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UNCERTAINTIES IN ENVIRONMENTAL RADIOLOGICAL ASSESSMENT MODELS AND THEIR IMPLICATIONS

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ABSTRACT

Environmental radiological assessments rely heavily on the use of mathematical models. The predictions of these models are inherently uncertain because these models are inexact representations of real systems. The major sources of this uncertainty are related to biases in model formulation and parameter estimation. The best approach for estimating the actual extent of over- or underprediction is model validation, a procedure that requires testing over the range of the intended realm of model application. Other approaches discussed are the use of screening procedures, sensitivity and stochastic analyses, and model comparison. The magnitude of uncertainty in model predictions is a function of the questions asked of the model and the specific radionuclides and exposure pathways of dominant importance. Estimates are made of the relative magnitude of uncertainty for situations requiring predictions of individual and collective risks for both chronic and acute releases of radionuclides. It is concluded that models developed as research tools should be distinguished from models developed for assessment applications. Furthermore, increased model complexity does not necessarily guarantee increased accuracy. To improve the "realism" of assessment modeling, stochastic procedures are recommended that translate uncertain parameter estimates into a distribution of predicted values. These procedures also permit the importance of model parameters to be ranked according to their relative contribution to the overall predicted uncertainty. Although confidence in model predictions can be improved through site-specific parameter estimation and increased model validation, risk factors and internal dosimetry models will probably remain important contributors to the amount of uncertainty that is irreducible. The authors contend that, at low doses and low dose rates of the same magnitude as current standards for routine exposures by members of the general public, the implications of model uncertainties may be greater for legal, political, and economic issues than for the protection of human health.

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INTRODUCTION

From other papers presented at this meeting, it is apparent that the application of mathematical models is an essential element of environmental radiological assessments. These models are used to predict the physical dispersion of radionuclides in the atmosphere and hydrosphere; deposition onto soils, sediments and other surfaces; transfer of material into terrestrial and aquatic food chains; and dose-equivalents and health risks to humans resulting from external and internal exposure pathways. The quantities predicted by these models are used to evaluate compliance with regulatory limits, optimize the design of engineered systems for the retention of radioactive wastes, guide decisions regarding the siting of facilities, and document potential environmental impacts resulting from planned and unplanned releases.

In the past, little emphasis was placed on the evaluation of uncertainties in radiological assessments because conservative approaches produced calculated doses which were usually only small fractions of applicable dose limits [1-3]. During the last decade, however, regulatory guides [4] and standards [5] have been issued that are more restrictive than the primary radiation protection standards originally specified by the Federal Radiation Council [6]. To decrease the chance that conservative models erroneously predict values that infringe upon or exceed these regulatory limits, emphasis is being placed on increasing the "realism" of model predictions. For example, the actual locations of exposed population groups and average estimates of agricultural production and food consumption for specific sites are being used rather

than values which tend to maximize calculated exposures (e.g., Ref. [7]). However, the removal of obvious sources of conservatism from the models will increase the potential for underestimating actual exposures unless the uncertainties remaining in other areas of the model are taken into account. Thus, there is now increased incentive for evaluating model uncertainties over that which has existed in the past.

In this paper, we attempt to review the sources, methods for analysis, and relative magnitudes of uncertainties in radiological assessment models. We also discuss what we believe are the main implications of these uncertainties. Our conclusions are based to a large extent on our personal judgment and points of view developed from more than seven years of active investigations on model strengths and weaknesses [3,8-18], including participation in Task Groups 2 and 3 of NCRP Scientific Committee No. 64 [19]. Nevertheless, definitive statements on model uncertainties are difficult to make in the absence of extensive testing (model validation) conducted over a range of conditions. Points of view different from our own are therefore expected and welcomed.

It should also be noted that it is not our purpose to review the important role that models have played in the radiation protection field for over 30 years. Nor do we intend to imply through this discussion of uncertainties that modeling efforts should cease or be curbed. Model development will always be the basic process by which empirical information is synthesized into an understanding of natural systems. Our attempt here is merely to help decision-makers make better use of these important scientific tools.

SOURCES OF UNCERTAINTY

We have found it convenient to place the sources of model uncertainty into four major categories: (1) incorrect model formulation, (2) incorrect parameter estimation, (3) failure to account for parameter variability, and (4) errors in programming, computation, calculation, and report writing. In this discussion, we consider only those sources of uncertainty related to the first three categories.

Incorrect Model Formulation

Incorrect model formulation is the primary underlying source of uncertainty, since all models at their very best are only inexact representations of real systems. The complex nature of environmental transport, food chain bioaccumulation, and human dosimetry is such that no model can be expected to exactly describe the behavior of radioactivity in the environment and its effects on exposed individuals and populations. Thus, models developed for one set of conditions may not perform well when extrapolations are made to new situations.

Intentional or inadvertent departures of model formulations (including their accompanying framework of assumptions) from real world conditions will result in predictive errors that systematically tend to over- or underpredict actual events. This tendency we refer to as "model bias." Intentional sources of model bias are usually related to the application of conservative assumptions. Unintentional sources of model bias may be related to unwarranted assumptions of steady-state conditions or first-order kinetics, unrecognized relationships between model parameters, inadvertent exclusion of significant exposure pathways

or important physical processes, and accidental incorporation of the same process into the model more than once due to inconsistencies within formulations of the model structure and the derivation of parameter values.

Incorrect Parameter Estimation

Bias in model predictions may result from improper estimation of model parameters (i.e., the independent variables of the model), in addition to improper model formulation. Proper parameter estimation is difficult because appropriate data are seldom available. Estimates must often be made on the basis of published data derived from experiments designed for other purposes, and dependencies of the data on site-specific or other factors are not commonly determined or reported [2,7,20-22].

Radiological assessment models generally employ a data base of "default values" recommended for use in the absence of site-specific or population-specific data (e.g., Refs. [7,23-28]). The degree to which these recommended "default values" are applicable to specific situations cannot be stated, although the values are frequently assumed to be conservative [2,3]. However, this assumption may not always be valid [7,9,11,13,19]. Parameter estimation, in our opinion, is highly dependent on the use of subjective judgment (i.e., educated guesses). The accuracy of model predictions is therefore a function of the quality of this judgment. Sometimes it is difficult to appreciate the influence of judgment on the accuracy of model predictions because of the elaborate nature of the calculations performed and the abundant quantitative information generated by computer implementations. Nonetheless, the importance of judgment in model development and application must not be underrated.

Parameter Variability

Parameter variability as a source of uncertainty is related to the general use of deterministic models in radiological assessments [10,19]. By a deterministic model, we mean any model whose output depends on the selection of a single value for each input parameter (Fig. 1). For a given radionuclide and exposure pathway, deterministic models produce a single estimate of dose. In so doing, the effects of imprecise parameter estimation and system variability are ignored.

Because of the inherent variability associated with measurements taken of natural systems, we believe that model parameters used in assessment models should be described by a range, or still better, a distribution of values. Ultimately, parameter distributions should be translated into a distribution of model predictions from which some judgment of model uncertainties can be made. By contrast, deterministic models, produce only a single predicted quantity from single value parameter estimates and thus tend to present a misleading impression of accuracy. This is especially evident when order-of-magnitude parameter estimates are used to predict values given to several significant digits without providing any statement about the predictive uncertainty of the models.

Fig. 1. The derivation of a predicted value produced by a deterministic model.

