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**A REVIEW OF UNCERTAINTY ESTIMATES
ASSOCIATED WITH MODELS FOR ASSESSING THE
IMPACT OF BREEDER REACTOR RADIOACTIVITY
RELEASES**

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CONTENTS

	<u>Page</u>
LIST OF TABLES	v
LIST OF FIGURES	v
SI UNITS	vi
HIGHLIGHTS	ix
ACKNOWLEDGMENTS	xi
1. INTRODUCTION	1
2. DOSE ESTIMATION	2
3. SOURCE TERM	5
4. ENVIRONMENTAL CONCENTRATIONS	5
4.1 Air Concentration	5
4.2 Ground Concentration	10
4.3 Surface Water Concentration	11
4.3.1 Model without sorption	12
4.3.2 Models with sorption	13
4.4 Concentrations in Food	14
4.4.1 Terrestrial food chain model	14
4.4.2 Aquatic food chain model	15
5. USAGE FACTORS	16
6. DOSE CONVERSION FACTORS	19
6.1 Internal Dosimetry	20
6.2 External Dosimetry	21

	<u>Page</u>
7. UNCERTAINTY IN DOSE TO MAN	22
7.1 Analysis of Predictive Uncertainties Based on Estimated Input Parameter Variability	22
7.2 Other Uncertainty Estimates	28
8. SUMMARY AND CONCLUSIONS	30
REFERENCES	36
APPENDIX A	51
APPENDIX B	57
APPENDIX C	63

LIST OF TABLES

	<u>Page</u>
1. An estimate of the uncertainty associated with prediction: from Gaussian plume atmospheric dispersion model	7
2. Comparison of computed and measured radionuclide concentrations in finfish and shellfish in the outfalls of two nuclear power stations (Adopted from Feldt and Kaye ⁸⁸) . . .	17
3. Estimates of the uncertainty associated with the annual rate of consumption by adults of various food products . . .	18
4. A summary of studies of the uncertainty in the dose to man based on an analysis of the variability of the input parameters used in the calculational model	24
5. A summary of studies of the uncertainty in the dose per unit intake based on an analysis of the variability of the input parameters used in the calculational model	27
6. Order-of-magnitude estimates of the uncertainty associated with various dose calculations based primarily on scientific judgement	29

LIST OF FIGURES

	<u>Page</u>
1. Basic pathways by which man may be exposed to radionuclides released to the environment by nuclear facilities (Meyer et al.)	3

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HIGHLIGHTS

As part of the U.S. Department of Energy Liquid Metal Fast Breeder Reactor safety effort, a project is underway at Oak Ridge National Laboratory which has as its objectives the identification of models available for environmental radiological assessments; evaluation of model structure, simplifying assumptions, and data bases; estimation of uncertainties in model output; and, when possible, recommendation of models and parameters which are best suited to particular assessment situations. When needs are identified, recommendations are also made for further environmental and biomedical research.

The purpose of this report is to summarize estimates based on currently available data of the uncertainty associated with radiological assessment models. The models being examined herein are those recommended previously for use in breeder reactor assessments. Uncertainty estimates are presented for models of atmospheric and hydrologic transport, terrestrial and aquatic food-chain bioaccumulation, and internal and external dosimetry. Both long-term and short-term release conditions are discussed.

The uncertainty estimates presented in this report indicate that, for many sites, generic models and representative parameter values may be used to calculate doses from annual average radionuclide releases

when these calculated doses are on the order of one-tenth or less of a relevant dose limit. For short-term, accidental releases, especially those from breeder reactors located in sites dominated by complex terrain and/or coastal meteorology, the uncertainty in the dose calculations may be much larger than an order of magnitude. As a result, it may be necessary to incorporate site-specific information into the dose calculation under these circumstances to reduce this uncertainty. However, even using site-specific information, natural variability and the uncertainties in the dose conversion factor will likely result in an overall uncertainty of greater than an order of magnitude for predictions of dose or concentration in environmental media following short-term releases.

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1. INTRODUCTION

As part of the U.S. Department of Energy Liquid Metal Fast Breeder Reactor safety effort, a project is underway at Oak Ridge National Laboratory which has as its objectives the identification of models available for environmental radiological assessments; evaluation of model structure, simplifying assumptions, and data bases; estimation of uncertainties in model output; and, when possible, recommendation of models and parameters which are best suited to particular assessment situations. When needs are identified, recommendations are also made for further environmental and biomedical research (see Appendix A).

A previous report contains recommendations concerning models and parameter values to be used to assess the impact on man of radioactivity releases to the environment by breeder reactor facilities.¹ The models considered include those developed for the prediction of atmospheric and hydrologic transport, terrestrial and aquatic food-chain bioaccumulation, and internal and external dosimetry. Both long-term and short-term release conditions are discussed. Many of these models and parameter values may also be appropriate for assessing the impact of other nuclear facilities as well as breeder reactors.

Whenever these or any other models are used for radiological assessment purposes, their predictions have some degree of uncertainty associated with them. There are two basic reasons for this uncertainty:²

1. all models are only approximations of reality,
2. the input parameters used in the models are inherently variable.

Such uncertainties are amplified when models and parameter values determined at one site for one averaging time are applied at a different site and/or over a different time period. These uncertainties must be considered whenever the predictions from environmental transport and dosimetry models are used for decision-making purposes.

The purpose of this report is to review previously published estimates of the uncertainty associated with predictions by radiological assessment models. The models examined herein are similar to those recommended previously for use in breeder reactor assessments.¹ No attempt is made in this report to present rigorous substantiation of each of the estimates given. Instead, the reader should refer to the cited literature for more information in any given case.

2. DOSE ESTIMATION

Radionuclide releases to the environment by breeder reactor facilities may result in a dose to man via a variety of routes. The basic pathways of potential exposure are illustrated diagrammatically in Figure 1. In this report we will consider primarily four modes of exposure: inhalation, ingestion, and immersion due to releases to the atmosphere and ingestion of fish as a result of aquatic releases.

The basic equation for the dose to man is⁴

$$R_{ijkn} = C_{ikn} U_{ijn} D_{ijn} \quad (1)$$

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Figure 1. Basic pathways by which man may become exposed to radionuclides released to the environment by nuclear facilities (Meyer et al.).

where

C_{ikm} = the concentration of radionuclide i in collector or accumulator k appropriate for mode of exposure m ;

D_{ijm} = the dose conversion factor for radionuclide i , organ j , and mode of exposure m ;

R_{ijkm} = the dose or dose rate to organ j from radionuclide i associated with collector or accumulator k and mode of exposure m ;

U_m = the exposure time or intake rate (usage) associated with mode of exposure m .

In the sections which follow we will examine the uncertainty associated with each of the three basic components of the dose calculation.

The uncertainties considered in this report will be expressed in two different ways. One expression will be in terms of the range of measured or expected values for the parameterization of interest. This way of expressing the uncertainty has the disadvantage of emphasizing extreme values. However, this type of uncertainty may be most appropriate if one is concerned with calculations of dose to any exposed individual,⁵ such as when considering accidental releases.⁶ The second way of expressing the uncertainty is in terms of the spread of the probability distribution of the values for the quantity of interest (see Appendix B). The variance of the logtransformed data is used for this purpose because it is often appropriate to assume a lognormal probability distribution for many of the quantities of interest in radiological assessments.² This measure of uncertainty may be more appropriate than the range if average or collective doses are of concern,⁵ such as is the case when considering routine radionuclide releases.

3. SOURCE TERM

Within a given pathway, the dose to humans from radionuclide releases to the environment is directly proportional to the amount of radioactivity released. Estimates of the composition and magnitude of various postulated releases from breeder reactors are available.⁷ A basic assumption in the discussions of model uncertainties which follow is that the source term is well defined. In reality, however, the specification of the source term may be the largest single source of uncertainty in the assessment calculation, especially for short-term releases.⁸ Detailed consideration and specification of the uncertainty associated with the source term is beyond the scope of this model evaluation project. However, this uncertainty must also be considered when the results of this project are applied in decision-making processes.

4. ENVIRONMENTAL CONCENTRATIONS

4.1 Air Concentration

One of the principal ways in which radionuclides from breeder reactor facilities reach the environment is via discharges to the atmosphere. Atmospheric dispersion calculations provide estimates of air concentration resulting from these releases. The most frequently used atmospheric dispersion model for estimating a collective dose,⁹ and the one recommended for most breeder reactor assessments,¹⁰ is the Gaussian plume model.¹¹ This model is widely used because¹² 1) it produces results similar to those produced by other models when comparisons are

made to experimental data, 2) mathematical operations are easily performed, 3) it is appealing conceptually, 4) it is consistent with the random nature of turbulence, 5) it is a solution to the Fickian diffusion equation for constant diffusion and wind speed, 6) other so-called theoretical formulas also contain large amounts of empiricism in their final stages, and 7) it is the model most often referred to in government guidebooks, such as USNRC Regulatory Guide 1.111 (Ref. 13). The Gaussian model is often applied when purely theoretical considerations call its use into question. This practice has been accepted when predicted impacts are so small as to diminish the importance of potential errors.¹⁴

The best way to determine the overall uncertainty associated with atmospheric dispersion models such as the Gaussian plume model is to compare their predictions with environmental measurements taken under release conditions similar to those assumed by the model. Such studies need to be conducted under a variety of terrain, release height, and meteorological conditions. Unfortunately, not enough model validation studies have been performed or are likely to be performed to allow for a reliable statistical analysis of the uncertainty associated with the Gaussian plume model.¹⁵⁻¹⁸ However, a number of validation studies have been reviewed, and a summary of their results is presented in Table 1.

