin (1980) have demonstrated that small

correlations among model parameters significantly reduced the total variability in model predictions over that estimated assuming such correlations were non-existent. Future extentions of the work presented herein should address the effect of possible correlations among model parameters.

The analysis for ^{3}H and ^{14}C indicated that calculations performed with the models and parameters in Regulatory Guide 1.109 are approximately factors of 3 and 3.6, respectively, less than upper-limit estimates. The upper-limit estimates assume that all of the ^{3}H in the human body is derived from the ^{3}H in atmospheric water vapor and that all ^{14}C in the body is derived from ^{14}C in atmospheric CO_2 at locations representing maximum atmospheric specific activities. These upper-limit estimates are reliable provided that the calculation of the atmospheric specific activity of these radionuclides is accurate and that the intake of air, beverages, or food derived from locations with greater specific activities does not occur.

REFERENCES

Baes, C. F., III and R. D. Sharp. (submitted). "A method for determination of leaching constants for elements in agricultural soils for use in assessment models." *Journal of Environmental Quality.*

Baes, C. F., III, R. D. Sharp, A. L. Sjoreen, and R. W. Shor. (In preparation). *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture.* ORNL-5786.

Barr, A. J., J. H. Goodnight, and J. P. Sall. 1979. *SAS User's Guide.* SAS Institute, Inc., Raleigh, N. C.

Bendat, J. S. and A. G. Piersol. 1966. *Measurement and Analysis of Random Data.* Wiley, New York.

Blanchard, R. L., W. L. Brinch, H. E. Kolde, H. L. Krieger, D. M. Montgomery, S. Gold, A. Martin, and B. Kahn. 1976. *Radiological Surveillance Studies at the Oyster Creek BWR Nuclear Generating Station.* U. S. Environmental Protection Agency, EPA-520/5-76-003.

Carney, J. H., R. H. Gardner, J. B. Mankin, and R. V. O'Neill. 1981. *SEAP: The Computer Program for the Error Analysis of a Stream Model.* ORNL/TM-7694, Oak Ridge, Tenn.

Conover, W. J. 1971. *Practical Nonparameteric Statistics.* John Wiley and Sons, Inc.

Crawford, T. V. (Chairman). 1978. "Atmospheric transport of radionuclides." In *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.

Cuddihy, R. G., R. O. McClellan, and W. C. Griffith. 1979. "Variability in target organ deposition among individuals exposed to toxic substances." *Toxicol. Appl. Pharmacol.* 49:179-187.

Eckerman, K. F., F. J. Congel, A. K. Roecklein, and W. J. Pasciak. 1980. *User's Guide to GASPAR Code.* NUREG-0597.

Etnier, E. L. 1980. "Regional and site-specific absolute humidity data for use in tritium dose calculations." *Health Phys.* 39:318-320.

Gardner, R. H., R. V. O'Neill, and F. O. Hoffman. (1982). "Assessing model uncertainties." In *Proceedings of a Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste*. Gatlinburg, Tennessee, March 9-13, 1981. (D. C. Kocher, ed.) CONF-810372, NUREG/CP-0022. pp. 199-208.

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, NUREG/CR-1004.

Iman, R. L., W. J. Conover, and J. E. Campbell. 1980. *Risk Methodology for Geologic Disposal of Radioactive Waste: Small Sample Sensitivity Analysis Techniques for Computer Models, with an Application to Risk Assessment*. NUREG/CR-1397.

International Commission on Radiological Protection. 1979. "Limits for Intakes by Workers." ICRP Publication 30, Part 1, *Annals of the ICRP*, Vol. 2, No. 314, Pergamon Press.

Kahn, B., R. L. Blanchard, H. E. Kolde, H. L. Krieger, S. Gold, W. L. Brinck, W. J. Averett, D. B. Smith, and A. Martin. 1971. *Radiological Surveillance Studies at a Pressurized Water Nuclear Power Reactor*. U. S. Environmental Protection Agency, USEPA, RD 71-1.

Killough, G. G. 1982. *Derivation of Dose Conversion Factors for Tritium*. NUREG/CR-2523, ORNL-5853.

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.* 34:141-159.

Kocher, D. C., R. W. Leggett, D. E. Dunning, Jr., M. T. Ryan, and K. F. Eckerman. 1980. *Uncertainties in the Calculation of Long-Term Collective Dose and Health Effects - A Preliminary Assessment*. ORNL/NUREG/TM-378, NUREG/CR-1303.

Kocher, D. C. 1981. *Radioactive Decay Data Tables, A Handbook of Decay Data for Application to Radiation Dosimetry and Radiological Assessments*. DOE/TIC-11026.