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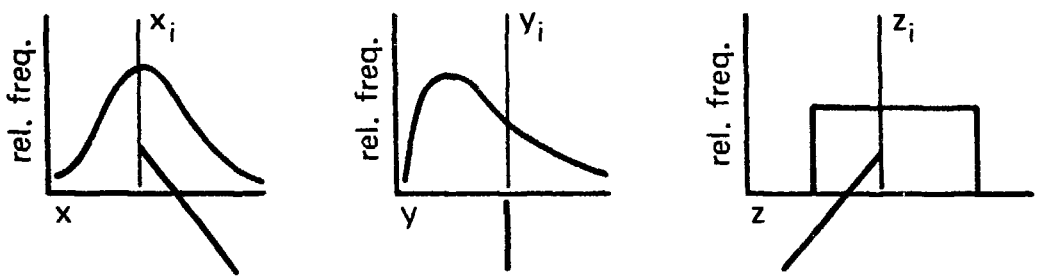
DETERMINISTIC ASSESSMENT MODELS

- ALTHOUGH A RANGE OF VALUES ARE POSSIBLE FOR EACH INPUT PARAMETER IN A MODEL,



- PARAMETERS ARE REPRESENTED BY A SINGLE QUANTITY

MODEL

$R_i = f(x_i, y_i, z_i)$

- THUS, A SINGLE VALUE IS PRODUCED AS MODEL OUTPUT

R_i = 0.3756 rem/yr

METHODS FOR EVALUATING UNCERTAINTY

Model Validation

The best procedure for estimating uncertainties in models is to test their predictions against observations made over the range of conditions for which the model is intended to be used. This procedure is commonly referred to as "model validation" [8-19]. The types of models that have been subjected most extensively to validation are those used to predict atmospheric dispersion [16,19,29,30]. Table 1 is a summary of the results obtained from a variety of tests performed on one basic type of dispersion model, i.e., the Gaussian plume model. Few attempts have been made to test the accuracy of an entire complement of radiological assessment models used to predict all aspects from a unit release to the environment out to human health risks. Perhaps the best examples involve tests of the accuracy of predicted concentrations of I-131 in milk following a chronic release to the atmosphere [31-34]. These tests have combined estimates of the chemical composition of the released I-131, atmospheric dispersion, wet and dry deposition, and agricultural food-chain transfer.

Most validation studies have usually been restricted to testing components of assessment models over relatively short time periods and at a few locations [3,16,19,29,30]. Testing over the full range of conditions for which model predictions are needed has not been conducted. Such extensive testing requires a substantial investment in time and financial resources. For this reason, model validation has not received a high research priority. In many cases, model validation may be nearly

Table 1. Estimates based on field validation studies of the ratio of predicted to observed air concentration associated with predictions by the Gaussian plume atmospheric dispersion model under various release conditions [16]

Conditions	Range
Annual average for a specific point, flat terrain	
0-10 km downwind	0.5 - 2
10-150 km downwind	0.25 - 4
Short-term, elevated releases	
Without building wake effects	0.1 - 10
With building wake effects	0.01 - 100
Short-term, surface-level releases	
Without building wake effects	0.3 - 100
With building wake effects	0.7 - 100
Complex terrain or meteorology (e.g., sea breeze regimes)	
Annual average concentrations	0.1 - 10
Short-term releases	0.01 - 100

impossible either due to extremely low levels of radionuclide concentrations in the environment or because the time periods considered by the model are prohibitively long [30].

Alternatives to Model Validation

Because of the expense and impracticality of model validation for many types of assessment models, other approaches must be considered for addressing model uncertainties. These approaches include the use of screening procedures to identify potentially important radionuclides and exposure pathways, sensitivity analyses to identify important groups of model parameters, stochastic analyses to determine the effect of parameter uncertainty on model predictions, and comparisons among the predictions of different models [10]. Each of these approaches requires extensive use of judgment and critical insight.

Screening procedures. Screening procedures are based on model predictions that are not likely to underestimate radionuclide concentrations in environmental media, dose, or health risk. Screening models are usually designed in such a manner that, as long as they predict values that do not infringe upon some established level of concern, there should be no further need to pursue an analysis of uncertainty even though the uncertainties may extend over several orders of magnitude. Screening calculations are very useful in that they focus attention on those few radionuclides and exposure pathways warranting further attention.

Few radiological assessment models currently in use have been officially designated for screening although screening techniques are widely used by modelers to define various aspects of a problem worth

detailed attention. The International Atomic Energy Agency has recently published a screening model for the evaluation of exposures to critical population groups from routine releases [25], and NCRP Scientific Committee 64 has organized a special task group to develop a screening approach for the United States. However, in the absence of formal analyses of model uncertainties, the amount of conservatism necessary to compensate for potential underprediction in screening calculations is largely dependent on subjective judgment [35].

Sensitivity analyses. Sensitivity analyses can be used to specify the relative effect of changes in the values of model parameters on the predicted quantity. If incorrect model formulation is not expected to be a major problem, then sensitivity analyses can be used as a measure of the potential importance of model parameters as contributors to uncertainties in model predictions [36,37]. In lieu of information on the range of uncertainty in parameter estimates, sensitivity analyses are usually performed by varying each parameter by a fixed amount and ascertaining the degree to which the predicted quantity is affected by these changes [10]. Such procedures are most effective when they are used as additional screening tools to eliminate from consideration model components whose influence on the model predictions can be neglected [10]. However, care must be taken in ranking the importance of model parameters and/or establishing research priorities based on sensitivities that have been determined only by perturbing model parameters by some fixed amount. This is because large differences in parameter uncertainties may overwhelm the effect of moderate differences in the functional relationships of model parameters within a deterministic model [10,25].

Stochastic analyses. In the absence of model validation, the next best approach for analyzing model uncertainties is the use of stochastic (i.e., probablistic) modeling procedures).* These procedures translate parameter uncertainties into uncertainties in model predictions by treating the uncertain parameters as random variables from which a distribution of values of model output is produced (Fig. 2). The effect of parameter variability on model output is treated explicitly. The distribution of values which is assumed for the uncertain model parameters depends on the quantity and quality of available data and investigator judgment, with the most commonly assumed distribution being the lognormal [9,18,19,38-41].

Stochastic approaches constitute the state-of-the-art in assessment modeling. They vary from simple statistical error propagation formulations to random sampling procedures requiring the use of a computer [8,9,11,18,36,38-46]. Stochastic procedures can be used to rank the importance of model parameters by accounting for both uncertainty and mathematical sensitivity [10] (e.g., Table 2).

The information included within the frequency distributions obtained from stochastic procedures can be impressive (Fig. 3). However, we caution against placing undue confidence in these distributions because of the judgmental nature with which input distributions are specified and the possible influence of unaccounted for sources of uncertainty in model formulation. From our experience, we have found

*Numerous terms have been used in the literature to describe methods of stochastic analyses. Among them are "parameter uncertainty analysis," "imprecision analysis," "statistical sensitivity analysis," "error analysis," "error propagation," and "probabilistic modeling" [8,10,11,17-19,25,30,36].

Fig. 2. A pictorial description of the basic procedure of stochastic analyses in which parameter uncertainty is translated into a distribution of model predictions.

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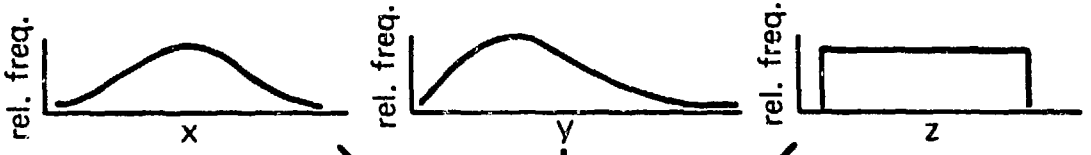
STOCHASTIC ANALYSIS