The references given in Table 1 represent both individual studies comparing observed and predicted air concentrations and reviews of other such studies. The range given for the ratio of the predicted air concentration to the observed air concentration for each condition is based on a review of all of the literature cited for that condition and is not necessarily representative of any one reference given. The variance was

Table 1. An estimate of the ratio of predicted to observed air concentration: associated with predictions made by the Gaussian plume atmospheric dispersion model under various release conditions

Conditions	Range	Variance of logtrans- formed data	References
Highly instrumented site; ground-level centerline concentration within 10 km of a continuous point source			
ground-level releases	0.3 - 1.2	0.01	19
elevated releases	0.65 - 1.35	0.04	19
maximum air concentration for elevated releases	0.5 - 1.5	0.10	19, 20
Annual average for a specific point, flat terrain, within 10 km downwind of the release point			
	0.5 - 2	0.15	21 - 23
Annual average for a specific point, flat terrain, 10 - 150 km downwind of the release point			
	0.25 - 4	0.64	24 - 29
Specific hour and receptor point, flat terrain, steady meteorological conditions, within 10 km of the release point			
elevated releases without building wake effects	0.1 - 10	1.77	30 - 35
elevated releases with building wake effects	0.01 - 100	7.07	36 - 38

Table 1 (continued)

Conditions	Range	Variance of logtrans- formed data	References
Short-term, surface-level releases with building wake effects using temperature gradient method of estimating atmospheric stability			
wind speeds over 2 m/s	0.7 - 100	2.05	39, 40
wind speeds under 2 m/s	1 - 100	1.77	40
Short-term, surface-level releases without building wake effects using temperature gradient method of estimating atmospheric stability			
wind speeds over 2 m/s	0.3 - 10	1.02	40, 41
wind speeds under 2 m/s	1 - 100	1.77	40, 41
Complex terrain or meteorology (e.g., sea breeze regimes)			
annual average concentrations	0.1 - 10	1.77	42 - 44
short-term releases	0.01 - 100	7.07	18, 44 - 51

calculated by assuming that the given range in each case represents the extremes of a loguniform probability distribution of predicted to observed ratios.⁵²

It can readily be seen from Table 1 that the uncertainty associated with Gaussian plume model predictions is highly dependent on the site and release conditions being considered. The meteorological and terrain condition, specified in the first entry in Table 1 are rather idealistic and seldom occur in the real world. As such, the uncertainty values given represent estimates of the expected maximum accuracy one can achieve with the Gaussian model. For routine release conditions considered over long time periods and over flat terrain, the uncertainties remain relatively small. As the release time becomes shorter and/or the terrain becomes more complex, however, it can be seen that the uncertainty increases. The uncertainty also increases when releases occur under complex site or meteorological conditions, such as building influences or sea breeze regimes. This increase in uncertainty is not unexpected in view of the fact that spatial and temporal changes in air flow induced by such release conditions are not accounted for in the normal straight-line Gaussian model.

In an effort to improve upon the uncertainty results presented in Table 1, a number of other models have been developed which attempt to include many of the physical processes in the atmosphere that can not be explicitly dealt with in a standard Gaussian plume model. Many of these models have been designed especially for use with short-term releases and/or complex terrain and meteorological conditions. These models range from a simple extension of the conventional Gaussian model to account for varying meteorological conditions⁵³ to more complex,

seemingly more realistic, dispersion models employing various numerical techniques.⁵⁴⁻⁵⁹ However, it should be noted that to run properly these models often require a much more extensive input data base than the Gaussian model, a computer with large storage capacity, and a long computer running time for each simulation desired. Such requirements severely limit the practicality of using many of these more complex models in assessment activities. There is also a shortage of results of field testing for these models. Therefore, at the present time a modeling approach built on the Gaussian plume model seems to remain the most appropriate choice for use in most assessments of breeder reactor radioactivity releases.^{15,25,60}

4.2 Ground Concentration

Particulates and reactive gases may deposit on the surface of the earth through the processes of dry and wet deposition. Wet deposition occurs sporadically during specific rain or snow events, but dry deposition processes occur continuously. A number of different measurements of dry deposition processes have been made.^{61,62} Also, a number of different models for estimating plume depletion have been proposed.⁶³ However, during the course of this project no data have been found that are adequate for validating plume depletion processes. For example, recent sodium release tests to the atmosphere^{64,65} have shown that sodium aerosols tend to form agglomerates that fall out rapidly. However, more such tests are needed to establish quantitatively the effect of this mechanism on subsequent downwind exposure to sodium. Also, calculations have been performed in which inclusion of depositional processes in the model predictions improved agreement with the observed air concentrations of fluorescein particles.^{30,66,67} However, no deposition

measurements were taken during the field studies considered in these comparisons. Good agreement has also been reported between three observed and predicted values of ^{131}I concentration on grass, but because of the input data used in the calculation the authors do not consider their methods to constitute a validation study.⁶⁸ Until such data sets become available, one cannot specify the accuracy associated with a calculation of concentration on ground or with an air concentration involving significant plume depletion.

4.3 Surface Water Concentration

In general, atmospheric releases from breeder reactor facilities are expected to contribute a larger fraction of the dose to humans than are releases to surface waters.⁶⁹ One reason for this expectation is that the source term for atmospheric releases is generally higher than the source term for aquatic releases. Also, humans tend to have more interaction with airborne than with waterborne radionuclides. However, variations between sites can be large enough that conditions prevailing at a specific site should be considered when attempting to determine the relative importance of aquatic or atmospheric exposures in the calculation of doses from a given facility.⁶⁹

Often, the dose to man due to radionuclide releases to surface waters is estimated by simply diluting or mixing the effluent stream in the receiving body of water.⁷⁰ Assessors often assume that, with regard to collective doses, such a procedure is conservative with respect to dose estimation; i.e., the procedure will tend to overestimate the actual population dose received by humans. This procedure may not be conservative, however, when estimating maximum individual doses, if the effluent radionuclides are not uniformly distributed in the cross-stream

direction, or if they can be adsorbed onto sediments.

On the basis of previous reviews of aquatic models and their associated uncertainties,^{71,72} Little⁷³ includes a discussion of those models that might be best suited to assess radionuclide releases from breeder reactors to surface water systems when a simple dilution approach has been judged to be inappropriate. The models considered are two-dimensional, either longitudinal-transverse or longitudinal-vertical. The models also differ in their capability to account for sorptive effects on the pollutant. The Yotsukura-Sayre model (longitudinal-transverse) does not consider sorptions, but the SERATRA (longitudinal-vertical) and PETRA (longitudinal-transverse) models do consider sorption and sediment transport. These models are discussed separately below.

4.3.1 Model without sorption

The aquatic transport models of Yotsukura and Cobb⁷⁴ and Yotsukura and Sayre⁷⁵ are concerned with transverse mixing of solutes in streams. These models employ an orthogonal curvilinear coordinate system to describe the geometrical configuration of the channel. This coordinate system, coupled with the flow distribution within it, allows the easy inclusion of the effects of channel irregularities and curvature.

Although both Yotsukura and Cobb⁷⁴ and Yotsukura and Sayre⁷⁵ published curves of observed and predicted concentration profiles, the agreement between predictions and observations cannot be gauged from those data, because only the best curves of prediction were plotted. The "best" prediction curve resulted from varying the value of several input parameters and generating a group of predictions for each parameter value, (i.e., the model was calibrated to fit the observations).

Nevertheless, Yotsukura, Cobb and Sayre felt that this process "verified" their model because the parameters being varied resulted in an average diffusion coefficient that was uniform for a given test and that was bracketed by published values.⁷² This model has also been suggested by the U.S. Nuclear Regulatory Commission⁷⁶ as an appropriate model to use for routine or continuous releases from nuclear power plants when their simpler approaches⁴ are deemed insufficient.

4.3.2 Models with sorption

If the pollutant being assessed is not in a soluble form or has a marked tendency to adsorb to sediments, it may be necessary to use a model which is capable of predicting sediment or particulate contaminant transport. For such an application, Little⁷³ has suggested considering usage of the PETRA model⁷⁷ in spite of the large number of input parameters it requires. PETRA and a similar model, SERATRA,^{78,79} both estimate sediment and pollution concentrations downstream from some release site as a function of time after the release. However, SERATRA estimates concentrations longitudinally and vertically, while PETRA considers longitudinal and lateral (transverse) distribution of the pollutant. These models are both much more complex than the Yotsukura model mentioned above.

The SERATRA model has been used to predict time-dependent longitudinal and vertical distributions of sediments and radionuclides in both the Columbia⁷⁹ and Clinch⁷⁸ rivers. Little and Miller⁷² found the largest discrepancy between observed and predicted concentrations in these studies was the underprediction of the observed concentration by more than a factor of 4, which occurred with ¹³⁷Cs in the Clinch River study. Unfortunately, the small number of field observations presented for each

study makes it difficult to decide whether the predictions are or are not in agreement with the measurements.

To the best of our knowledge, FHTRA has not been applied to any radiological assessments. However, it has been used to predict the transport of sediment and particulate but not dissolved kepone in the James River Estuary of eastern Virginia. The data were published in numerous figures of sediment or kepone concentration as a function of distance downstream in the James River.⁷⁷ Little and Miller⁷⁰ found from examining these curves that, at a maximum, FHTRA underpredicted particulate kepone concentrations by about 40% and underpredicted sediment concentrations by nearly 70%.

4.4 Concentration in Food

4.4.1 Terrestrial food chain model

The terrestrial food chain model most commonly used for assessment purposes is a quasi-equilibrium model which assumes steady-state relationships between various food-chain compartments. Such a model has been recommended for use in breeder reactor assessments.⁸⁰ This model was developed primarily for use in the evaluation of routine emissions of radionuclides, but given appropriate input parameter values it may also be used for calculating the time-integrated concentrations following an accidental release.^{81,82}

Only limited validation data are available for terrestrial transport models.⁸³ Further, only portions of the total pathway are usually considered and only a very few radionuclides have been analyzed. For example, Hoffman⁸⁴ has reviewed published values for the milk concentration to air concentration ratio (C_m/C_a) for iodine. Because of the difficulty in determining the physical and chemical form of the iodine

monitored in air and the degree to which measured concentrations represent equilibrium conditions, the applicability of these values to validation of predicted values of (C_m/C_a) is limited. Hoffman's analysis indicates, however, that the value of $1900 \text{ m}^3 \text{ l}^{-1}$ for C_m/C_a calculated using a given set of generic default parameter values⁴ may overestimate milk concentrations of ^{131}I resulting from weapons fallout or accidental releases containing a mixture of physicochemical forms of ^{131}I and may underestimate milk concentrations resulting from an air concentration composed only of $^{131}\text{I}_2$.