Little, C. A. and C. W. Miller. 1979. *The Uncertainty Associated with Selected Environmental Transport Models*. ORNL-5528.

Mankin, J. B., R. V. O'Neill, H. H. Shugart, and B. W. Rust. 1977. "The importance of validation in ecosystem analysis." In *New Directions in the Analysis of Ecosystems* (G. S. Innis, Ed.). Simulation Councils Proceedings Series, Vol. 5(1):63-71.

Matthies, M., K. Eisfeld, H. Paretzke, E. Wirth. 1981. "Stochastic calculations for radiation risk assessment: A Monte Carlo approach to the simulation of radiocesium transport in the pasture-cow-milk food chain." *Health Phys.* 40:764-769.

McKay, M. D., W. J. Conover, and R. J. Beckman. 1979. "A comparison of three methods for selecting values of input variables in the analysis of output from a computer code." *Technometrics* 21(2):239-245.

Miller, C. W. 1979. "Validation of a Model to Predict Aerosol Interception by Vegetation," pp. 251-361 in *Proceedings of an International Symposium on Biological Implications of Radionuclides Released from Nuclear Facilities*, Vol. II, Vienna, Austria, March 26-30, 1979, IAEA-SM-237.

Miller, C. W. 1980. "An analysis of measured values for the fraction of a radioactive aerosol intercepted by vegetation." *Health Phys.* 38(4): 705-712.

Miller, C. W. and F. O. Hoffman. 1981. "An analysis of reported values of the environmental half-time for radionuclides deposited on the surfaces of vegetation." In *International Symposium on Migration in the Terrestrial Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle.* Knoxville, Tennessee, July 27-31, 1981. IAEA-SM-257/63.

Murphy, C. E., Jr. and M. M. Pendergast. 1979. *Behaviour of Tritium in the Environment*, p. 361. STI/PUB/498 IAEA, Vienna.

National Council on Radiation Protection and Measurement. 1979. *Tritium in the Environment.* NCRP Report No. 62, Washington, D.C.

Neter, J., W. Wasserman, and G. A. Whitmore. 1978. *Applied Statistics.* Allyn and Bacon, Inc., Boston, Mass.

Ng, Y. C. 1981. *A Review of Transfer Factors for Assessing the Dose from Radionuclides in Agricultural Products.* UCRL-85138, Rev. 1 (also submitted to *Nuclear Safety*).

Ng, Y. C., C. S. Colsher, and S. E. Thompson. (In preparation). *Soil-to-Plant Concentration Factors for use in Radiological Assessments.* Lawrence Livermore National Laboratory Report.

Ng, Y. C., C. S. Colsher, and S. E. Thompson. (In preparation). *Transfer Coefficients for Assessing the Dose from Radionuclides from Meat and Eggs.* Lawrence Livermore National Laboratory Report.

O'Neill, R. V., R. H. Gardner, and J. B. Mankin. 1980. "Analysis of parameter error in a nonlinear model." *Ecological Modelling* 8:297-311.

O'Neill, R. V., R. H. Gardner, F. O. Hoffman, and G. Schwarz. 1981. "Parameter uncertainty and estimated radiological dose to man from atmospheric ^{131}I releases: a Monte Carlo approach." *Health Phys.* 40:760-764.

Reiter, E. R. 1971. "Carbon dioxide." In *Atmospheric Transport Processes Part 2: Chemical Tracers,* pp. 95-112. AEC Critical Review Series.

Rohwer, P. S. 1976. *Relative Radiological Importance of Environmentally Released Tritium and Krypton-85.* IAEA-SM-172/76, p. 79-90.

Rubenstein, R. Y. 1981. *Simulation and the Monte Carlo Method.* John Wiley and Sons, Inc., New York

Rupp, E. M. 1979. "Annual dietary intake and respiration rates, U_{ap}." In *A Statistical Analysis of Selected Parameters for Predictions of Food Chain Transport and Internal Dose of Radionuclides* (F. O. Hoffman and C. F. Baes III, Eds.). ORNL/NUREG-282.

Rupp, E. M., F. L. Miller, and C. F. Baes III. 1980. "Some results of recent surveys of fish and shellfish consumption by age and region of U. S. residents." *Health Phys.* 39(2):165-175.

Rupp, E. M. 1980. "Age-dependent values of dietary intake for assessing human exposures to environmental pollutants." *Health Phys.* 39:151-163.

Schubert, J., A. Brodsky, and S. Tyler. 1967. "The lognormal function as a stochastic model of the distribution of Strontium-90 and other fission products in humans." *Health Phys.* 13:1187-1204.