- UNCERTAIN PARAMETERS DESCRIBED AS RANDOM VARIABLES



- DISTRIBUTIONS INPUT TO PRODUCE A DISTRIBUTION OF PREDICTIONS

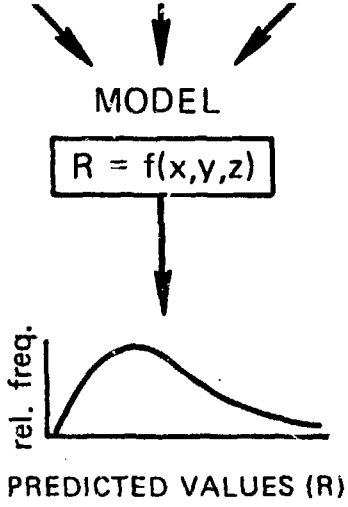


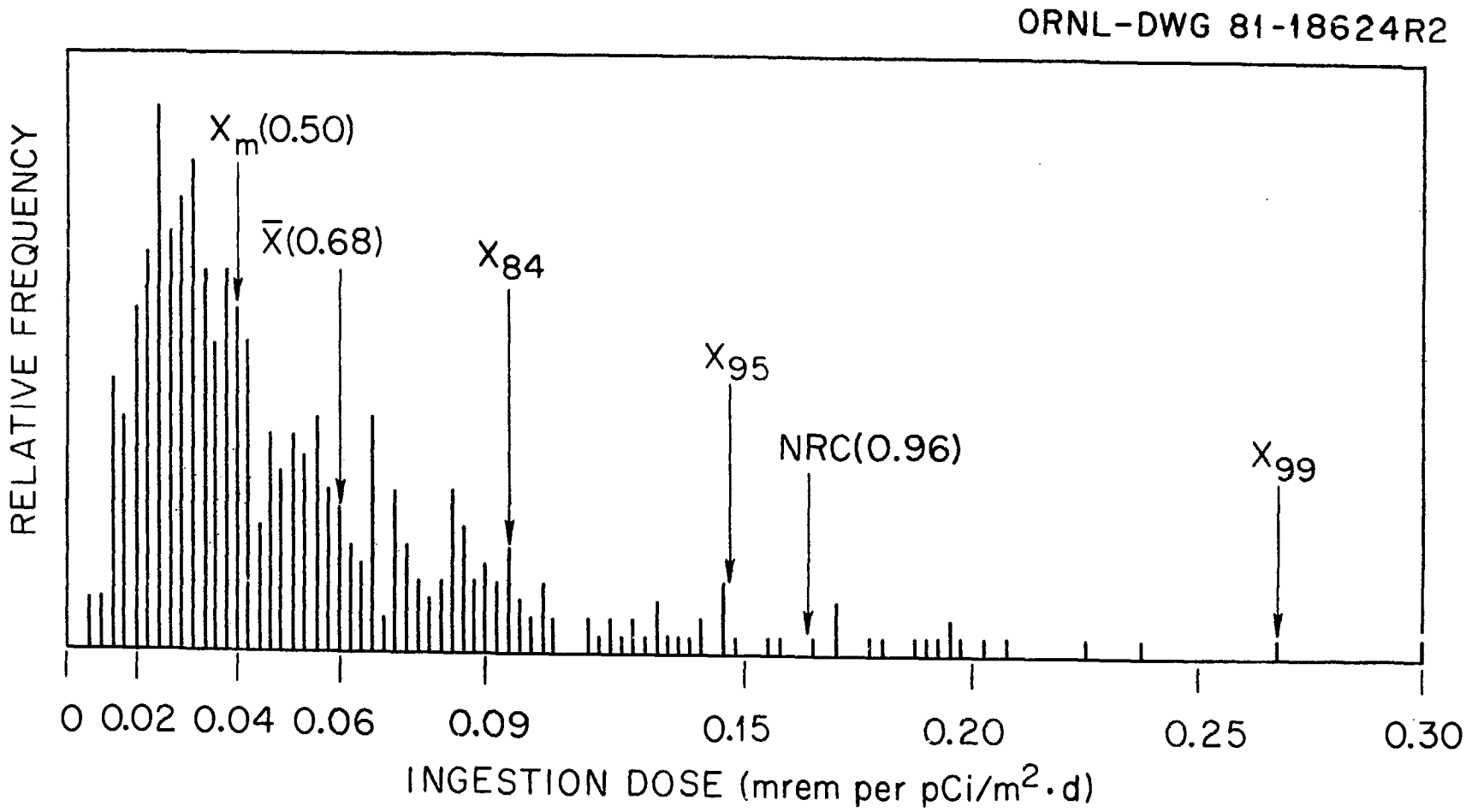
Table 2. An example of stochastic model rankings of parameter and pathway contributions to uncertainties in dose predictions for terrestrial and aquatic food chain transport of Cs-137 [11]

Pathway	Important pathways and parameters	Importance rank	Fractional contribution to total uncertainty
Aquatic pathways	Parameters		
Water-fish-man	Bioaccumulation factor, B_{fp}	1	0.6[a]
	Rate of fish consumption, U_f	2	0.42[a]
	Dose conversion factor, D_{ij}	3	0.05[a]
Combined terrestrial pathways	Pathways		
	Deposition-pasture-meat man	1	0.73[b]
	Deposition-pasture-milk-man	2	0.3[b]
	Deposition-nonleafy vegetables-man	3	0.05[b]
	Deposition-leafy vegetables-man	4	0.03[b]
	Parameters		
Deposition-all terrestrial pathways-man	Meat transfer coefficient, F_{if}	1	0.25[b]
	Mass interception factor, r/Y_v (pasture)	2	0.15[b]
	Dose conversion factor, D_{ij}	3	0.1[b]
	Loss rate from vegetation, λ_w (pasture)	4	0.09[b]
	Rate of meat consumption, U_f	4	0.09[b]
	Rate of milk consumption, U_m	5	0.0[b]
Single terrestrial pathways	Parameters		
Deposition-pasture-meat-man	Meat transfer coefficient, F_{if}	1	0.43[b]
	Rate of meat consumption, U_f	2	0.19[b]
	Mass interception factor, r/Y_v	3	0.1[b]
	Loss rate from vegetation, λ_w	4	0.07[b]
	Dose conversion factor, D_{ij}	5	0.0[b]

[a]Determined using analytical techniques by dividing the variance of log-transformed parameter distribution by the variance of the log-transformed distribution of dose prediction.

[b]Determined using numerical techniques by squaring the coefficient for the rank order correlation between parameter values and the predicted dose.

Fig. 3. A frequency distribution of stochastic predictions of the Cs-137 ingestion dose to the total body resulting from exposure to multiple terrestrial food pathways [11]. Predicted values are 50-year committed dose equivalents resulting from an annual exposure to a continuous rate of deposition onto an agricultural system for a duration of 15 years. The locations of the geometric mean (X_m), arithmetic mean ($\overline{X}$), and the 84th (X_{84}), 95th (X_{95}), and 99th (X_{99}) percentiles of the distribution are given as well as that of the deterministic prediction of NRC Regulatory Guide 1.109 [28]. The NRC value is located at the 96th percentile of this distribution, while X_m and $\overline{X}$ are located at the 50th and 68th percentiles, respectively.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RELATIVE FREQUENCY
X_m(0.50)
$\bar{X}$(0.68)
X_{84}
X_{95}
NRC(0.96)
X_{99}
0 0.02 0.04 0.06 0.09 0.15 0.20 0.25 0.30
INGESTION DOSE (mrem per pCi/m^2·d)

it more appropriate to derive from the output distribution a "probable range," based on a 95% or 99% confidence interval [10,16]. This range is interpreted in a qualitative sense as having a reasonably high probability of including the true value, assuming that biases in model formulation and parameter estimation are not unduly large. Examples of assessment model uncertainties determined using stochastic procedures are given in Table 3.

Model comparison. Another method that is suggested on occasion for estimating potential uncertainties in model predictions has been comparison of the results produced by different models, e.g., reference [47]. Unless one of the models in this comparison has been previously validated, this approach may be highly misleading. Extreme care should be taken to identify differences in model structure, underlying assumptions, and sources of data from which parameters have been estimated [10]. Although a large number of computer codes exist for radiological assessments [12], the mathematical forms of the models used by these codes are often very similar as are the sources from which their parameters have been estimated [3,7,9,12,19-28]. Thus, comparison among models may not reflect an evaluation of truly independent results. This effect is illustrated in Fig. 4, in which much less discrepancy is exhibited among the predictions of four deterministic assessment models than is indicated by the 95% range produced from stochastic modeling or from the range of observed values obtained from the long-term monitoring of Cs-137 in global fallout.

Table 3. A summary of results using stochastic modeling for a variety of environmental exposure pathways. Values presented are the 5th percentile (X_{05}), geometric mean (X_m), and upper 95th (X_{95}) percentile of predicted distributions

Radionuclide	Exposure pathway	X_{05}	X_m	X_{95}	Reference
Pu-239	Soil-air-inhalation-lung[a,b] (mrad/year per pCi/g soil)	2.5×10^{-3}	9.4×10^{-3}	3.5×10^{-2}	[11]
Pu-239	Soil-vegetation-ingestion-bone[a,b] (mrad/year per pCi/g soil)	4.1×10^{-4}	1.1×10^{-2}	3×10^{-1}	[11]
I-131	Air-pasture-milk-ingestion-thyroid[c,d] (rem/year per pCi/m^3 air)	0.66	3.9	23	[11]
Sr-90	Water-fish-ingestion-bone surface[b,e] (mrem/year per pCi/L water)	9.5×10^{-3}	0.25	6.5	[11]
Sr-90	Deposition-multiple terrestrial food pathways-bone surface[c,e] (mrem/year per pCi/m^2·d)	0.28	1.2	5.1	[11]
Cs-137	Water-fish-ingestion-total body[b,e] (mrem/year per pCi/L water)	9.3×10^{-2}	0.67	4.8	[11]
Cs-137	Deposition-multiple terrestrial food pathways-total body[c,e] (mrem/year per pCi/m^2·d)	1.2×10^{-2}	4.4×10^{-2}	0.15	[11]
Cs-137	Deposition-soil-pasture-milk[c,f] (pCi/L per Ci/km^2)	10	30	220	[11]

[a] Estimated dose rate at age 70 years from a lifetime exposure.