Shaeffer⁸⁵ used weapons fallout data from two different sites in the United States⁸⁶ to examine the grass-to-milk transfer factor for radioiodine. He calculated a chronic release equilibrium value of 0.19 kg l^{-1} for both sites. This value was shown to be an average of 2.5 times greater than the value derived using the default parameters recommended in Regulatory Guide 1.109 (Ref. 4).

4.4.2 Aquatic food chain model

The approach recommended⁸⁷ to predict the transfer of radionuclides through the various pathways from water to the food for man is to lump all processes and mechanisms into a single factor, called the bioaccumulation factor or concentration ratio. Therefore, the predicted concentration of a radionuclide in an aquatic food is simply the product of the radionuclide concentration in water and the bioaccumulation factor, B , for the radionuclide and food organism. As with the terrestrial food chain model, this model can be used to predict a steady-state concentration in an aquatic food organism as a result of a chronic release situation or a time-integrated concentration as the result of an acute release provided that initial conditions are comparable.⁸¹

Values of B have been measured for fish in numerous artificial and natural systems. It has been shown that B may vary as a function of a number of factors, such as the stable element concentration in the receiving waters, the chemical form of the radionuclide released, the experimental procedures employed, and others.^{88,89} As a result, most of the default values of B chosen for use in radiological assessment models in the absence of site-specific information have been intentionally designed to tend to overpredict radionuclide concentrations in aquatic foods. As seen in Table 2, limited comparisons of observed radionuclide concentrations in fish with predictions made using selected default values of B generally support this contention.⁸⁸

5. USAGE FACTORS

As noted in Section 2, for a particular mode of exposure, the dose to man from a given radionuclide is directly dependent on the exposure time or intake rate. Recommended values for usage factors have been presented for adults.⁹⁴ Examination of the literature, however, indicates that any such values for usage factors are subject to a large amount of variability, especially between different age groups within the population and among specific population groups.⁹⁵⁻⁹⁹

A limited number of statistical analyses have been performed in an attempt to quantify the variability associated with various usage factors. The results of these analyses are summarized in Table 3. These results are based on measured values for the various food product consumption rates, but considerable judgement was used in deriving the statistical parameters given.⁹⁸ It is interesting to note that the

Table 2. Comparison of computed and measured radionuclide concentrations in finfish and shellfish in the outfalls of two nuclear power stations (adapted from Feldt and Kaye⁸⁸)^a

Radionuclide	Organism	Bioaccumulation factor ^b	Computed concentration, Bq/Kg ^c	Measured mean concentration, Bq/Kg
Marine				
⁵⁴ Mn	Clam muscle	12,000	155	<0.74
⁶⁰ Co	Clam muscle	600	17	7 ± 1.5
³² P	Fish muscle	30,000	62	<7.4
⁵⁴ Mn	Fish muscle	600	7.8	0.52 ± 0.4
⁶⁰ Co	Fish muscle	100	2.8	0.96 ± 0.7
¹³⁷ Cs	Fish muscle	30	0.93	2.1 ± 1.9
Freshwater				
³ H	Fish muscle	0.9	322	133 ± 59
¹⁴ C	Fish muscle	4,500	7.4	3.7 ± 5.6
³² P	Fish muscle	100,000	37	<7.4
⁵⁴ Mn	Fish muscle	100	1.6	<0.74
⁶⁰ Co	Fish muscle	20	0.41	<0.74
¹³⁷ Cs	Fish muscle	400	7.4	3.7 ± 1.9

^aCompiled by R. L. Blanchard from references 90 and 91.

^bBioaccumulation factors taken from the following references: For shellfish - Reference 92; for finfish - reference 93.

^cComputed concentration = Quantity discharged x Dilution factor x Bioaccumulation factor.

Table 3. Estimates of the uncertainty associated with the annual rate of consumption by adults of various food products

Food	Geometric mean	Variance of logtransformed data	Reference
Freshwater finfish	14 kg/yr	0.59	Derived in (98) from (97)
Leafy vegetables	18 kg/yr	0.23	Derived in (98) from (95) and (96)
Non-leafy vegetables	45 kg/yr^a	0.59	Derived in (98) from (96)
Milk	95 l/yr	0.64	Derived in (98) from (95)
Beef	94 kg/yr	0.25	Derived in (98) from (95) and (96)

^aAssumes that only 0.25 of total consumption is grown at exposure site.

uncertainty associated with these parameters, as expressed by the variance of the logtransformed data, is comparable to that given in Table 1 for annual average air concentrations. Also, it has been suggested⁹⁹ that a large source of uncertainty associated with calculations of external dose is variations in individual habits such as the fraction of time spent outdoors or away from a particular location. It should be noted, however, that many usage factors are relatively easy to measure on a site-specific basis, and hence, it should be possible to reduce the uncertainty in the dose calculation at a given site due to the variability in the usage parameter.

6. DOSE CONVERSION FACTORS

Dose conversion factors, or dose rate factors, are used to relate a radionuclide exposure to an absorbed radiation dose in humans. Such factors are typically derived from mathematical models for a reference individual with specified anatomical, physiological, and metabolic characteristics.¹⁰⁰ Parameters used in these models are often quantified by a single value, with the recognized variability among individuals being ignored.¹⁰¹

It is not practical to directly measure the absorbed dose in the tissues of a living person.⁹⁹ Components of the models used to calculate dose have been compared to experimental observations, thus resulting in "partial validation" of the dosimetric models. However, validation of the entire dosimetric calculation would seem to be precluded.⁹⁹

Separate sets of dose conversion factors are used for different modes of exposure, two of which are discussed separately below.

6.1 Internal Dosimetry

Internal dose conversion factors relate the dose to a particular body organ to the amount of a given radionuclide that enters the body either through inhalation or ingestion. Separate radionuclide-specific dose conversion factors applicable to adults have been recommended for each of these two pathways.¹⁰² Various metabolic models are used to determine the amount of the radionuclide in an organ of interest. These models have been partially tested for selected radionuclides using observations in man and other mammalian species.⁹⁹ The relationship between dose rate and organ burden is based on well founded physical principles and subjected to empirical tests in physical phantoms of the body.⁹⁹

6.2 External Dosimetry

Determination of dose conversion factors for various body organs from an external exposure to photons (gamma and x-ray) and electrons (beta particles and conversion electrons) involves consideration of two facets:

1. characterization of the radiation field,
2. a model combining the field with a description of the human body.⁹⁹

Computational methods based on sound physical principles are available for characterizing the radiation field, but they are often difficult to employ. As a result, idealized distributions of the radiation field are often assumed. For example, an infinitely thin source of infinite spatial extent is often assumed for radionuclides deposited on ground surfaces; likewise, a semi-infinite cloud of uniform spatial concentration

is generally used for airborne radionuclides. These idealizations are designed to bias the resulting dose predictions by a factor of 2 to 5 overprediction in most cases.⁹⁹ It should be noted, however, that the dose to man from a plume of energetic gamma rays passing overhead may be underestimated if the plume has not yet touched ground and a simple dose conversion factor based on the semi-infinite cloud model is being used.¹⁰³

Absorbed dose estimates for various body organs have been developed from Monte Carlo radiation transport calculations using mathematical models of the human body.¹⁰⁴ Results of some such calculations compared favorably to measurements in a physical phantom of the human body.¹⁰⁵ As noted above, however, no validation data in humans exists for the external dose calculation.

Kocher¹⁰⁴ has extensively reviewed the sources of uncertainty associated with use of external dose conversion factors, but he did not attempt to quantify this uncertainty. The uncertainty in the external dose conversion factor is also probably less than the uncertainty in the internal dose conversion factor, too, since little variation in doses from external sources is expected due to individual anatomical differences.⁹⁹ Kocher suggests that the uncertainty in the dose conversion factor is probably less than the uncertainty in the environmental concentration used in the dose calculation.

7. UNCERTAINTY IN DOSE TO MAN

In the previous sections we have considered the uncertainty associated with various environmental transport and dosimetry models and parameters used to assess the impact on man of radionuclides released to the environment by fast breeder reactors. The purpose of this section is to examine the resulting uncertainty in estimates of the dose to man from the combined use of these models and parameters.

7.1 Analysis of Predictive Uncertainties Based on Estimated Input Parameter Variability

It is clear from the results presented in the previous sections that not enough information exists to specify the uncertainty in the calculation of the dose to man on the basis of validation studies alone. Therefore, alternative methods to determine the potential uncertainty in environmental radiation assessments have been developed and applied to dose prediction models.¹⁰⁶⁻¹¹² These methods are based solely on an analysis of the variability of model input parameters (Appendix C) and the resulting impact on the variability in predicted dose. Such methods provide an indication of the variability of potential model predictions and an estimate of the bias due to the selection of single parameter values in assessment models, but the additional bias inherent in the model structure remains untested.

It must be constantly remembered that these analyses rest on three key assumptions:

1. the model's structure is unbiased, so correct input values will give correct output;

2. the parameters in the model are statistically independent and therefore not correlated, or else the correlations among various parameters is known;
3. the parameter values and their estimated distributions are relevant to the true population of parameter values prevailing for the conditions being addressed by the model.

In general, only model validation studies can substantiate or refute the correctness of all of these assumptions.

The results of a number of analyses of the uncertainty in the dose to man based on an analysis of the variability of input parameters are summarized in Table 4. These studies include both numerical and analytical techniques for propagating the input parameter uncertainties through the model. The estimated range given represents the expected interval to include 95% of all possible outcomes predicted by the model. This interval is determined for an assumed lognormal probability distribution.

An important submodel contained in each of the analyses reported in Table 4 is that for internal dosimetry. The uncertainty associated with various internal dose conversion factors has also been analyzed by examining the variability of the input parameters used in the model. These results are summarized in Table 5.

It can be seen in Table 4 that the estimated range in dose is approximately 1 to 2 orders of magnitude in all of the studies referenced. Also, the geometric mean is generally within an order of magnitude of the upper limit of the estimated range. The variance of the logtransformed data is also very similar in all of these studies.