Schwarz, G. 1980. "General aspects of accuracy in dose calculations." In *Accuracy in Dose Calculations for Radionuclides Released to the Environment* (K. H. Lindakers and H. Bonnenberg, Eds.). GUW, Aldenhoven.

Schwarz, G. and D. E. Dunning, Jr. (in press). Imprecision in estimates of dose from ingested ^{137}Cs due to variability in human biological characteristics. *Health Physics.*

Schwarz, G. and F. O. Hoffman. 1981. "Imprecision of dose predictions for radionuclides released to the environment: an application of a Monte Carlo simulation technique." *Environ. Int.* 4:289-297.

Schwibach, J., H. Riedel, and J. Bretschneider. 1978. *Investigations into the Emission of Carbon-14 Compounds from Nuclear Facilities, Its Measurement and the Radiation Exposure Resulting from the Emissions.* Commission of the European Communities, Luxembourg.

Shaeffer, D. L. 1980. "A model evaluation methodology applicable to environmental assessment models." *Ecological Modelling* 8:275-295.

Shor, R. W. (Private communication). Health and Safety Research Division, Oak Ridge National Laboratory.

Shor, R. W. and D. E. Fields. 1979. "Animal feed consumption rate, Q_F," and "The fraction of total feed composed of fresh forage, f_s," and "The fraction of the year fresh forage is utilized, f_p." In *Statistical Analysis of Selected Parameters for Predictions of Food Chain Transport and Internal Dose of Radionuclides* (F. O. Hoffman and C. F. Baes III, Eds.), ORNL/NUREG-282, NUREG/CR-1004.

Shor, R. W., C. F. Baes III, and R. D. Sharp. (1982). *Agricultural Production in the United States by County: A Compilation of Information from the 1974 Census of Agriculture for use in Terrestrial Food Chain Transport and Assessment Models.* ORNL-5768.

Snedecor, G. W. and W. G. Cochran. 1967. *Statistical Methods.* Iowa State University Press, Ames, Iowa.

Till, J. E., H. R. Meyer, E. L. Etnier, E. S. Bomar, R. D. Gentry, G. G. Killough, P. S. Rohwer, V. J. Tennery, and C. C. Travis. 1980a. *Tritium — An Analysis of Key Environmental and Dosimetric Questions.* ORNL/TM-6990.

Till, J. E., E. L. Etnier, and H. R. Meyer. 1980b. "Updating the tritium quality factor – the argument for conservatism." In *Proceedings of American Nuclear Society Topical Meeting*. Dayton, Ohio, April 29 – May 1, 1980.

U. S. Nuclear Regulatory Commission. 1975. Numerical guides for design objectives and limiting conditions for operation to meet the criterion "as low as practicable" for radioactive material in light-water-cooled nuclear power reactor effluents. Opinion of the Commission. Docket No. RM-50-2.

U. S. Nuclear Regulatory Commission. 1977. "Calculation of annual doses to man from routine releases of reactor effluents for the purpose of evaluating compliance with 10 CFR 50, Appendix I." *Regulatory Guide 1.109*. Rev. 1.

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.

Vogt, K. J. 1979. *Behaviour of Tritium in the Environment*, p. 521, STI/PUB/498 IAEA, Vienna.

APPENDIX A

DOSIMETRY OF RADIOSTRONTIUM EXPOSURE

K. F. Eckerman

A.1. Introduction

The purpose of this Appendix is to summarize the current approach to the dosimetry of radiostrontium (a bone seeker) and to set forth estimates of the dose conversion factor, i.e., committed dose equivalent per unit intake, and its variability for the ingestion of ^{90}Sr. The dosimetric approach to bone seeking radionuclides has been revised in recent recommendations of the International Commission on Radiological Protection (ICRP) as contained in Publication 30 (ICRP, 1979a). These recommendations supersede those contained in ICRP Publication 2 (ICRP, 1960) upon which the dose conversion factors of Regulatory Guide 1.109 (USNRC, 1977) are based.

Strontium is a member of the alkaline earth chemical family and is metabolized by the body in a manner similar to calcium. The major site of deposition and retention of ^{90}Sr in the body is the bone mineral matrix of the skeleton. Because ^{90}Sr (T 1/2 = 28.6 y) and its daughter ^{90}Y (T 1/2 = 64.1 h) both are beta emitters, the tissues which are subject to significant irradiation are those of the skeleton.

A.2. Tissues of the Skeleton

The skeleton is a complex structure composed of bone mineral (5 kg), yellow (fatty) marrow (1.5 kg), red (active) marrow (1.5 kg), and assorted connective tissues (2 kg) (ICRP, 1975). The numerical values given here and throughout the text are for the adult. Bone can be divided into two categories, structural bone and metabolic bone. "Structural" is in reference to the mechanical function of the skeleton, and "metabolic" refers to the role the bone mineral plays in regulating the extracellular calcium levels, particularly blood plasma. In this text, bone refers to the skeletal mineral, i.e., the 5 kg of mineral.