[b] Simple analytical procedures based on lognormal statistics used to propagate parameter error.

[c] Monte Carlo computer techniques used to propagate parameter error.

[d] Estimated thyroid dose for children of the age group 0.5 to 2.0 years.

[e] Estimated 50-year committed dose equivalent from a 1 year intake.

[f] Values approximated from published figures using lognormal statistics.

Fig. 4. A comparison of Cs-137 concentrations in milk (pCi/liter) predicted by four internationally recognized terrestrial food chain models (AIRDOS/EPA [26], NRC [28], NRPB [24,27], and IAEA [25]) with the second percentile (X_{02}), geometric mean (X_m), and 99th percentile (X_{97}) predicted using stochastic approaches [11], and with the range (UN_{low}, UN_{high}) derived from time integrations of long-term, global monitoring of Cs-137 in nuclear weapons fallout [48]. Milk concentrations are normalized for 30 years of continuous deposition at a rate of 1 pCi/m^2 d.

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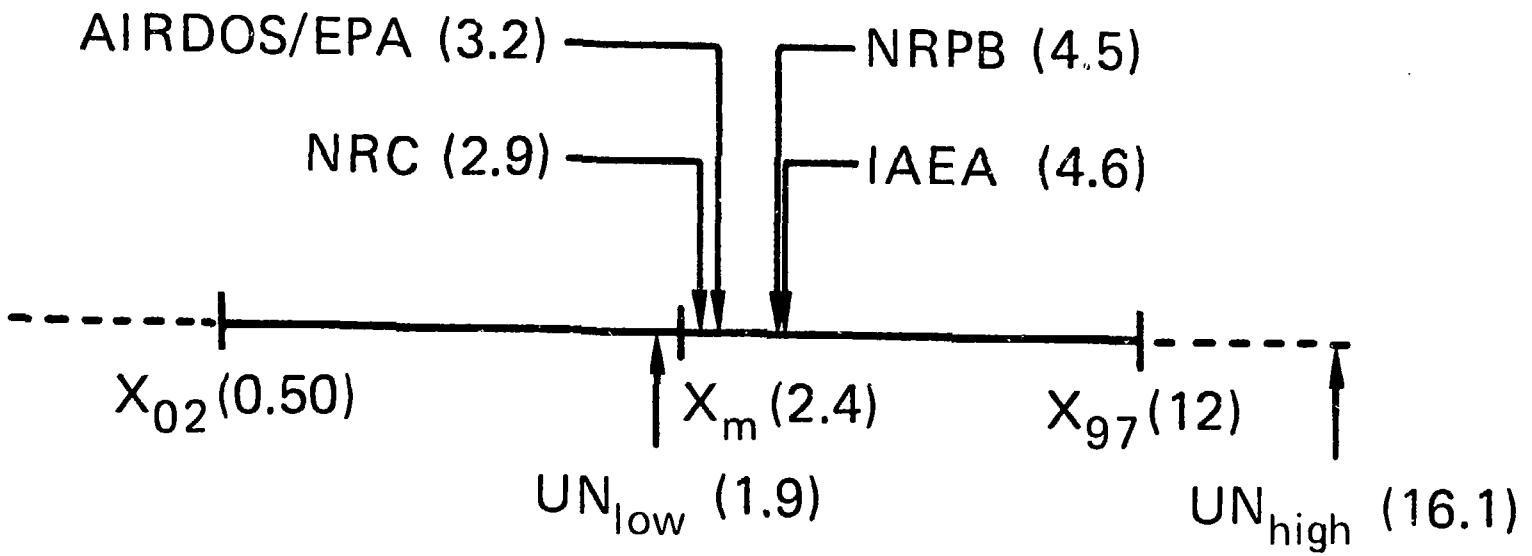

THE RELATIVE MAGNITUDE OF UNCERTAINTIES

As mentioned previously, a detailed uncertainty analysis has never been made for all exposure pathways and radionuclides nor is such an analysis likely to be made in the near future. It is possible, however, to form some generalizations on model uncertainties by combining information from limited model validation studies, stochastic modeling, and the careful use of expert judgment. From this information, it is apparent that the relative magnitude of uncertainties will be influenced by the nature of the questions addressed by the model and by the specific radionuclides and exposure pathways of dominant importance.

The Influence of Different Radionuclides and Exposure Pathways

Estimates of environmental transport and dosimetry of frequently studied radionuclides (e.g., H-3, Kr-85, Sr-90, I-131, and Cs-137) are expected to be more certain than those of other nuclides that have been less intensively researched (e.g., Tc-99, Nb-95, Np-237). Likewise, the prediction of dose for external modes of exposure is expected to be less uncertain than dose predictions from internal exposures which are more dependent on variable biological processes [16,30]. Predictions of exposures resulting from chronic emissions when the atmosphere and surface waters are predominant media of contamination are also expected to be less uncertain than predictions when groundwater, sediments, and surface soils become the predominant source of radionuclides in the environment [19,22,49].

For most nuclear fuel cycle facilities and most radionuclides, groundwater, sediments, and ground surface contamination will not be important sources of exposure until either the facilities have ceased operation or until after releases have persisted over long time periods (more than 30 to 100 years). These media are, however, most important for the evaluation of radiological exposures from waste repositories [19,49]. Usually only a few radionuclides and exposure pathways dominate the radiological importance of a release; however, when several exposure pathways contribute to the total exposure, uncertainties will tend to be less than when exposures are dominated by a single pathway due to the process of adding independent distributions [11]. An example of this effect is given in Table 4 whereby stochastic analyses show the uncertainties resulting from the exposure to Sr-90 and Cs-137 via multiple terrestrial food-chain pathways to be less than those from single pathways.

The Influence of Different Assessment Questions

The influence of different assessment questions on the magnitude of model uncertainties is illustrated in Tables 5 and 6. We have found that uncertainties will usually decrease with increased time- and space-averaging. Therefore, uncertainties in these tables tend to be less for questions related to routine (chronic) releases than for accidental (acute) releases. Uncertainties are also less for questions requiring assessments of collective or average individual risk than for questions about the risk to individuals, or more correctly, small population groups.

Table 4. Stochastic predictions of dose-equivalents for the deposition and subsequent transport of Sr-90 and Cs-137 through selected terrestrial food chain pathways [11]

Pathway	Geometric mean	Uncertainty factor[a]
Deposition-leafy vegetables-man (mrem per pCi/m^2·d)		
Sr-90 (bone surface)	0.27	9.0
Cs-137 (total body)	1.4×10^{-3}	5.8
Deposition-nonleafy vegetables-man (mrem per pCi/m^2·d)		
Sr-90 (bone surface)	0.34	14
Cs-137 (total body)	3.3×10^{-3}	11
Deposition-pasture-milk-man (mrem per pCi/m^2·d)		
Sr-90 (bone surface)	0.15	14
Cs-137 (total body)	8.2×10^{-3}	10
Deposition-pasture-meat-man (mrem per pCi/m^2·d)		
Sr-90 (bone surface)	0.055	18
Cs-137 (total body)	0.021	8.4
Exposure to all terrestrial pathways combined (mrem per pCi/m^2·d)		
Sr-90 (bone surface)	1.2	5.8
Cs-137 (total body)	0.044	4.8

[a]The uncertainty factor is the ratio of the maximum value of a 95% probability interval to the geometric mean.