Table 4. A summary of studies of the uncertainty in the dose to man based on an analysis of the variability of the input parameters used in the calculational model

Quantity	Radio-nuclide	Variance of logtrans-formed data ^a	Estimated range ^b	Geometric mean	Reference
Dose rate at age 70 from lifetime exposure to unit activity in soil ($\mu\text{Gy/yr}$)					
lung (inhalation)	^{239}Pu	0.67	0.018 - 0.46	0.09	113
bone (ingestion)	^{239}Pu	4.0	0.0021 - 6.04	0.11	113
Dose rate to thyroid of infants per unit air concentration from ingestion of milk (Sv/yr per Bq/m^3)					
	$^{131}\text{I}_2$	1.0	0.18 - 9.8	1.3	84
Dose rate to thyroid of infants per unit air concentration from ingestion of milk (Sv/yr per Bq/m^3)					
	$^{131}\text{I}_2$	1.3	0.11 - 10.0	1.1	110
Dose equivalent to adults per unit water concentration from ingestion of fish ($\mu\text{Sv per Bq/l}$)					
bone surface	^{90}Sr	3.9	1.3 - 3500	68	98
whole body	^{137}Cs	1.4	17 - 1900	180	98

Table 4 (continued)

Quantity	Radio- nuclide	Variance of logtrans- formed data ^a	Estimated range ^b	Geometric mean	Reference
Dose equivalent to adults per unit ground deposition rate from ingestion of leafy vegetables ($\mu\text{Sv per Bq/m}^2\text{-d}$)					
bone surface	⁹⁰ Sr	1.2	3.1 - 560	73	98
whole body	¹³⁷ Cs	0.77	0.066 - 2.2	0.38	98
Dose equivalent to adults per unit ground deposition rate from ingestion of non-leafy vegetables ($\mu\text{Sv per Bq/m}^2\text{-d}$)					
bone surface	⁹⁰ Sr	1.7	6.7 - 1300	92	98
whole body	¹³⁷ Cs	1.4	0.684 - 10	0.92	98
Dose equivalent to adults per unit ground deposition rate from ingestion of milk ($\mu\text{Sv per Bq/m}^2\text{-d}$)					
bone surface	⁹⁰ Sr	1.7	3.0 - 560	41	98
whole body	¹³⁷ Cs	1.4	0.22 - 23	2.2	98

Table 4 (continued)

Quantity	Radio- nuclide	Variance of logtrans- formed data ^a	Estimated range ^b	Geometric mean	Reference
Dose equivalent to adults per unit ground deposition rate from ingestion of meat ($\mu\text{Sv per Bq/m}^2\text{-d}$)					
bone surface	⁹⁰ Sr	2.1	0.84 - 260	15	98
whole body	¹³⁷ Cs	1.1	1.4 - 100	12	98

^aUnitless.^bBased on 95% confidence interval of assumed lognormal distribution.

Table 5. A summary of studies of the uncertainty in the internal dose conversion factor based on an analysis of the variability of the input parameters used in the calculational model

Radio- nuclide	Mode of exposure	Organ	Age group	Variance of logtrans- formed data ^a	Estimated range ^b (Sv/MBq)	Geometric mean (Sv/MBq)	Reference
¹³¹ I	Inges- tion	Thyroid	Newborn	0.45	1.4 - 21	5.3	101
			Children (0.5 - 2 Yrs.)	0.43	0.82 - 11	3.1	101
			Adolescents (6 - 16 Yrs.)	0.35	0.59 - 6.3	1.9	101
			Adults	0.35	0.10 - 1.0	0.3	101
¹²⁹ I	Inges- tion	Thyroid	Adults	0.66			Estimated from (101) in (114)
¹³⁷ Cs	Inges- tion	Total body	Children and adolescents (0.1 - 18 Yrs.)	0.18	0.004 - 0.019	0.009	115
			Adults	0.09	0.005 - 0.017	0.010	115
⁹⁰ Sr	Inges- tion	Bone surface	Adults	0.12	0.0002 - 0.0008	0.0004	98

^aUnitless.

^bBased on 95% confidence interval of assumed lognormal distribu-
tion.

The variance of the logtransformed data is an especially useful measure of model and parameter uncertainty because it provides a direct comparison of the variability among model parameters as well as a way of analyzing the variability in model output when the parameters are linear, multiplicative components of the model, as is the case with Eq.(1) (Refs. 2, 114). By comparing values of this parameter in Table 4 with the values of the variance of the logtransformed data given in Table 3 and Table 5, it can be seen that the usage factors and the dose conversion factors can contribute a large fraction to the total estimated uncertainty in the dose to man. Many of the dose estimates given in Table 4 are for releases to the atmosphere. By examining Table 1 it can be seen that for annual average dose estimates the uncertainty in air concentration, as expressed in the variance of the logtransformed data, is generally less than the uncertainties in the remaining portions of the dose calculation. As the averaging time gets shorter, however, the uncertainty in the air concentration under some circumstances may far exceed the remaining uncertainties in the dose calculation as estimated in Table 4.

7.2 Other Uncertainty Estimates

Order-of-magnitude estimates of the uncertainty associated with various aspects of the calculation of the dose to man from various long-lived radionuclides have been made. These estimates are summarized in Table 6. The values given in this table are based more on judgement than on rigorous mathematical analysis. It is interesting to note, however, that the uncertainty estimates given in Table 6 do not differ greatly from the estimates presented in previous tables in this report.

Table 6. Order-of-magnitude estimates of the uncertainty associated with various dose calculations based primarily on scientific judgement

Quantity	Radionuclide	Range (Orders of magnitude)	Reference
Estimated maximum dose equivalent rate to the most exposed individuals near various reference nuclear facilities	^3H , ^{14}C , ^{85}Kr , ^{131}I	1 - 2	116
Estimated global effects for a period of 10,000 years following a release to the atmosphere			
atmospheric specific activity	^{14}C	<1	117
dose rate conversion factor	^{14}C	<1	117
Estimated long-term population dose resulting from a known release to a freshwater surface-water system			
environmental concentration	Pu	2	5
individual intake	Pu	2	5
dose rate conversion factor	Pu	3	5

The uncertainty estimates given in Table 6 for ^3H and ^{14}C are based on a "specific activity" approach for calculating the dose from these radionuclides. The methods that have been recommended for use for hypothetical breeder reactor releases of ^3H and ^{14}C are also based on a specific activity approach.¹¹⁸ This approach assumes that the radionuclide and stable isotopes of the same element behave essentially identical in the environment. If complete equilibrium between a receptor in a critical group and its environment are assumed, the specific activity approach should result in an upper limit estimate of the initial dose to that receptor.^{98,116}

8. SUMMARY AND CONCLUSIONS

Estimates based on currently available data of the uncertainty associated with models and parameters used to assess the impact on man of radioactivity releases to the environment by breeder reactor facilities have been summarized. The models examined are similar to those recommended for use in breeder reactor radiological assessments.¹ They include those developed for the prediction of atmospheric and hydrologic transport, terrestrial and aquatic food-chain bioaccumulation, and internal and external dosimetry. Both long-term and short-term release conditions are discussed.

Validation studies for the straight-line Gaussian plume atmospheric dispersion model under various release conditions have been examined. Annual average air concentrations over flat terrain can be predicted within a factor of 2 to 4. However, it was also found that the accuracy of this model decreases as the averaging time decreases and/or the complexity of the meteorological and terrain conditions increase. More data are needed

to perform a statistical analysis of the uncertainty associated with the model although sufficient data for this purpose are unlikely to be available. Such data will also allow a clearer determination of whether more complex models should supplement the common Gaussian plume model in radiological assessments under some circumstances.

The uncertainty associated with three aquatic transport models was examined, two of which include the effects of radionuclide sorption onto sediments. These models have been recommended for use when simple stream dilution models are judged to be inappropriate. Much less validation data were reviewed for these aquatic models than for atmospheric dispersion models. None of the models compared with this limited data base overpredicted by more than a factor of 2 or underpredicted by more than a factor of 4. This level of ability to predict aquatic transport of materials is probably adequate for most routine assessments since doses from aquatic sources are generally small compared to other dose pathways.

Only limited validation data were found for predictions of radionuclide concentrations in terrestrial and aquatic foods. One study indicated a tendency for a given set of generic default parameter values to overpredict the concentration of ^{131}I in milk resulting from an air concentration composed of a mixture of physical and chemical forms of ^{131}I . It was also found that most of the default values of the fish bioaccumulation factor chosen for use in radiological assessment models in the absence of site-specific information lead to overpredictions of radionuclide concentrations.

Dose conversion factors are used to relate a radionuclide exposure to an absorbed radiation dose in humans. Validation of these factors has not been done, however, because it is not practical to directly measure

absorbed dose in the tissues of a living person. Internal dose conversion factors have been developed using physical phantoms of the body, as have some external dose conversion factors. The uncertainty in the latter quantity is expected to be less than the uncertainty in the internal dose conversion factor because little variation in doses from external sources is expected due to individual anatomical and physiological differences.

A limited number of statistical analyses have been performed for the rates of intake by adult, of various food products. The values of these and other usage factors are highly age dependent, and the variability of a given factor within a particular age category may be quite large. Additional variability may be associated with the location from which food is obtained with respect to individual residences. These variabilities may have a significant impact on the overall uncertainty of the final dose calculation.

Not enough information exists to specify the overall uncertainty in the calculation of the dose to man on the basis of validation studies. As a result, alternative methods to determine the potential uncertainty associated with dose prediction models have been developed. In this study both estimates of uncertainty based on an analysis of the variability of input parameters and estimates of uncertainty based primarily on scientific judgement have been considered. The range in predicted doses per unit environmental concentration was estimated to be 1 to 2 orders of magnitude with the extreme values having a very low probability of occurrence. The uncertainty in the dose conversion factor and the uncertainty in the usage factor may be significant contributors to the overall uncertainty in dose. For annual average releases to the atmosphere from many sites the uncertainty in the air concentration is small relative to the overall

uncertainty in dose. As the averaging time for the release decreases, however, the uncertainty in the air concentration may become increasingly significant.

If the uncertainties in the dose calculation are to be reduced, the priority of which uncertainty source to examine first should be addressed. Hoffman¹¹⁹ has suggested the following ranked uncertainties for investigation from highest to lowest priority:

1. Source term.
2. Food chain bioaccumulation and transport.
3. Human usage.
4. Physical dispersion.
5. Dose conversion factors.