In terms of the radiosensitivity of the skeleton, there now is general agreement that the radiosensitive tissues are the hemotopoietic

stem cells of the active (red) marrow and the osteogenic cells, particularly those on the endosteal surfaces of bone (ICRP, 1968). Developing red blood cells are found in various stages of maturation within the active marrow. Thus the active marrow is of primary concern as a target tissue with respect to leukemia induction. The osteogenic cells are the precursors of the cells involved in the formation of new bone (osteoblasts) and the resorption of bone (osteoclasts) and are of concern as a target tissue with respect to induction of bone cancer. Note that bone is not considered a tissue at risk; only soft tissue regions of the skeleton are considered to be at risk.

In the past, the dose equivalent associated with bone seekers was averaged over the entire bone; a mass of 7 kg was assumed in Publication 2 (ICRP, 1960). The effective energy deposited in bone by the nuclide of interest was compared with that of radium. In estimating the effective energy, a modifying factor of 5 was applied to all non-radium isotopes to account for lack of knowledge of the deposition pattern and its relationship to tissues at risk. In present day evaluations of the irradiation of the two tissues considered at risk, it is necessary to classify the bone into two regions, trabecular and cortical bone.

In the mature skeleton, two bone types are reasonably distinct in terms of both appearance and their retention of bone seekers (ICRP, 1968; ICRP, 1973; and ICRP, 1975). Cortical bone is the hard compact bone found largely in the shafts of the long bones. This bone type constitutes about four-fifths of the skeletal mineral, i.e., 4 kg (ICRP, 1975). The dominant microscopic structure of cortical bone is the Haversian system. Each Haversian system centers around a canal or space containing blood vessels, osteoblasts (bone-forming cells), and undifferentiated cells. These canals are typically 50 microns in diameter (ICRP, 1975) and with supporting channels serve to supply nutrients to the interior of the skeleton. The soft tissues lining the Haversian system are a component of the endosteal tissue considered at risk for bone cancer (ICRP, 1968; ICRP, 1979a).

Trabecular bone, sometimes referred to as cancellous bone, is the soft spongy bone composed of an apparently fragile lattice-work which

lies to the interior of the flat bones and the ends of the long bones of the body. The interlacing splinters of bone mineral (trabeculae) form cavities in which the active (red) marrow is found. By mass, trabecular bone comprises about one-fifth (1 kg) of the skeletal mineral (ICRP, 1975). As the active marrow is contained in trabecular bone this mineral region represents the major source region from which beta particles irradiate the active marrow.

A.3. Dosimetric Formulation

The ^{90}Sr committed dose equivalent per unit intake, i.e., the dose conversion factor, for the two radiosensitive tissues of the skeleton can be written as (ICRP, 1979a):

$$H_{50}\ (\text{Red marrow}) = K \cdot U_{50}\ (\text{Trabecular bone}) \cdot \text{SEE}\ (\text{Red marrow} \leftarrow \text{Trabecular bone}) \qquad \text{(A-1)}$$

and

$$H_{50}\ (\text{Bone surface}) = K[U_{50}\ (\text{Cortical bone}) \cdot \text{SEE}\ (\text{Bone surface} \leftarrow \text{Cortical bone}) + U_{50}\ (\text{Trabecular bone}) \cdot \text{SEE}\ (\text{Bone surface} \leftarrow \text{Trabecular bone})],$$

where

$H_{50}(T)$ = committed dose equivalent for tissue T per unit intake of ^{90}Sr,

$U_{50}(S)$ = number of nuclear transformations of ^{90}Sr occurring in source region S over a period of fifty years following the intake of a unit activity,

$SEE(T \leftarrow S)$ = specific effective energy absorbed in the target region T per nuclear transformation of ^{90}Sr in source region S, and

K = any constant required by the units of U and SEE.

The reader will note that the dosimetric expressions have been somewhat simplified in that we have assumed ^{90}Y will be in equilibrium with its parent (ICRP, 1979b). The numerical value of SEE should then include the energy contribution from both ^{90}Sr and ^{90}Y.

The estimation of the dose conversion factor and its possible variability centers around two derived parameters; U and SEE. The former representing the metabolism of strontium and the latter involving the anatomical relationship between the source and target regions.