Table 5. Estimates of uncertainties associated with calculations of the radiological impact of routine releases

Model component	Uncertainty factor[a]		References
	Individual[b]	Population[c]	
Release (source term)	10-100[d]	10-100[d]	[50,51]
Physical dispersion (atmosphere and surface water)	2-4	1.5-2	[3,16,19,29]
Food chain transport	3-50	1.5-10	[2,7,9,11,17, 19-22,30,35]
Usage factors (rates of intake)	1.3-7	1.2-2	[9,19,52-54]
Dose factors[e]	1.5-30	1.3-10	[16,38,41,55,56]
Risk factors[f]	2-20?	1.2-10?	[48,56-60]

[a]The ratio of a likely maximum or minimum value to the expected median value; estimated using extensive judgment in concert with information given in the cited references.

[b]Includes estimates for critical groups of the population.

[c]Includes estimates for average individuals.

[d]Uncertainties primarily due to conservative bias in engineering estimates of source terms.

[e]Includes errors due to extrapolations from animal data.

[f]Uncertainty estimates do not include bias dose to extrapolations to low doses; since there is the possibility that large overestimates of health risk may occur at doses which are only small fractions of natural background, a question mark is included after the maximum estimate of risk factor uncertainty.

Table 6. Estimates of uncertainties associated with calculations of the radiological impact of accidental releases

Model component	Uncertainty factor[a]		References
	Individual[b]	Population[c]	
Release (source term)	10-10^4?[d]	10-10^4?[d]	[55,61,62]
Physical dispersion[e] (atmosphere and surface water)	3-100	2-10	[3,16,55,63,66]
Food chain transport[f]	5-100	2-10	[18,55,64-66]
Usage factors (rates of intake)[g]	3-10	2-4	[19,52-54]
Dose factors[h]	2-30	1.5-10	[38,41,55,56]
Risk factors[i]	2-20	1.2-10	[48,55-60]

[a]The ratio of a likely maximum or minimum value to the expected median value; estimated using extensive judgment in concert with information given in the cited references.

[b]Includes estimates for critical groups of the population.

[c]Includes estimates for average individuals.

[d]Uncertainties primarily due to conservative bias in engineering estimates of source terms; question marks indicate a lack of consensus among consulted experts.

[e]Does not include uncertainties due to acute precipitation events.

[f]It is expected that environmental monitoring will serve to reduce this uncertainty in the event of an actual accidental radionuclide release.

[g]Following an actual release, we assume that emergency response measures will reduce the number of pathways through which the population may be exposed, with the possible exception of inhalation and initial external exposure to contaminated air and surfaces. Usage factors therefore refer to estimates of inhalation rates and the time-integrated exposure to radionuclides in air and on surfaces.

[h]Includes errors due to extrapolations from animal data.

[i]Excludes potential bias due to extrapolations to low doses; uncertainties are expected to decrease as exposure rates increase.

A major fraction of the total uncertainty in assessment calculations is related to the uncertainty in the estimate of the source term, although source-term estimates, because of the nature of engineering judgment, are usually biased on the side of conservatism. The effect of source-term uncertainties on the overall uncertainty is expected to be more pronounced for assessments of collective or average risk than for estimates of the risk to critical population groups or individuals. This is because uncertainties in source terms are unaffected by the increase in space-averaging that occurs for model predictions of environmental transport, bioaccumulation, dosimetry, and health risk when questions change emphasis from the radiological impact on individuals or small population groups to the collective impact on larger populations extending over large regions.

When in-plant monitoring effectively reduces source-term uncertainties, or when release limits are derived through back-calculation from established risk or dose limits, the uncertainties in environmental transport, dosimetry, and risk models take on increased importance. For assessments of routine emissions that may result in significant ingestion exposures, uncertainties in food-chain transport, internal dosimetry, and risk factors tend to be more important than uncertainties in dietary habits or atmospheric and surface-water transport. By contrast, uncertainties in atmospheric transport are very important for the assessment of accidental releases. The uncertainty estimates given in Table 6 for physical dispersion under accidental release conditions do not include the influences of acute precipitation events.

The estimates of uncertainties listed for the risk factors in Tables 5 and 6 reflect variability in data. No consideration has been given in these estimates to the well-known arguments concerning extrapolation of available data at high doses and dose rates to the region of low doses which are of primary concern in environmental radiological assessments (model bias). Because of extrapolation uncertainties, we expect greater potential for error in estimating risk factors for routine emissions which are associated with low-dose rates than for accidental releases during which higher doses would likely occur. However, the assumption of a linear relationship between dose and risk with no threshold at low doses is standard practice for radiological assessments, even though such an assumption may result in large overestimates of health risk at very low doses and dose rates [48,56-60].

Influence of Specific Model Assumptions

Assumptions leading to conservative bias. Although there are numerous assumptions employed in environmental radiological assessment models that directly affect the formulation of equations and the selection of parameter values, we focus our attention on a few specific examples commonly used in assessment models that are likely to cause predictive bias. Generally, both the source terms given by the engineering community and the health risks assumed at low doses can be considered to be conservative. The use of H-3 and C-14 specific activity models to assess the radiological importance of releases to the atmosphere will also be conservative for maximumly exposed individuals and for members of critical population groups if the assumption is made that all of the hydrogen and carbon in a human receptor is derived

from the atmosphere at the specified location [11,19]. More realistic assessments, which require estimates of the intakes of carbon and hydrogen from sources with lower specific activities of C-14 and H-3, may reduce dose estimates by an order of magnitude or more [11,19,25,30].

The assumption of geometric means as representative parameter estimates. Recently, there has been a tendency to assume that the geometric mean of a distribution of available data represents a "best" estimate of a parameter value for use in deterministic assessment models [7,19,21,22,32]. We caution against the indiscriminate use of geometric means for such purposes because they tend to underpredict collective or average exposures. Most data for model parameters are positively skewed [7,9,19-22,38], and thus the geometric mean is always less than the average or arithmetic mean (Fig. 3). If the predicted value from the model is directly proportional to the parameter estimate, then the extent of underpredicting the average will depend on the skewness of the distribution of data and the degree to which the data and model structure are biased toward underprediction.

However, at the very worst, underpredictions of the mean should not exceed one order of magnitude for routine release conditions. More likely, we expect the use of geometric means in deterministic calculations to result in underpredictions of the arithmetic mean that are less than a factor of three. Although, substantial underestimates of the arithmetic mean will occur when the uncertainties in the model output span several orders of magnitude, we expect that some degree of conservative bias will have been incorporated in the model to compensate for such potentially gross uncertainties.

Estimates of Overall Uncertainty

On the basis of our experience, we find that the largest uncertainties are associated with model predictions of the radiological consequences of long-term future emissions from geological repositories of high-level radioactive wastes [49,56] and with predictions of individual dose-rates resulting from acute, accidental releases from nuclear power facilities [55,65,66]. Quantities predicted for these situations are dependent on many unknown variables which lead to uncertainties that may span several orders of magnitude. Usually, conservative assumptions that are employed in these cases in order to compensate for the effect of unknown variables.

For the assessment of exposures to local population groups in the vicinity of nuclear fuel cycle facilities, we do not expect current deterministic assessment models (e.g., [25-28]) to underpredict by more than one order of magnitude actual exposures to routine releases, nor do we expect overpredictions to exceed two orders of magnitude, given reasonably accurate estimates of the annual rates of radionuclides released from these facilities. These latter estimates are not dissimilar to those made by others who have addressed the issue of model uncertainties [2,3,30]. The validity of these statements, however, will always be subject to contention in the absence of model validation.

CONCLUSIONS AND IMPLICATIONS

From our participation in various individual and group efforts to evaluate model strengths and weaknesses, we have developed some general conclusions concerning the implications of model uncertainties.

Assessment Models and Research Models Serve Different Purposes

First, we believe that there is a need to distinguish between research and assessment models. Assessment models are designed to serve as tools for decision-making, whereby research models are developed to improve understanding of the function and structure of real systems through explicit identification and simulation of operating processes. Within assessment models, underlying processes may be aggregated, for example, into general transfer coefficients that relate the concentration of a radionuclide in one compartment of a model to that in another compartment. Therefore, assessment models, as a rule, tend to be less complex in structure and more amenable to field testing than process-level research models [35].

Research models may be used in assessments to permit estimation of the importance of specific processes not explicitly accounted for in assessment models. Research models are also employed when it becomes necessary to quantify key parameters for which appropriate empirical relationships are nonexistent. It should be recognized, however, that the increased level of complexity represented by research models may not necessarily guarantee increased accuracy in model predictions [3,19,29,30,35]. Although increased complexity offers the advantage of increased flexibility in terms of the number of questions a model can address, it is our contention that the best model for assessment purposes is usually the simplest model that will produce results within an acceptable degree of accuracy [19,30].