Both the relative magnitude of each term's uncertainty and the practicability of reducing this uncertainty were considered when compiling this list. For example, the dose conversion factor was given the lowest priority on this list because it is perhaps the most difficult to improve due to natural human variability and the impracticability of measuring absorbed dose in humans.

Many of the uncertainty estimates presented in this report represent only estimates of the variability of a model's output and bias due to selection of input parameters. More validation studies need to be performed for these assessment models wherever possible to determine their accuracy. Also, the uncertainty in the amount of material released to the environment must be considered when estimating the overall accuracy of

predictions of the dose to man as a result of radionuclide releases by breeder reactor facilities. However, despite these limitations, it is hoped that the uncertainty estimates presented in this report can provide guidance to decision makers as they examine the expected uncertainty associated with specific dose assessment calculations such as those found in environmental impact statements and safety analysis reports.

The International Atomic Energy Agency (IAEA) is in the process of preparing a report containing recommended generic models and parameter values to be used to predict the dose to the most exposed individuals (critical groups) as a result of routine radionuclide releases from nuclear facilities.¹²⁰ The models presented in this IAEA report are similar to those recommended for use for breeder reactor radiological assessments.¹ Not all of the parameter values in the IAEA report, however, are the same as those recommended for breeder reactor assessments. The IAEA report suggests that when doses calculated using their generic models and parameter values are within an order of magnitude of a relevant dose limit that the calculations should be carefully evaluated to determine if the models and parameter values being used are appropriate for the specific site being assessed. Provided that the parameter values used are representative of the population groups being assessed, the uncertainty estimates presented in this report indicate that this is a reasonable guideline for most sites when generic models and parameter values are used to calculate doses from annual average radionuclide releases. For accidental releases, especially those from reactors located in sites dominated by complex terrain and/or coastal meteorology, the uncertainty in the dose calculations may be much

larger than an order of magnitude. As a result, it may be necessary to use a significant amount of site-specific information to reduce the amount of uncertainty in the dose calculations under these circumstances.

REFERENCES

1. Miller, C. W., C. F. Baes III, D. E. Dunning, Jr., E. L. Etnier, K. A. Kirkscey, D. C. Kocher, C. A. Little, L. M. McDowell-Boyer, H. R. Meyer, E. M. Rupp, and R. W. Shor. 1980. Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
2. 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. NUREG/CR-1004. ORNL/NUREG/TM-282.
3. Meyer, H. R., J. E. Till, E. S. Bomar, W. D. Bond, L. E. Morse, V. J. Tennery, and M. G. Yalcintas. 1979. Radiological impact of thorium mining and milling. Nucl. Safety 20(3):319-330.
4. 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 Part 50, Appendix I. Revision 1, Regulatory Guide 1.109.
5. Kocher, D. C., R. W. Leggett, D. E. Dunning, Jr., M. T. Ryan, and K. F. Eckerman. 1982. Uncertainties in the assessment of long-term collective dose and health effects. pp. 279-290. In Proceedings of the Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste, ed. D. C. Kocher. NUREG/CP-0022. CONF-810372.
6. U. S. Nuclear Regulatory Commission. 1979. Atmospheric Dispersion Models for Potential Accident Consequence Assessments at Nuclear Power Plants. Regulatory Guide 1.145.
7. Kelly, G. N., J. A. Jones, and B. W. Hunt. 1977. An Estimate of the Radiological Consequences of Notional Accidental Releases of Radioactivity from a Fast Breeder Reactor. National Radiological Protection Board Report NRPB-R53.

8. Bonnenberg and Drescher. 1981. Proceeding of the Workshop on Evaluation and Mitigation of Accidental Releases of Radioactivity - Identification of Uncertainties. Aldenhoven, Federal Republic of Germany (in press).
9. Hoffman, F. O., C. W. Miller, D. L. Shaeffer, C. T. Garten, Jr., R. W. Shor, and J. T. Ensminger. 1977. A Compilation of Documented Computer Codes Applicable to Environmental Assessment of Radioactivity Releases. ORNL/TM-5830.
10. Miller, C. W. 1980. Atmospheric dispersion and deposition. pp. 7-22. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
11. Gifford, F. A., Jr. 1968. An outline of theories of Diffusion in the lower layers of the atmosphere. Chap. 3 In Meteorology and Atomic Energy - 1968, ed. D. Slade. USAEC Report TID-24190.
12. Hanna, S. R., G. A. Briggs, and R. P. Hosker, Jr. 1982. Handbook on Atmospheric Diffusion. USDOE Report DOE/TIC-11223.
13. U.S. Nuclear Regulatory Commission. 1977. Methods for Estimating Atmospheric Transport and Dispersion of Gaseous Effluents in Routine Releases from Light-Water-Cooled Reactors. Revision 1, Regulatory Guide 1.111.
14. Rohwer, P. S., F. O. Hoffman, and C. W. Miller. 1981. Intermediate range atmospheric transport and technology assessments: nuclear pollutants. pp. 25-36. In Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment, ed. C. W. Miller, S. J. Cotter, and S. R. Hanna. CONF-801064.
15. Crawford, T. V. (Chairperson). 1978. Atmospheric transport of radionuclides. pp. 5-32. In Proceedings of a Workshop on the Evaluation of Models Used for the Environmental Assessment of Radionuclide Releases, ed. F. O. Hoffman, D. L. Shaeffer, C. W. Miller, and C. T. Garten, Jr. CONF-770901.

16. Faberick, A., R. Sklarew, and J. Wilson. 1977. Point Source Modeling. Form and Substance, Inc., Westlake Village, California.
17. Hilst, G. R. 1978. Plume Model Validation. Report EA-917-SY, Electric Power Research Institute, Palo Alto, California.
18. Koch, R. C., W. G. Biggs, P. H. Hwang, I. Leichter, K. E. Pickering, E. R. Sawdey, and J. L. Swift. 1977. Power Plant Stack Plumes in Complex Terrain: An Appraisal of Current Research. EPA 600/7-77-020.
19. Pasquill, F. 1974. Atmospheric Diffusion, 2nd Edition. John Wiley and Sons, New York.
20. Weber, A. H. 1976. Atmospheric Dispersion Parameters in Gaussian Plume Modeling Part I. Review of Current Systems and Possible Future Developments. EPA-600/4-76-030a.
21. Gogolak, C. V. 1975. A Data Set for Noble Gas Plume Exposure Model Validation. HASL-296.
22. Gogolak, C. V., H. L. Beck, and M. M. Pendergast. 1981. Calculated and observed ⁸⁵Kr concentrations within 10 km of the Savannah River Plant chemical separation facilities. Atmos. Environ. 15:497-507.
23. Ruff, R. E., H. S. Javitz, and J. S. Irwin. 1980. Development and application of a statistical methodology to evaluate the realtime air quality model (RAM). pp. 663-669. In Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.
24. Buckner, M. R. 1981. Proceedings of the First SRL Model Validation Workshop (November 19-20, 1980 at Hilton Head, South Carolina). DP-1597.
25. Draxler, R. R. 1980. An improved Gaussian model for long-term average air concentration estimates. Atmos. Environ. 14:597-601.

26. Fields, D. E., C. W. Miller, and S. J. Cotter. 1981a. The AIRDOS-EPA computer code and its application to intermediate range transport of ⁸⁵Kr from the Savannah River Plant. pp. 49-60. In Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment, ed. C. W. Miller, S. J. Cotter, and S. R. Hanna. CONF-801064.

27. Huang, J. C. 1980. Evaluation of a modified Gaussian plume model for travel distances 25-150 km. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

28. Pendergast, M. M. 1979. Model evaluation for travel distances 30-140 km. In Proceedings, Fourth Symposium on Turbulence, Diffusion, and Air Pollution. American Meteorological Society, Boston.

29. Telegadas, F., G. J. Ferber, J. L. Heffter, and R. R. Draxler. 1978. Calculated and observed seasonal and annual Krypton-85 concentrations at 30-150 km from a point source. Atmos. Environ. 12:1769-1775.

30. Fields, D. E., C. W. Miller, and S. J. Cotter. 1981. Comparison of observed and predicted sector-averaged air concentrations for elevated releases of fluorescein particles. J. Air Poll. Control Assoc. 31(8):871-876.

31. Gogolak, C. V. 1973. Comparison of Measured and Calculated Radiation Exposure from a Boiling Water Reactor Plume. HASL-277.

32. Guzewich, D. C., and W. J. B. Pringle. 1977. Validation of the EPA-PTMTP short-term Gaussian dispersion model. J. Air Poll. Control Assoc. 27(6):540-542.

33. Miller, C. W., and S. J. Cotter. 1982. Comparison of observed and predicted normalized air concentrations for particles released from a height of 111 meters. pp. 271-274. In Preprint Volume, Third Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

34. Partridge, J. E., J. A. Broadway, C. R. Phillips, S. T. Windham, and C. B. Nelson. 1976. Air Pathway Exposure Model Validation Study at the Monticello Nuclear Generating Plant. EPA-520/5-76-015.

35. Trout, D. A. 1980. Comparisons of predictions with observations and analyses of observations of high short-term pollutant concentrations monitored in the vicinity of tall stacks. pp. 650-657. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

36. Howroyd, G. C., and P. D. Gutfreund. 1980. An evaluation of the Briggs downwash formulation. pp. 185-192. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

37. Koziar, P., B. Adamski, and B. Anderson. 1980. A sensitivity analysis of three downwash models under neutral atmospheric stability. pp. 121-128. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

38. Miller, C. W. 1981. Comparison of Gaussian plume model air concentration predictions using Huber's building wake effects adjustment with Rancho Seco observations. pp. 248-249. In Extended Abstracts, Fifth Symposium on Turbulence, Diffusion, and Air Pollution. American Meteorological Society, Boston.

39. Deluca, P. M., Jr., J. A. Bauhs, and D. W. Pearson. 1981. Calculated and measured HTO atmospheric dispersion rates within meters of a release site. Health Phys. 41(1):191-195.