A.4. Strontium Metabolism

Strontium, as a member of the alkaline earth chemical family, is metabolized by the body in a manner similar to calcium and the widely studied radionuclide radium. In the 1959 recommendations of the ICRP (ICRP, 1960) the uptake of strontium from the gastrointestinal tract to blood was characterized by a fractional value of 0.3 (the f_1 parameter) with 95% of the strontium entering blood assumed to be deposited in the skeleton where it was retained with a biological half-time of 1.8×10^4 day (50 yr). The effective halftime for retention is then 6.4×10^3 day. It was known at that time that the retention of bone-seeking radionuclides often could not be reasonably characterized by a single exponential term, i.e., a constant fraction being eliminated per unit time, and in an Appendix to Publication 2, the use of a power function was investigated. Application of these functions yielded MPC (maximum permissible concentration) values that were higher (less restrictive) than the single exponential model by a factor of 6 to 8 depending on the parameter values of the power function (ICRP, 1960).

Following the issuances of Publication 2, Committee 2 of the ICRP formed a Task Group to develop further the quantitative information on the distribution and retention of alkaline earth radionuclides in bone. This task group proposed a metabolic model, ICRP Publication 20, for these radionuclides which is in accordance with all the major well established biological data for the chemical family (ICRP, 1973). The alkaline earth metabolic model is probably the most comprehensive metabolic model used in radiation dosimetry. The development of the model,

choice of parameter values, and discussion of various tests of the model are contained in ICRP Publication 20 (1973). The accuracy with respect to existing data was stated to be about 20%.

The number of nuclear transformations occurring in a source region per unit activity entering blood is given by the integral of the retention function for the region. The integral of the trabecular (cancellous) and cortical (compact) bone retention functions for 50 years following a unit input to the blood are given (see Table 34 of ICRP Publication 20, 1973) as 157 and 398 days, respectively. This is equivalent to a 555 μCi-day residence in bone per μCi entering the blood. The corresponding value for the single exponential retention model of Publication 2 (ICRP, 1960) is 7560 μCi-day per μCi entering blood. Thus the single exponential model presented in ICRP Publication 2 (ICRP, 1960) and employed in Regulatory Guide 1.109 (USNRC, 1977) grossly overestimates the retention of ^{90}Sr, a factor greater than ten relative to the more sophisticated alkaline earth model.

The number of nuclear transformations occurring in source regions of bone per unit activity ingested is given by f_1 times the above integral values. Thus,

$$U_{50}\ (\text{Trabecular bone}) = \frac{0.3\ \mu\text{Ci to blood}}{\mu\text{Ci-ingested}} \times 157\ \text{days}$$

$$= 47\ \frac{\mu\text{Ci-day}}{\mu\text{Ci-ingested}}$$

or 4.1×10^6 nuclear transformations/Bq-ingested, and

$$U_{50}\ (\text{Cortical bone}) = \frac{0.3\ \mu\text{Ci to blood}}{\mu\text{Ci-ingested}} \times 398\ \text{day} = 120\ \frac{\mu\text{Ci-day}}{\mu\text{Ci-ingested}}$$

or 1.0×10^7 nuclear transformation/Bq-ingested.

In the supplement to ICRP Publication 30, Part 1 (ICRP, 1979b), U_{50} (Trabecular bone) is given as 4.6×10^6 and U_{50} (Cortical bone) as 1.1×10^7 nuclear transformations/Bq-ingested. The difference noted here is a result of reevaluation of the mathematical expressions of the alkaline earth model not a result of change in parameter values and are within the 20% error estimate given in Publication 20 (ICRP, 1975).

A.5. Estimation of energy deposition

The estimation of energy deposition in the target regions at risk per nuclear transformation occurring in source regions of bone is a complex problem due to the geometric relationships between these regions. The trabeculae (mineral region) and the marrow cavities they form in trabecular bone cannot be represented by simple solid geometric forms. To derive an estimate of the energy deposition in the marrow cavities, one considers the potential path a beta particle may take in its traversal of trabeculae and marrow cavities. The particle's energy upon entering a cavity will depend on its initial energy and that disipated in traveling to the cavity. The amount of energy deposited within the cavity is dependent on the path the particle takes through the cavity and the energy it had on entrance. If the particle has sufficient energy to traverse the cavity it will encounter the trabeculae on the "farside." Then, if energetically possible, it will traverse and enter another cavity. Calculations of the energy deposition have been facilitated by compilations of pathlength distributions through trabeculae and cavities for various bones of the body (Beddoe, Darley, and Spiers, 1976). Using these data and Monte Carlo sampling techniques, the flight of beta particles can be simulated, as outlined above. The results of such calculations by Whitwell and Spiers (1976) are shown in Table A-1 along with the discriptive parameters for the various bones.