An example of a simple empirical model which produces more accurate predictions than a more complex model is given in Fig. 5. In this example, the simple model is composed of a distribution of values derived

Fig. 5. A comparison of the stochastic predictions of a simple two-parameter model and a more complex ten-parameter model with observations of milk concentrations of I-131 averaged for a period of three months among two dairy cows at each of two distinct locations in the vicinity of the Quad Cities Nuclear Power Station. The observed average values are taken from data published by Voilleque et al. [32]. The distribution of values for the two-parameter model is derived from data published on steady-state or time-integrated air-to-milk relationships observed for I-131, I-127, and I-129 [9,67]. The results of the more complex model [17] include the assumption that only 50% of the airborne I-131 is in the elemental form.

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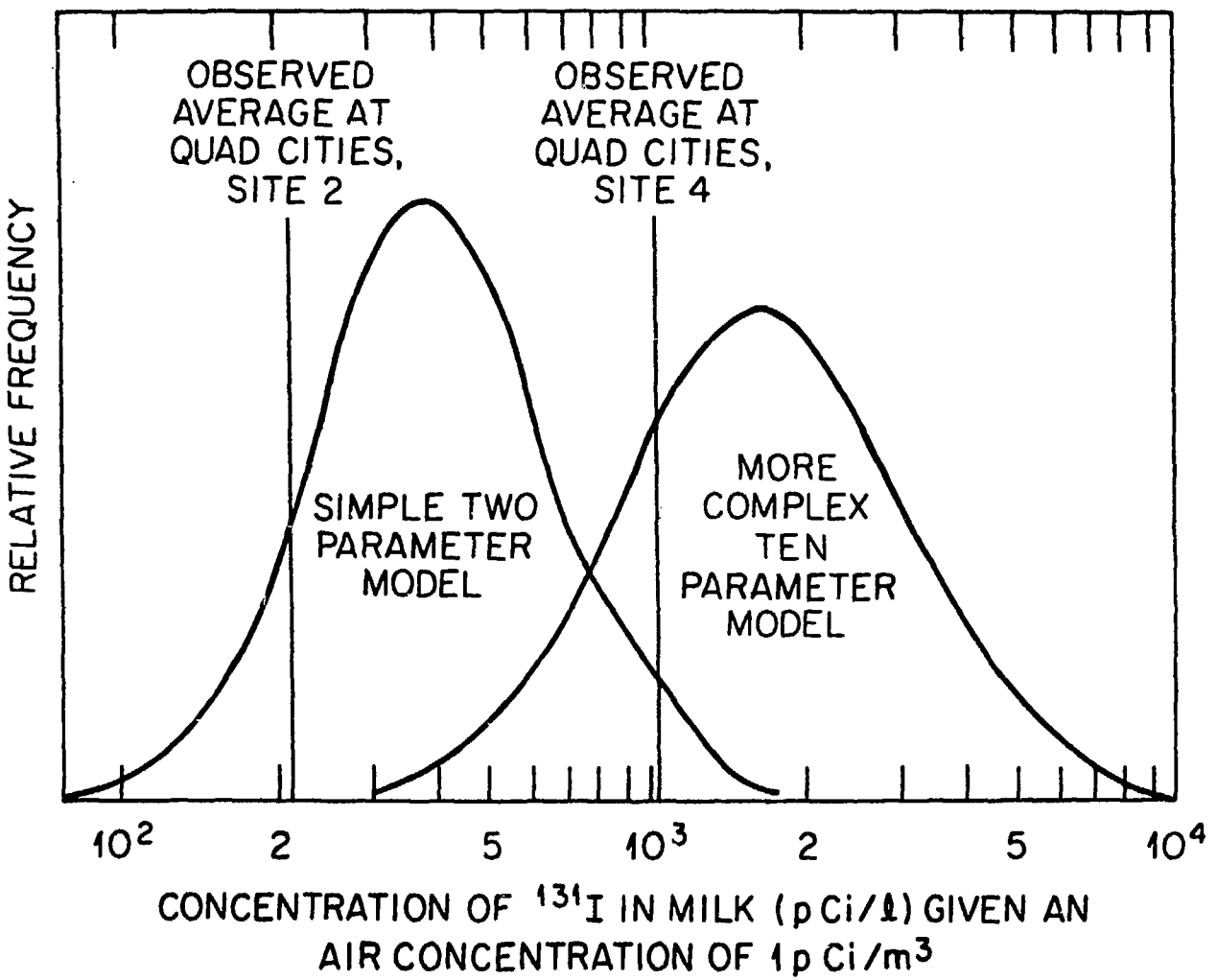

from reported steady-state or time-integrated relationships of iodine concentrations in air and in the milk of cows grazing on pasture. The more complex model is made up of 10 parameters that explicitly account for variability in the deposition onto ground; the interception, uptake, and retention of I-131 by vegetation; the variable grazing habits of dairy cows; dairy management practices; and the steady-state transfer of I-131 to milk from a daily rate of ingestion of forage. When compared with two time-averaged values of the air-to-milk transfer obtained from measurements at two different sites near the Quad Cities Nuclear Power Plant, the simple model produces better results. This outcome is probably because important interactions and processes implicitly included by the empirical data used in the simple model are excluded from the formulation of the more complex model, thus implying that the more complex model is not complex enough! Usually, the quality of available data will limit the amount of complexity in model structure that will contribute to increased predictive accuracy [10,19].

Deterministic Models Should Include Limits of Application

Our second conclusion is that deterministic models are only useful as tools for decision making when their predictions are accompanied by limits that clearly specify when the modeling results may not be applicable. Currently, there is only one deterministic assessment model that we know of that formally has this feature. This is the generic assessment model developed by the International Atomic Energy Agency for the evaluation of exposures to critical population groups from routine emissions [25]. This model requires further evaluation of the model structure and data base whenever predicted doses are within one

order of magnitude below a relevant dose limit. Specifying limits of application for deterministic models serves to protect against model misuse and to provide incentive for improvement of the assessment process by requiring renewed scruitiny of the model and its parameters whenever these limits of model application approach a level of concern to the decision-maker. Without any estimate of the limits of model application, decisions based on deterministic predictions may be in gross error. One method that has been suggested for overcoming such errors is to use the model (and its associated base of data) that was originally used for evaluating the feasibility of implementing regulations as the accepted approach for establishing compliance with these regulations, e.g. [68]. However, this procedure has the undesirable effect of reducing incentive to improve the accuracy of model predictions while allowing decisions to be based on calculated values having little or no relevance to reality.

Realistic Assessment Models Should Include Stochastic Procedures

We strongly recommend that any attempt to improve the "realism" of assessment modeling include the use of stochastic procedures to preserve the effect of parameter uncertainty within model predictions. These procedures are useful not only for initial decision-making but also for directing future research efforts through identification of the radionuclides, exposure pathways, and model parameters that contribute most to the overall predictive uncertainty. Nevertheless, extreme care must be given when applying stochastic procedures. Unrecognized correlations among parameters, improper model formulation, and indiscriminate specification of parameter variability can produce misleading

results. Despite their obvious utility for radiological assessments, stochastic modeling procedures are not a substitute for model validation.

Scientific Method Mandates Model Validation

The importance of model validation cannot be overemphasized. In fact, we contend that the scientific method mandates model validation through field testing of model predictions. In the absence of model validation, assessment models constitute untested hypotheses.

The importance of model validation is illustrated in Fig. 6. This figure compares stochastic and deterministic model predictions of the steady-state transfer of radioactive aerosols from air to vegetation with field observations [69]. Without validation data, comparison of the stochastic model with the deterministic model would indicate that deterministic predictions are probably conservative. When model predictions are compared with data obtained from air and vegetation samples collected in California, the initial indication provided by the stochastic model appears to be confirmed. However, both deterministic and stochastic models substantially underpredict observations made in Tennessee. Thus, validation over a range of conditions is needed to reveal model bias.