40. Van der Hoven, I. 1981. A Comparison of Measured Versus Model-Predicted Effluent Diffusion for Ground Releases. NOAA Technical Memorandum ERL ARL-105.

41. Miller, C. W., and C. L. Little. 1980. Accuracy of Gaussian plume dispersion model predictions as a function of three atmospheric stability classification calculations. Health Phys. 39(5): 773-782.

42. Miller, C. W., and R. E. Moore. 1977. Verification of a methodology for computing ground-level air concentration of SO_2 and suspended particulates for both point and dispersed sources. pp. 321-326. In Preprint Volume, Joint Conference on Applications on Air Pollution Meteorology. American Meteorological Society, Boston.
43. Smith, D. B., and R. B. Ruch, Jr. 1979. Comparative performance in complex terrain of several air quality impact assessment models based on aerometric program data. pp. 225-228. In Preprints, Fourth Symposium on Turbulence, Diffusion, and Air Pollution. American Meteorological Society, Boston.
44. Wilson, D. A., H. E. Cramer, J. F. Bowers, Jr., and H. V. Geary, Jr. 1977. Detailed diffusion modeling as a method of interpreting and supplementing air quality data. pp. 315-320. In Preprints, Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.
45. Bendel, W. B., and G. A. Cresswell. 1977. A comparison of predicted EPA VALLEY model concentrations to measured ground level tracer gas concentrations obtained in complex terrain. pp. 176-181. In Preprints, Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.
46. Egan, B. A., R. D'Errico, and C. Vando. 1979. Estimating air quality levels in regions of high terrain under stable atmospheric conditions. pp. 177-181. In Preprints, Fourth Symposium on Turbulence, Diffusion and Air Pollution. American Meteorological Society, Boston.
47. Elliott, R. D., E. L. Hovind, and M. W. Edelstein. 1977. Observations and model calculations of plume impaction on rising terrain. pp. 182-187. In Preprints, Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.
48. Iwanchuk, R. M., G. F. Hoffnagle, C. J. Vando, D. M. Shea, and J. T. Ferrill. 1980. Model comparisons for the High Terrain Monitoring Program. pp. 600-607. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

49. Lague, J. S., E. M. Irvine, and T. F. Lavery. 1980. Validation of alternate turbulence typing models for estimating air quality effects of coastal sources near rough terrain. pp. 586-592. In Conference Papers, Second Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

50. Lavery, T. F., and L. L. Schulman. 1977. The validity of a Gaussian plume point source diffusion model for predicting short-term SO_2 levels in the vicinity of electric generating plants in New York State. pp. 291-298. Preprints, Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

51. Septoff, M., L. J. Brunton, and L. H. Teuscher. 1977. Results of an offshore dispersion program conducted at the San Onofre nuclear generating station. pp. 5-8. In Preprints, Joint Conference on Applications of Air Pollution Meteorology. American Meteorological Society, Boston.

52. Netter, J., W. Wasserman, and G. A. Whitmore. 1978. Applied Statistics. Allyn and Bacon, Inc., Boston.

53. Vogt, K. J., H. Geiss, J. Straka, and F. Rohloff. 1981. A new trajectory model for intermediate range transport under changing weather conditions. pp. 177-186. In Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment, ed. C. W. Miller, S. J. Cotter, and S. R. Hanna. CONF-801064.

54. Begovich, C. L., B. D. Murphy, and C. J. Nappo, Jr. 1978. RETADD: A Regional Trajectory and Diffusion-Deposition Model. ORNL/TM-5859.

55. Heffter, J. L., and A. D. Taylor. 1975. A Regional - Continental Scale Transport, Diffusion, and Deposition Model, Part I: Trajectory Model. NOAA Technical Memorandum ERL-ARL-50.

56. Lange, R. 1978. ADPIC - A three-dimensional particle-in-cell model for the dispersal of atmospheric pollutants and its comparison to regional tracer studies. J. Appl. Meteorol. 17:320-329.

57. Murphy, B. D. 1981. PHENIX - A Pseudo-Spectral Model of Long-Range Atmospheric Transport. ORNL-5761.
58. Sklarew, R. C., and V. A. Mirabella. 1979. Experience in IMPACT modeling of complex terrain. pp. 205-212. In Preprint Volume, Fourth Symposium on Turbulence, Diffusion, and Air Pollution. American Meteorological Society, Boston.
59. Watson, C. W., and S. Barr. 1976. Monte Carlo Simulation of the Turbulent Transport of Airborne Contaminants. LA-6103.
60. Miller, C. W., S. J. Cotter, and S. R. Hanna (eds.). 1981. Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment. CONF-801064.
61. McMahon, T. A., and P. J. Denison. 1979. Empirical atmospheric deposition parameters - a survey. Atmos. Environ. 13:571-585.
62. Sehmel, G. A. 1980. Particle and gas dry deposition: a review. Atmos. Environ. 14:983-1011.
63. Hosker, R. P., Jr. 1981. Practical application of air pollution deposition models - current status, data requirements, and research needs. In Proceedings of the International Conference on Air Pollutants and Their Effects on the Terrestrial Ecosystem, eds. S. V. Krupa and A. H. Legge. John Wiley and Sons, New York (in press).
64. Johnson, R. P., K. Hardeastle, C. T. Nelson, B. Perez, C. Guderjohn, and C. R. Dickson. 1978. Test Report for the Characterization of Released Particle Tests Conducted at INEL During FY 1978. N707TR130025, Rockwell International.
65. Johnson, R. P., and C. T. Nelson. 1979. Interim Test Report for the Characterization of Released Particle Tests Conducted at INEL During FY 1979. N707TR130028, Rockwell International.

66. Little, C. A., C. W. Miller, and S. J. Cotter. 1982. Comparison of observed and predicted air concentrations for 18.6 μm fluorescein particles. Health Phys. (in press).
67. Piepho, M. G. 1981. Gaussian model predictions of elevated particle releases using tilted plume and deposition model features. pp. 343-349. In Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment, ed. C. W. Miller, S. J. Cotter, and S. R. Hanna. CONF-801064.
68. Hubschmann, W. (Chm.) and C. W. Miller (Sec.). 1980. Final report of the working group atmospheric transport. In Proceedings of a Workshop on Accuracy in Dose Calculations for Radionuclides Released to the Environment. GUV Gesellschaft fur Umweltuberwachung mbH, Aldenhoven, Federal Republic of Germany.
69. Little, C. A. 1980. A comparison of exposure pathways. pp. 103-115. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
70. Project Management Corporation. 1975. Clinch River Breeder Reactor Plant Environmental Report.
71. Gloyna, E. F. (Chairperson). 1978. Hydrologic transport of radionuclides. pp. 33-54. In Proceedings of a Workshop on the Evaluation of Models Used for the Environmental Assessment of Radionuclide Releases, ed. F. O. Hoffman, D. L. Shaeffer, C. W. Miller, and C. T. Garten, Jr. CONF-770901.
72. Little, C. A., and C. W. Miller. 1979. The Uncertainty Associated with Selected Environmental Transport Models. ORNL-5528.
73. Little, C. A. 1980. Surface water transport. pp. 55-61. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.

74. Yotsukura, N., and E. D. Cobb. 1972. Transverse diffusion of solutes in natural streams. U.S. Geol. Survey Prof. Paper 582-C, U.S. Government Printing Office.
75. Yotsukura, N., and W. W. Sayre. 1976. Transverse mixing in natural channels. Water Resour. Res. 12(4):695-704.
76. U.S. Nuclear Regulatory Commission. 1977. Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix I. Regulatory Guide 1.113.
77. Onishi, Y., and S. E. Wise. 1978. Mathematical Simulation of Transport of Sediment and Kepone in the James River Estuary. PNL-2731.
78. Onishi, Y. 1977. Finite Element Models for Sediment and Contaminant Transport in Surface Waters-Transport of Sediments and Radionuclides in the Clinch River. BNWL-2227.
79. Onishi, Y. 1977. Mathematical Simulation of Sediment and Radionuclide Transport in the Columbia River. BNWL-2228.
80. McDowell-Boyer, L. M., and C. F. Baes III. 1980. Terrestrial food chain transport. pp. 23-53. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
81. Bennett, B. G. 1981. The exposure commitment method in environmental pollutant assessment. Environ. Monitor. Assess. 1:21-36.
82. Kaye, S. V., F. O. Hoffman, L. M. McDowell-Boyer, and C. F. Baes III. 1981. Development and application of terrestrial food chain models to assess health risks to man from releases of pollutants to the environment. In Proceedings of the International Symposium on Health Impacts of Different Sources of Energy. IAEA-SM-254/63. International Atomic Energy Agency, Vienna (in press).

83. Hoffman, F. O. (Chm.), and H. D. Break (Sec.). 1980. Final report of the working group terrestrial food chains. In Proceedings of a Workshop on Accuracy in Dose Calculations for Radionuclides Released to the Environment. GUV Gesellschaft für Umweltüberwachung mbH, Aldenhoven, Federal Republic of Germany.

84. Hoffman, F. O. 1979. Prediction of the transport of ^{131}I through the grass-cow-milk pathway and the subsequent dose to an infant's thyroid. pp. 149-162. In A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides. ed. F. O. Hoffman and C. F. Baes III. NUREG/CR-1004. ORNL/NUREG/TM-282.

85. Shaeffer, D. L. 1981. A two-compartment model for transport of ^{131}I via the pasture-cow-milk pathway. Health Phys. 41(1):155-164

86. Weiss, B. H., P. G. Voilleque, J. H. Keller, B. Kahn, H. L. Krigger, A. Martin, and C. R. Phillips. 1975. Detailed Measurement of ^{131}I in Air, Vegetation and Milk Around Three Operating Reactor Sites. NUREG-75/021.

87. Shor, R. W. 1980. Aquatic food chain transport. pp. 63-70. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.

88. Feldt, W. (Chm.) and S. V. Kaye (Sec.). 1980. Final report of the working group aquatic food chains. In Proceedings of a Workshop on Accuracy in Dose Calculations for Radionuclides Released to the Environment. GUV Gesellschaft für Umweltüberwachung mbH, Aldenhoven, Federal Republic of Germany.

89. 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.