As shown in Table A-1, Whitwell and Spiers normalize their Monte Carlo estimates of the dose for the endosteal (D_S) and marrow (D_M) tissues to the absorbed dose in a small soft tissue inclusion within mineral bone, i.e., the Bragg-Gray dose denoted as D_o. The Bragg-Gray dose D_o is given as

Table A-1. Skeletal and dosimeter parameters for ^{90}Sr-^{90}Y

Bone	Mean path-length (μm)		S/V[a]	D_s/D_o[b]	D_m/D_o[c]
	Cavities	Trabeculae	cm^2/cm^3		
Parietal	389	514	78	0.690	0.653
Rib	1706	265	185	0.245	0.182
Iliac crest	907	232	172	0.365	0.315
Cervical vertebra	914	271	166	0.285	0.229
Lumbar vertebra	1237	244	197	0.332	0.295
Head of femur	1156	220	191	0.282	0.235
Neck of femur	1656	320	154	0.281	0.218

[a]Surface to volume ratio.

[b]Ratio of the dose to the endosteal layer to the dose to a small tissue inclusion in mineral bone.

[c]Ratio of the mean dose absorbed to marrow to the dose to a small tissue inclusion in mineral bone.

$$D_o = kD_b, \qquad \text{(A-2)}$$

where

k = ratio of the mass stopping power of soft tissue and bone for the beta particle, and

D_b = equilibrium dose to bone, i.e., the energy absorbed per unit mass being equal to the energy emitted per unit mass.

The data of Table A-1 indicate the extent these ratios or dose factors vary among the various bones of the body. Note that with the exception of the parietal bone of the skull the D_s/D_o and D_m/D_o values are similar. The parietal bone is not typical in that the mean pathlength through the trabeculae is greater than that through the cavities while the inverse holds for all other tissues.

The data presented in Table A-1 can be used to derive estimates of the SEE parameter as outlined in Publication 30 (ICRP, 1979a). The results of that effort yields the following SEE values for ^{90}Sr-^{90}Y (ICRP, 1979b):

$$\text{SEE (Red marrow} \leftarrow \text{Trabecular bone)} = 2.7 \times 10^{-4} \text{ MeV/g-n.t.},$$

$$\text{SEE (Bone surface} \leftarrow \text{Trabecular bone)} = 2.3 \times 10^{-4} \text{ MeV/g-n.t.},$$

$$\text{SEE (Bone surface} \leftarrow \text{Cortical bone)} = 1.4 \times 10^{-4} \text{ MeV/g-n.t.},$$

where n.t. denotes nuclear transformation. Using these data in conjunction with the number of nuclear transformation data per intake ingested, then application of Eq. (A-1) with $K = 1.6 \times 10^{-10}$ joules-g/MeV-kg (note Sv = joule/kg) yields the committed dose equivalent per unit activity ingested:

$$D \text{ (Red marrow)} = 1.9 \times 10^{-7} \text{ Sv/Bq } (7.2 \times 10^{-4} \text{ mrem/pCi}),$$

$$D \text{ (Bone surface)} = 4.1 \times 10^{-7} \text{ Sv/Bq } (1.6 \times 10^{-3} \text{ mrem/pCi}).$$

Other organs of the body experience a committed dose equivalent per unit intake ranging from 5.7×10^{-9} to 1.5×10^{-9} Sv/Bq (Eckerman, Ford, and Watson, 1981).

A.6. Variability Associated with Sr-90 Dose Conversion Factor

Variability in the ^{90}Sr dose conversion factor for red marrow and the endosteal tissues, i.e., the committed dose equivalent per unit intake, arises as a consequence of variability in both metabolism and energy deposition. The latter reflects the anatomical relationships between the bone mineral matrix (source region) and the target tissues of interest. Variability in metabolism is largely associated with uncertainties in the fractional absorption to blood of the ingested activity, the f_1 parameter, and the retention within the bone.

Calculation of the energy deposition within the endosteal tissues and the marrow cavities is a complex problem. Information on the microscopic details as to the distribution of pathlength which beta particles might follow in traversing the mineral and soft tissue regions are required. Compilation of such information has been carried out by the Bone Dosimetry Research Unit at Leeds University in Great Britain (Darley, 1968; Beddoe, Darley, and Spiers, 1976; and Beddoe, 1977), however, as far as is known, there is no equivalent set of data available for any direct comparison. Beddoe and Spiers (1979) estimate that the path-length distributions have standard errors typically of 5%. They report the results of a series of measurements carried out on lumbar vertebra samples from a number of individuals. The coefficient of variation on the marrow dose factor, D_m/D_o, varied between 11 and 20% depending on the energy of the beta particles, whereas the endosteal dose factor, D_s/D_o, varied between 1 and 12%, again dependent on energy (Beddoe and Spiers, 1979). These data provide some indication of the variability, however they exist only for a single bone type.