Bias in the models used in the above example originates from the assumption that dry deposition dominates wet deposition. The primary difference in the two sets of observed data is the effect of wet deposition, with wet deposition being almost negligible during the summer months in California. We also note that the stochastic predictions in Fig. 6 indicate more variability than is evident by either of

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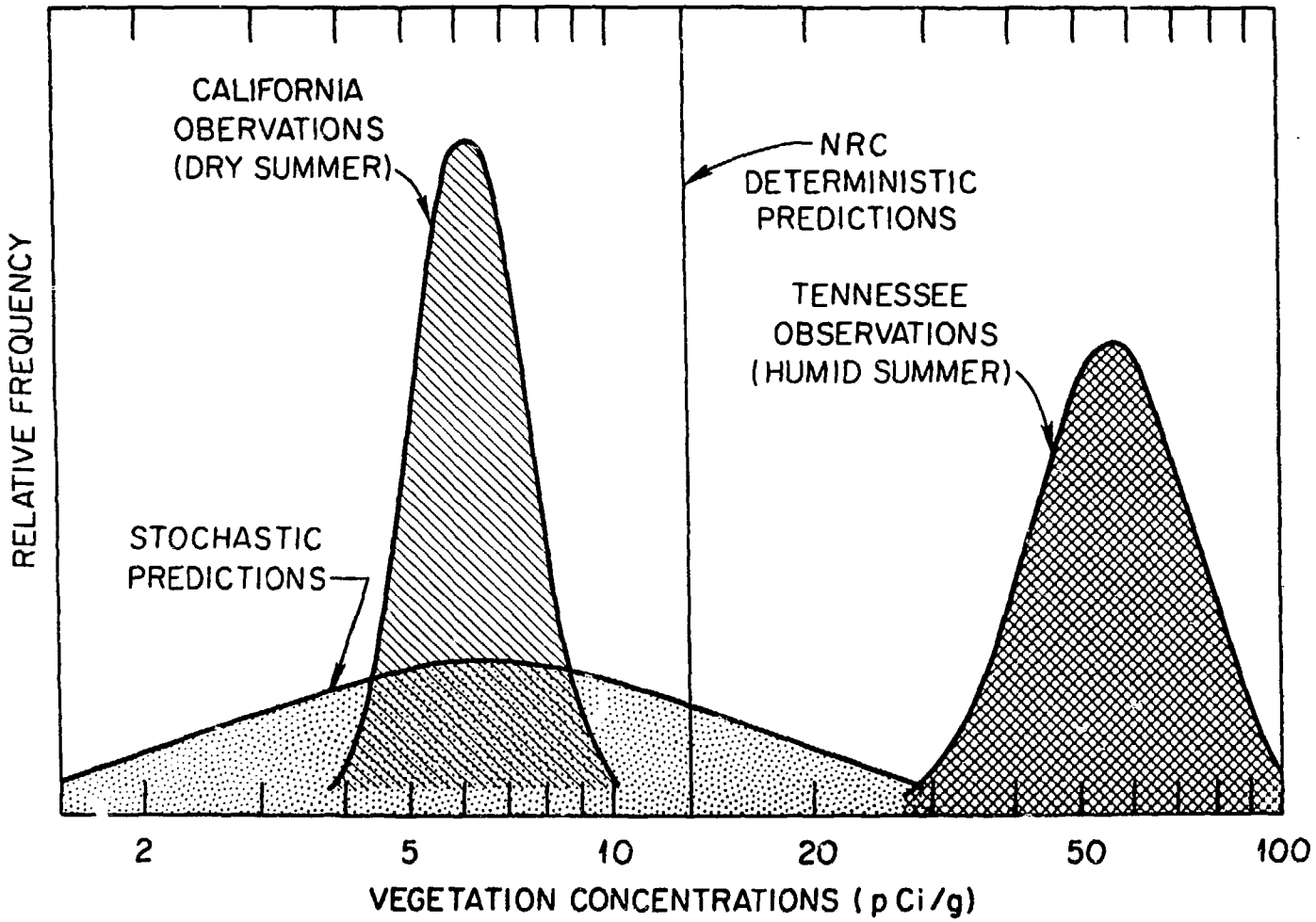

Fig. 6. A comparison of model predictions and observations of the transfer of submicron particulate radionuclides from air to leafy vegetation. The observations [68] are representative of two different climatic regimes of the United States. The predicted and observed quantities are normalized to an air concentration of 1 pCi/m^3. Stochastic predictions are obtained by specifying distributions for model parameters estimated from literature data [15,69]. Deterministic predictions were made using NRC Regulatory Guides 1.109 [28] and 1.111 [70], with deposition rates (from [70]) normalized for a given ground-level air concentration [71].

the two distributions of observed data. We suspect that this is the result of the manner in which parameter distributions were estimated from the general literature and the effect of unspecified correlations among model parameters.

Improving Confidence in the Use of Assessment Models

Given that there are numerous sources of uncertainties which can affect the accuracy of assessment models, how can we improve confidence in their use? In many cases, the importance of large uncertainties can be negated through the use of screening procedures, as long as conservatively biased-predicted quantities remain below dose limits or other levels of concern. Screening calculations would be most effective if official *de minimis* levels of risk could be established, because model uncertainties resulting in risks below *de minimis* levels could be properly dismissed as being of trivial importance.

In some cases, confidence in model predictions can be increased through better parameter estimation. For empirical assessment models, attempts should be made to improve parameter estimation through correlation with readily available site-specific information such as local meteorological conditions; soil pH, type, and texture; water quality; vegetation type; agricultural practices, etc. [2,3,19,20,30]. Site-specific parameter estimates are especially desirable within terrestrial and aquatic food-chain transport models [2,3,19,20,30,73]. For example, a major reduction in the overall variability of data for freshwater finfish bioaccumulation factors for cesium and strontium was demonstrated by Vanderploeg et al. [74] through correlations with the

suspended sediment content and the concentration of potassium and calcium in water (e.g., Fig. 7).

Increased understanding of biophysical processes and mechanisms of importance to human pathways of exposure will eventually improve the confidence with which model relationships can be extrapolated into situations different from those for which they were initially developed. Such fundamental improvements in knowledge are most likely to come from long-term investment in basic research. In the interim, confidence should be improved as model validation is extended to a variety of radionuclides and exposure pathways over a range of exposure conditions. The need for model validation has been widely recognized and urged by the numerous groups that have addressed questions of model uncertainties [3,19,25,30,73]. However, not all aspects of radiological assessment models are amenable to validation, the most notable examples being risk factors and internal dosimetry models. At least for the near term, we anticipate that risk factors and internal dosimetry models will remain important contributors to the irreducible uncertainty in radiological assessments.

Implications for Radiological Protection

The implications of model uncertainties for the protection of human health are greatest at high doses and dose rates. However, at low doses and dose rates comparable to recent regulatory limits for routine emissions from nuclear power facilities [4,5], the implications of model uncertainties for human health protection are small (if not negligible). Currently, the main implications of model uncertainties are associated with demonstration of compliance with regulatory

Fig. 7. Bioaccumulation factors for Cs-137 and stable cesium in freshwater fishes as a function of potassium concentration in water (from Vanderploeg et al. [73]).