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

91. Kahn, B., R. L. Blanchard, W. L. Brink, H. L. Krieger, H. E. Kolde, W. J. Averett, S. Gold, A. Martin, and G. L. Gels. 1974. Radiological Surveillance Study at the Haddam Neck PWR Nuclear Power Station. EPA-520/3-74-007.
92. Lowman, F. G., T. R. Zice, and F. A. Richards. 1971. Accumulation and redistribution of radionuclides by marine organisms. In Radioactivity in the Marine Environment. National Academy of Sciences.
93. Thompson, S. E., C. A. Burton, D. J. Quinn, and Y. C. Ng. 1972. Concentration Factors of Chemical Elements in Edible Aquatic Organisms. UCRL-50554, Rev. 1.
94. Rupp, E. M. 1980. Adult dietary intake and inhalation rates. pp. 71-74. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
95. Rupp, E. M. 1979. Adult dietary intake and respiration rates, U. pp. 109-132. In A Statistical Analysis of Selected Parameters for^{sp} Predicting Food Chain Transport and Internal Dose of Radionuclides, ed. F. O. Hoffman and C. F. Baes III. NUREG/CR-1004. ORNL/NUREG/TM-282.
96. Rupp, E. M. 1980. Age dependent values of dietary intake for assessing human exposures to environmental pollutants. Health Phys. 39(2): 151-163.
97. 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.
98. Hoffman, F. O., R. H. Gardner, and K. F. Eckerman. 1982. Variability in Dose Estimates Associated With the Food Chain Transport and Ingestion of Selected Radionuclides. NUREG-CR-2612.

99. Eckerman, K. F. (Chm.), and H. D. Roedler (Sec.). 1980. Final report of the working group internal and external dosimetry. In Proceedings of a Workshop on Accuracy in Dose Calculations for Radionuclides Released to the Environment. GUV Gesellschaft fur Umweltuberwachung mbH, Alderhoven, Federal Republic of Germany.

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

101. Dunning, D. E., Jr., and G. Schwarz. 1981. Variability of human thyroid characteristics and estimates of dose from ingested ¹³¹I. Health Phys. 40(5):661-675.

102. Meyer, H. R., D. E. Dunning, Jr., D. C. Kocher, and K. K. Kanak. 1980. Dose conversion factors. pp. 75-94. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.

103. Lahti, G. P., R. S. Hubner, and J. C. Golden. 1981. Assessment of gamma-ray exposures due to finite plumes. Health Phys. 41(2): 319-340.

104. Kocher, D. C. 1981. Dose-Rate Conversion Factors for External Exposure to Photons and Electrons. NUREG/CR-1918. ORNL/NUREG-79.

105. Chen, W., J. W. Poston, and G. G. Warner. 1978. An Evaluation of the Distribution of the Absorbed Dose in Child Phantoms Exposed to Diagnostic Medical X-rays. ORNL/TM-5933.

106. International Commission on Radiological Protection. 1978. Radionuclide Release into the Environment: Assessment of Doses to Man. ICRP Publication 29. Pergamon Press, New York.

107. Matthies, M., K. Eisfeld, H. Paretzke, and 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(5):764-769.

108. O'Neill, R. V., R. H. Gardner, F. O. Hoffman, and G. Schwarz. 1981. Parameter uncertainty and estimated radiological dose to man from atmospheric ¹³¹I releases: a Monte Carlo approach. Health Phys. 40(5):760-764.

109. Schubert, Y., and A. Bradsky. 1967. The lognormal function as a stochastic model of the distribution of strontium-90 and other fission products in humans. Health Phys. 13:1187.

110. 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. Intl. 4:289-297.

111. Shaeffer, D. L., and F. O. Hoffman. 1979. Uncertainties in radiological assessments - a statistical analysis of radiiodine transport via the pasture-cow-milk pathway. Nuclear Technol. 45:99-106.

112. Stocum, W. E. 1970. Variability in the parameters used to predict dose to the thyroid from the ingestion of ¹³¹I in milk. In Environmental Surveillance in the Vicinity of Nuclear Facilities, Proceedings of a Symposium sponsored by the Health Physics Society.

113. Garten, C. T., Jr. 1980. Statistical uncertainties in predicting plutonium dose to lung and bone from contaminated soils. Health Phys. 39(1):99-103.

114. 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 Proceedings of the International Symposium on Migration in the Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle. IAEA-SM-257/63. International Atomic Energy Agency, Vienna (in press).

115. Schwarz, G., and D. E. Dunning. 1981. Imprecision in estimates of dose from ingested ^{137}Cs due to variability in human biological characteristics. Health Physics (submitted).
116. Nuclear Energy Agency, Group of Experts. 1980. Radiological Significance and Management of Tritium, Carbon-14, Krypton-85, Iodine-129 Arising from the Nuclear Fuel Cycle. Organization for Economic Co-operation and Development, Paris.
117. 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. NUREG/CR-1303. ORNL/NUREG/TM-378.
118. Binier, E. L. 1980. Doses due to atmospheric releases of ^3H and ^{14}C . pp. 95-102. In C. W. Miller et al., Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.
119. Hoffman, F. O. 1982. Personal communication.
120. International Atomic Energy Agency. 1982. Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides in Predicting Exposures to Critical Groups from Routine Releases. Vienna (in press).

Appendix A
The Model Evaluation Project

The U.S. Department of Energy (DOE) Liquid Metal Fast Breeder Reactor (LMFBR) safety effort is divided into four Lines of Assurance (LOAs) structured as four protective barriers, all of which must be breached before the public is endangered by an LMFBR malfunction.

The four Lines of Assurance are:

LOA-1: Prevent Accidents (by system reliability),

LOA-2: Limit Core Damage (by fault accommodation),

LOA-3: Maintain Containment Integrity (by energy
and debris accommodation), and

LOA-4: Attenuate Radiological Consequences.

There is also a coordinating area called LOA Support and Integration.

Experimental and analytical work is underway in each of these areas by various DOE contractors. This work is sponsored by the Safety and Physics Division of the Office of Reactor Research and Technology and is managed by the Fast Reactor Safety Technology Management Center located at the Argonne National Laboratory.^{A1} One project underway at Oak Ridge National Laboratory as part of the LMFBR safety program is entitled Model Evaluation of Breeder Reactor Radioactivity Releases. The purpose of this model evaluation project is to develop and validate the technology base of analytical tools (models) specifically required to achieve LOA-4 goals for capabilities to predict individual and population exposures and health effects resulting from postulated breeder reactor radionuclide releases to the environment.

As noted by Hoffman in one of the first reports prepared by the staff of this model evaluation project,^{A2}

(t)he determination of the impact of hypothetical radioactivity

releases from fast breeder reactors requires the use of mathematical models. These models are employed as tools to predict the environmental transport and subsequent dose associated with specific radionuclides that may be released during routine or accidental situations involving the operation of these facilities. Because these models can only be approximations of real events, a degree of uncertainty is associated with their use. Generally, this uncertainty has been assumed to result in overestimates of actual environmental concentrations or doses. Such conservative models are useful when their predictions lead to estimates of environmental concentrations or doses that are no cause for concern. If, however, these conservative models predict concentrations or doses which approach or exceed established standards, it is possible that expensive measures would be initiated to alleviate a problem generated solely by the model. On the other hand, attempts to reduce the conservatism employed by a model to improve its predictive accuracy must also focus on improving predictive precision. A model aimed at predicting only the average or most probable concentration or dose resulting from a given release may run a high risk of underestimating actual concentrations or doses unless the uncertainty associated with its predictive capability is reduced or, at least, taken into account. There is a need, therefore, to evaluate the models to determine the degree of uncertainty associated with their predictions.

During this project accomplishments have been made in five major work areas:

1. The models to be evaluated have been compiled.^{A3}
2. The models' conceptual framework, simplifying assumptions, and data bases are being evaluated.^{A4-A7}
3. Uncertainty estimates have been made based on scientific judgments, input parameter analyses, and validation studies.^{A4, A8-A10}
4. Recommendations concerning models and parameter values are being made.^{A11, A12}
5. Recommendations for research are being made.^{A12, A13}

The progress made in each of these activities has been documented in a number of Laboratory reports, open literature publications, and

conference proceedings, in addition to those publications cited above.^{A14}

Work on this project is continuing with current efforts being focused in three major areas:

1. Comparison of predictions from various environmental transport models with new measured field data as it becomes available to estimate the uncertainty in model output;
2. Continued updating of recommendations concerning models and parameters best suited to breeder reactor environmental radiological assessments;^{A11} and
3. Identification of further model evaluation, environmental research, and biomedical research needs.^{A13},

As this work continues, modifications of the information contained in this report will be made as required.

REFERENCES FOR APPENDIX A

- A1. Wantland, J. L. 1981. Personal communication.

- A2. Hoffman, F. O. 1978. Introduction. pp. 1-3. In C. W. Miller (comp.), The Evaluation of Models Used for the Assessment of Radionuclide Releases to the Environment, Progress Report for the Period April 1976 through December 1977. ORNL-5382.

- A3. Hoffman, F. O., C. W. Miller, D. L. Shaeffer, C. T. Garten, Jr., R. W. Shor, and J. T. Zasminger. 1977. A Compilation of Documented Computer Codes Applicable to Environmental Assessment of Radioactivity Releases. ORNL/TN-5830.

- A4. Hoffman, F. O., D. L. Shaeffer, C. W. Miller, and C. T. Garten, Jr. (Eds.). 1978. Proceedings of a Workshop on the Evaluation of Models Used for the Environmental Assessment of Radionuclide Releases. CONF-770901.

- A5. Miller, C. W. (compiler). 1978. The Evaluation of Models Used for the Assessment of Radionuclide Releases to the Environment, Progress Report for the Period April 1976 through December 1977. ORNL-5382.

- A6. Till, J. E., F. O. Hoffman, and D. E. Dunning, Jr. 1978. Assessment of ⁹⁹Tc Releases to the Atmosphere - A Plan for Applied Research. ORNL/TN-6260.

- A7. Miller, C. W., F. O. Hoffman, and D. E. Dunning, Jr. 1981. Comparison of the COMRADEX-IV and AIRDOS-EPA Methodologies for Estimating the Radiation Dose to Man from Radionuclide Releases to the Atmosphere. ORNL/TN-6495.