While an adequate data base does not exist for assessing the variability, other observations suggest it probably is small. The beta particles emitted in the ^{90}Sr-^{90}Y decay are quite energetic, extending up to an endpoint energy of 2.28 MeV. The range of such particles in soft tissue is about one-half cm and thus one can expect that structural features of linear dimension on the order of 100's of microns do not significantly alter the energy deposition pattern. One also notes that the dose factors, D_m/D_o and D_s/D_o, for ^{90}Sr-^{90}Y are numerically the same for either a volume or surface distribution of the activity (Eckerman, in preparation). Thus it appears reasonable to suggest that for ^{90}Sr-^{90}Y the variability in energy deposition will not be a significant contributor to uncertainty in the committed dose equivalent per unit intake.

The question of possible bias in the estimation of energy deposition however needs to be addressed. Recent calculations of the energy deposition in the endosteal layer of trabecular bone (Eckerman, in preparation) indicates that in the earlier work of Whitwell and Spiers (1976) the energy deposition was underestimated. In their study, the assumption had to be made that the energy deposited per beta particle traversing the endosteal region was the same as the energy deposited by a particle traversing an average path through the region. Eckerman (in preparation), formulating the calculational procedure in a different manner, was able to simulate the various paths beta particles could take through this region. Using the characterization of the bones from the group at Leeds, the results of his calculations indicate about a factor of two higher energy deposition in the endosteal region of trabceular bone then that of Whitwell and Spiers. These results have been discussed with Dr. Spiers by Eckerman. Note the above is with respect to the endosteal region of trabecular bone, dose estimates for the endosteal region of cortical bone are not subject to this revision. The overall effect on the committed dose equivalent per unit intake for the endosteal tissue, that is the endosteal tissues of both cortical and trabecular bone, will be about a 50% increase in the value recommended here.

The metabolism of ^{90}Sr in the body is strongly influenced by metabolic processes associated with calcium. The extent of this association is such that Leggett, Eckerman and Williams (in press) were able to construct an age-dependent strontium model using only physiological, anatomical, and calcium metabolic data. The calcium content of body fluids, in particular blood plasma, must be maintained within rather close tolerances; 9 to 11 mg calcium per 100 ml of plasma is required for proper muscle action. These levels are not the result of a similar tolerance on daily dietary intake but reflect the regulation of both the rate of resorption of skeletal mineral and the intestinal uptake of calcium.

The major source of uncertainty in metabolic models generally lies with the f_1 parameter, the fraction of the radionuclide transferred from the gastrointestinal tract to blood. As noted above, in ICRP Publication 30 (ICRP, 1979a) an f_1 value of 0.3 was used for soluble strontium compounds in setting the occupational guidance. A value of 0.2 was suggested in the alkaline earth metabolic model of ICRP Publication 20 (ICRP, 1973), based on the observed levels in human bone of ^{90}Sr from weapon's fallout. This value is more consistent with environmental concerns then the value assumed in ICRP Publication 30. As noted by Leggett, Eckerman, and Williams (in press), during periods of skeletal growth or decline in mineralization with age, the gastrointestinal uptake may be enhanced or reduced. Assuming young adults and a typical diet supplying about a gram per day of calcium, the bodies regulatory mechanisms limit the variability in the f_1 parameter.

Retention of strontium in the mineral matrix of the skeleton is influenced by the continuously occurring remodeling of the skeleton. The data which might be reviewed to assess variability in retention is limited. However, a wealth of human data has been compiled for the continuous dietary exposure to fallout ^{90}Sr. In order to use these data to examine variability in retention, one must employ a metabolic model due to the continuous and time dependent nature of the intake. Given this data base it is more appropriate to examine the observed variability in the skeletal concentrations and contribute it to metabolic processes.

Schubert, Brodsky, and Tyler (1967) and more recently Cuddihy, McClellan, and Griffith (1979) examined the data on ^{90}Sr concentration in human bone in efforts to understand the variability in the exposed population. These data were found to be lognormally distributed with a geometric standard deviation ranging from 1.4 to 1.6. This variability would include components arising from dietary preferences, variability of ^{90}Sr transfer in food pathways and that arising from metabolism. Since the former components are part of variability in environmental transport, not the dose conversion factor, we have assumed that the lower range of the observed variation might be assigned to the metabolism. Furthermore, since the variability in the committed dose equivalent per unit intake is suggested to be dominanted by the metabolism we assign the above variability to the dose factor. For purposes of assessing the uncertainty in assessment models, we recommend that the numerical values for the committed dose equivalent per intake for the active red marrow and the endosteal region cited above be taken as the geometric mean and that a lognormal geometric standard deviation (s_g) of 1.4 be assigned to characterize the distribution.