- A8. Miller, C. W., D. E. Dunning, Jr., E. L. Etnier, F. O. Hoffman, C. A. Little, H. R. Meyer, D. L. Shaeffer, and J. E. Till. 1979. Validation of Selected Predictive Models and Parameters for the Environmental Transport and Dosimetry of Radionuclides. OENL/TM-6663.
- A9. Little, C. A., and C. W. Miller. 1979. The Uncertainty Associated with Selected Environmental Transport Models. OENL-5528.
- A10. Cotter, S. J., C. W. Miller, R. E. Moore, and C. A. Little. 1980. Estimates of Dose Due to Noble Gas Releases from the Three Mile Island Incident Using the AIRDOS-EPA Computer Code. OENL-5649.
- A11. Miller, C. W., C. F. Baes, III, D. E. Dunning, Jr., E. L. Etnier, K. A. Kirkscey, D. C. Kocher, C. A. Little, L. M. McDowell-Boyer, H. R. Meyer, E. M. Rupp, and R. W. Shor. 1980. Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. OENL-5529.
- A12. Miller, C. W., S. J. Cotter, and S. R. Hanna (eds.). 1981. Proceedings, Symposium on Intermediate Range Atmospheric Transport Processes and Technology Assessment. CONF-801064.
- A13. Miller, C. W., D. E. Dunning, Jr., E. L. Etnier, D. C. Kocher, L. M. McDowell-Boyer, H. R. Meyer, and P. S. Rohwer. 1980. Recommendations Concerning Research and Model Evaluation Needs to Support Breeder Reactor Environmental Radiological Assessments. OENL/TM-7491.
- A14. Kanak, K. K., and C. W. Miller. 1980. The Evaluation of Models Used for the Assessment of Radionuclide Releases to the Environment, A Summary of Documentation for the Period April 1976 through June 1979. OENL-5573.

Appendix B
Some Statistical Properties of the Lognormal Distribution

The statistical distribution associated with many environmental quantities will approximate the lognormal distribution. This appendix, adapted from one in Dunning and Schwarz,^{B1} briefly describes some of the fundamental relationships of lognormal distributions.

Consider a positive variate X such that $\ln X$ is normally distributed with mean μ and variance σ^2 . Variate X has the positively skewed lognormal distribution $L(\mu, \sigma)$. Many of the statistical properties of the frequency density distribution $L(\mu, \sigma)$ may be expressed by the functional relationship between μ and σ (Ref.B2).

The value of several measures of position within the distribution, i.e., the mode (X_p), median (X_m), mean ($\bar{X}$), the 2.25 percentile ($X_{2.25}$), and the 97.75 percentile ($X_{97.75}$) are given by

$$X_p = e^{\mu - \sigma^2}$$

$$X_m = e^{\mu}$$

$$\bar{X} = e^{\mu + \frac{\sigma^2}{2}}$$

$$X_{2.25} = e^{\mu - 2\sigma}$$

$$X_{97.75} = e^{\mu + 2\sigma}$$

In this study, the probable range in X was assumed to be equal to the difference between $X_{97.75}$ and $X_{2.25}$ or approximately 95% of the possible outcomes. The standard deviation or the variance, s or s^2 respectively, of the frequency distribution of X may be found from

$$s^2 = e^{2\mu + \sigma^2} (e^{\sigma^2} - 1)$$

Given a set of data of unknown distribution, several tests may be performed to examine the assumption of lognormality. An informal test is to plot the raw data on logarithmic probability paper. If the data can be reasonably fitted with a straight line, a lognormal distribution may be assumed. More rigorous tests, especially when trying to choose between an assumption of lognormality or normality, are given by the Kolmogorov-Smirnov one-sample test or others. ^{B3, B4}

The parameters μ and σ^2 may be estimated from N independent and individual observations of X_i ($i = 1$ to N) by

$$\mu = \frac{1}{N} \sum_{i=1}^N X_i$$

$$\sigma^2 = \frac{1}{N-1} \sum_{i=1}^N (\ln X_i - \mu)^2$$

For a lognormal distribution, the parameters μ and σ may be estimated from the arithmetic mean ($\bar{X}$) and the standard deviation (s) of the raw data by

$$\mu = \ln \frac{\bar{X}}{\left[1 + \left(\frac{s}{\bar{X}} \right)^2 \right]^{1/2}}$$

$$\sigma^2 = \ln \left[1 + \left(\frac{s}{\bar{X}} \right)^2 \right]$$

The last two equations listed above were used to derive values of μ and σ^2 , respectively, from some of the cited references as needed for the construction of Table 4.

If only the range in variate X is known, it may be reasonable to assume that X is associated with a continuous uniform or loguniform rectangular probability distribution. If a and b represent the minimum and maximum values of the range in X , respectively, the mean μ and the variance, σ^2 , of the loguniform distribution is estimated by^{B5}

$$\mu = \frac{(\ln b + \ln a)}{2}$$

$$\sigma^2 = \frac{(\ln b - \ln a)^2}{12}$$

The latter equation was used to calculate the variance of the logtransformed data presented in Table 1.

REFERENCES FOR APPENDIX B

- B1. Dunning, D. E., Jr., and G. Schwarz. 1981. Variability of human thyroid characteristics and estimates of dose from ingested ¹³¹I. Health Phys. 40(5): 661-675.
- B2. Aitchison, J. and J. A. C. Brown. 1969. The Lognormal Distribution. Cambridge University Press, New York.
- B3. Fields, D. E. 1981. TERPED: A Versatile Code for Examining the Distribution of Experimental Data. ORNL-5689.
- B4. Siegel, S. 1956. Nonparametric Statistics for the Behavioral Sciences. McGraw-Hill, New York.
- B5. Netter, J., W. Wasserman, and G. A. Whitmore. 1978. Applied Statistics. Allyn and Bacon, Inc., Boston.

As noted in Section 7, in the absence of extensive validation data an estimate of the potential uncertainty associated with model predictions can be obtained through an analysis of the variability of the input parameters used in the model. It must be emphasized again, however, that any such analysis may provide an indication of the variability of model output but not of its accuracy. Only model validation experiments can provide information on the accuracy of a model's output.

Searches of the literature have been made for reported values of the input parameters used in various food chain transport models. Typically, a single value for each of these parameters is used in radiological assessment models such as those recommended for breeder reactor assessments.^{C1} It has been found, however, that there is generally a spread of values reported in the literature for each of these parameters. The resulting probability distributions have been estimated, and the statistical parameters necessary to describe these distributions have been calculated.

The results of a number of analyses of selected input parameters used in various food chain transport models are summarized in Table C.1 (Ref. C2). The parameters listed in Table C.1 are assumed to be associated with lognormal probability distributions. The original parameter values used to derive Table C.1 were obtained from the literature. As a result, the data have not been obtained at random from an appropriate experimental design, and the amount of information available to interpret the quality of reported values for the various parameters varies from parameter to parameter and from document to document for any given parameter.^{C2} Furthermore, the quantity of data available is not uniformly distributed among the various parameters listed in Table C.1.

Table C.1. Variability indices determined for selected input parameters used in environmental radiological assessment models (adopted from Miller and Hoffman⁽²⁾)

Parameter	Range	Variance of logtrans- formed data	Reference
Environmental half-time on growing vegetation, days			
all values for herbaceous vegetation	2.8 - 34	0.30	C2
I ₂ vapor on herbaceous vegetation	4.5 - 14	0.11	C2
I particulates on herbaceous vegetation	2.8 - 16	0.31	C2
other particulates on herbaceous vegetation	9.0 - 34	0.14	C2
Dry deposition velocity, cm/s			
particulates	0.001 - 60	4.2	Estimated in (C2) from values tabulated in (C4)
reactive gases	0.015 - 8.0	1.4	
Interception fraction for forage grasses	0.02 - 0.82	1.5	C3
Vegetation density^a			
forage, kg (dry wt.)/m ²	0.04 - 1.6	0.35	C5
leafy vegetables, kg (fresh wt.)/m ²	0.36 - 5.3	0.21	C5

Table C.1 (continued)

Parameter	Range	Variance of logtrans- formed data	Reference
Total feed consumption rate for dairy cows,^a kg (dry wt.) per cow per day			
	6 - 25	0.014	C5
Milk transfer factor for dairy cows. days/liter			
iodine	0.0027 - 0.035	0.30	C5
strontium	0.00064 - 0.0045	0.28	C5
cesium	0.0025 - 0.016	0.34	C5
Beef transfer factor, days/kg			
cesium	0.0047 - 0.097	0.66	C5

^aThese parameters are readily improved on a site-specific basis.

Therefore, direct comparison of the results presented in Table C.1 is limited because of inequalities in the quantity and quality of data for each parameter.^{C2}

The variability represented in Table C.1 for the various model input parameters should be larger than one might expect for a given parameter at a specific site because values of the individual parameters were obtained from a variety of locations. Therefore, the variabilities reported in Table C.1 should be more indicative of the overall variability associated with the generic application of default values of these parameters.^{C2} The variability in model predictions as a result of the variability in the model input parameters can be reduced if the correlations between various input parameters can be quantified.^{C3} Such determination is not a simple task, however, because simultaneous measurement of the various correlated parameters is required at various intervals in time.^{C2}

REFERENCES FOR APPENDIX C

- C1. Miller, C. W., C. F. Baes, III, D. E. Dunning, Jr., E. L. Etnier, K. A. Kirkscey, D. C. Kocher, C. A. Little, L. M. McDowell-Boyer, H. R. Meyer, E. M. Rupp, and R. W. Shor. 1980. Recommendations Concerning Models and Parameters Best Suited to Breeder Reactor Environmental Radiological Assessments. ORNL-5529.

- C2. 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 Proceedings of the International Symposium on Migration in the Environment of Long-Lived Radionuclides from the Nuclear Fuel Cycle. IAEA-SM-257/63. International Atomic Energy Agency, Vienna (in press).

- C3. 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.

- C4. Sehmel, G. A. 1980. Particle and gas dry deposition: a review. Atmos. Environ. 14:983-1011.

- C5. 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. NUREG/CR-1004. ORNL/NUREG/TM-282.