A.7. ^{90}Sr dose factors of NRC Regulatory Guide 1.109

Regulatory Guide 1.109 (USNRC, 1977) presents no value for the dose factors of the active marrow and endosteal cells of the bone because the dosimetric approach employed did not address these tissues. The numerical value for the ^{90}Sr ingestion dose conversion factor for the tissue "bone" (Table 2) is greater than either the endosteal or active marrow values cited on page 69. This is largely due to an overestimation of ^{90}Sr retention in the "bone" and the use of a modifying factor of 5 in computing the effective energy for the ^{90}Sr dose conversion factor used in Regulatory Guide 1.109.

REFERENCES FOR APPENDIX A

Beddoe, A. H., P. J. Darley, and F. W. Spiers. 1976. "Measurements of Trabecular Bone Structure in Man." *Phys. Med. Biol.* 21:589-607.

Beddoe, A. H. 1977. "Measurements of the Microscopic Structure of Cortical Bone." *Phys. Med. Biol.* 22:298-308.

Beddoe, A. H. and F. W. Spiers. 1979. "A Comparative Study of the Dosimetry of Bone-Seeking Radionuclides in Man, Rhesus Monkey, Beagle, and Miniature Pig." *Rad. Res.* 80:423-439.

Cuddihy, R. G., R. O. McClellan, and W. C. Griffith. 1979. "Variability in Target Organ Deposition Among Individuals Exposed to Toxic Substances." *Toxicol. Appl. Pharmacol.* 49:179-87.

Darley, P. J. 1968. "Measurement of Linear Pathlength Distributions in Bone and Bone Marrow Using a Scanning Technique." In *Proceedings of the Symposium on Microdosimetry*, Ispra, Italy, 509-526.

Eckerman, K. F. (In preparation). *Absorbed Fraction Data for Radiosensitive Tissues of the Skeleton: Part 1, Beta Emitters in Trabecular Bone.*

Eckerman, K. F., M. R. Ford, and S. B. Watson. 1981. Internal Dosimetry Data and Methods of ICRP-Part 2, Vol. I: Committee Dose Equivalent and Secondary Limits. ORNL/NUREG/TM-433/VI.

International Commission on Radiological Protection. 1960. Report of Committee 2 on Permissible Dose for Internal Radiation. *ICRP Publication 2.* Pergamon Press, Oxford.

International Commission on Radiological Protection. 1968. A Review of the Radiosensitivity of the Tissues in Bone. *ICRP Publication 11,* Pergamon Press.

International Commission on Radiological Protection. 1973. Task Group on Alkaline Earth Metabolism in Adult Man. *ICRP Publication 20,* Pergamon Press.

International Commission on Radiological Protection. 1975. Task Group Report on Reference Man. *ICRP Publication 23,* Pergamon Press.

International Commission on Radiological Protection. 1977. ICRP Publication 26, *Annals of the ICRP,* Vol. 1, No. 3, Pergamon Press.

International Commission on Radiological Protection. 1979a. Limits for Intakes by Workers. ICRP Publication 30, Part 1, *Annals of the ICRP*, Vol. 2, No. 3/4, Pergamon Press.

International Commission on Radiological Protection. 1979b. Limits for Intakes by Workers. ICRP Publication 30, Supplement to Part 1, *Annals of the ICRP*, Vol. 3, Nos. 1-4, Pergamon Press.

Leggett, R. W., K. F. Eckerman, and L. R. Williams. (In press.) "Strontium-90 in Bone: A Case Study in Age-dependent Dosimetric Modeling," to appear in *Health Physics*.

Schubert, J., A. Brodsky, and S. Tyler. 1967. "The Log-normal Function as a Stochastic Model of the Distribution of Strontium-90 and Other Fission Products in Humans." *Health Phys.* 13:1187-1204.

Spiers, F. W., J. R. Whitwell, and A. H. Beddoe. 1978. "Calculated Dose Factors for the Radiosensitive Tissues in Bone Irradiated by Surface-Deposited Radionuclides." *Phys. Med. Biol.* 23(3):481-494.

U.S. Nuclear Regulatory Commission. 1977. *Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR 50, Appendix I*, Regulatory Guide 1.109, Revision 1.

Whitwell, J. R. and F. W. Spiers. 1976. "Calculated Beta-ray Dose Factors for Trabecular Bone." *Phys. Med. Biol.* 21(1):16-38.