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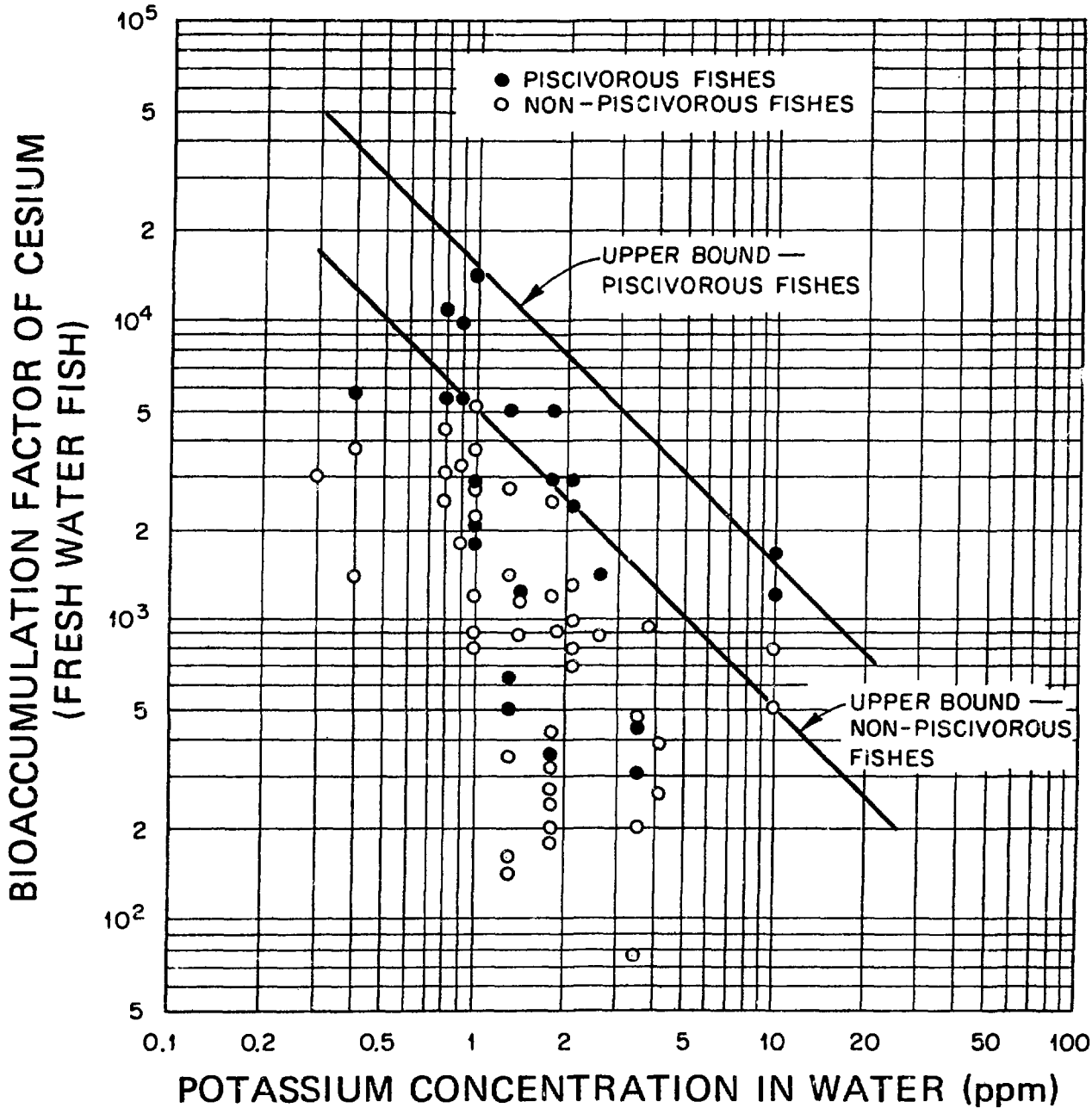

standards. Because some level of uncertainty will always be associated with model predictions, an absolute guarantee cannot be given that standards will not be exceeded.

Furthermore, the legal implications may be profound if regulations require that "no individual receive a dose that is greater than the specified dose limit," and models are used beyond their limits of intended application to demonstrate compliance with the limit. For this reason, we recommend that deterministic model predictions that are not conservatively biased be carefully scruitinized whenever calculated doses approach dose limits by an order of magnitude. Considering the relative consequences of model misprediction at low doses and low-dose rates, we contend that the implications of model uncertainties are of greater significance for legal, political, economic, and social issues than for the protection of human health.

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REFERENCES

1. Hoffman, F. O., and Kaye, S. V. "Terrestrial exposure pathways: potential exposures of man from the environmental transport of waste nuclides," in Proceedings of the International Symposium on the Management of Wastes from the LWR Fuel Cycle, CONF-760701, pp. 524-538, NTIS, Springfield, Virginia (1976).

2. Vaughan, B. E., Soldat, J. K., Schreckhise, R. G., Watson, E. C., and McKenzie, D. H. "Problems in evaluating radiation dose via terrestrial and aquatic pathways," Environ. Health Persp. 42:149-161 (1981).

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

4. U. S. Nuclear Regulatory Commission (USNRC). Numerical Guides for Design Objectives and Limiting Conditions for Operation to Meet the Criterion 'as Low as Practical' for Radioactive Material in Light-Water-Cooled Nuclear Power Reactor Effluents. Opinion of the Commission, Docket RM-50-2, USNRC, Washington, D.C. (1975).

5. U. S. Environmental Protection Agency (USEPA). 40 CFR Part 190, "Environmental radiation protection standards for nuclear power operations," Fed. Regist. 42(9)2858-61 (1977).

6. Federal Radiation Council. Background Material for the Development of Radiation Protection Standards, FRC Report No. 1, Federal Radiation Council, Washington (1960).

7. Baes, C. F., III, Sharp, R. D., Sjoreen, A. L., and Shor, R. W. A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture, ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1983, in press).

8. Gardner, R. H., O'Neill, R. V., and Hoffman, F. O. "Assessing model uncertainties," in Proceedings of a Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste (ed. D. C. Kocher), CONF-810372, pp. 199-208, NTIS, Springfield, Virginia (1982).

9. Hoffman, F. O., and Baes, C. F., III (eds.). A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides, ORNL/NUREG/TM-282, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1979).

10. Hoffman, F. O., and Gardner, R. H. "Evaluation of uncertainties in radiological assessment models," Chap. 11 in Radiological Assessments, U. S. Nuclear Regulatory Commission, Washington, D.C. (1983, in press).

11. Hoffman, F. O., Gardner, R. H., and Eckerman, K. F. Variability in Dose Estimates Associated with the Food Chain Transport and Ingestion of Selected Radionuclides, NUREG-CR-2612, USNRC, Washington, D. C., (1982).

12. Hoffman, F. O., Miller, C. W., Shaeffer, D. L., and Garten, C. T., Jr. "Computer codes for the assessment of radionuclides released to the environment," Nucl. Saf. 18(3):343-354 (1977).

13. Hoffman, F. O., Shaeffer, D. L., Baes, C. F., III, Little, C. A., Miller, C. W., Dunning, D. E., Jr., Rupp, E. M., and Shor, R. W. "An evaluation of uncertainties in radioecological models," Radioaktivitat und Umwelt II (1978) 610, Fachverband für Strahlenschutz e.V.

14. Miller, C. W. (compiler). 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, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1978).

15. Miller, C. W., and Hoffman, F. O. "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 (1982).

16. Miller, C. W., and Little, C. A. A Review of Uncertainty Estimates Associated with Models for Assessing the Impact of Breeder Reactor Radioactivity Releases, ORNL-5832, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1982).

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

18. Shaeffer, D. L., and Hoffman, F. O. "Uncertainties in radiological assessments — a statistical analysis of radioiodine transport via the pasture-cow-milk pathway," Nucl. Technol. 45:99-106 (1979).

19. National Council on Radiation Protection and Measurements. Radiological Assessment: Predicting the Transport, Bioaccumulation, and Intake by Man of Radionuclides Released to the Environment. Report of NCRP SC-64, NCRP, Bethesda, Maryland (1983, in press).

20. Ng, Y. C. "A review of transfer factors for assessing the dose from radionuclides in agricultural products," Nucl. Saf. 23:57-71 (1982).

21. Ng, Y. C., Colsher, C. S., and Thompson, S. E. Transfer Coefficients for Assessing the Dose from Radionuclides in Meat and Eggs, NUREG/CR-2976 (UCID-19464). Lawrence Livermore National Laboratory, Livermore, California (1982).

22. Ng, Y. C., Colsher, C. S., and Thompson, S. E. Soil-to-Plant Concentration Factors for Radiological Assessments, NUREG/CR-2975, UCID-19463, Lawrence Livermore National Laboratory, Livermore, California (1982).

23. Baker, D. A., Hoenes, G. R., and Soldat, J. K. FOOD — An Interactive Code to Calculate Internal Radiation Doses from Contaminated Food Products, BNWL-SA-5523, Battelle Northwest Laboratory, Richland, Washington (1976).

24. Haywood, S. M., Simmonds, J. R., and Linsley, G. S. The Development of Models for the Transfer of Cs-137 and Sr-90 in the Pasture-Cow-Milk Pathways using Fallout Data, NRPB-R110, National Radiological Protection Board, Chilton, Didcot, Oxon, United Kingdom (1980).

25. International Atomic Energy Agency. Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides in Predicting Exposures to Critical Groups from Routine Releases, Safety Series No. 57, IAEA, Vienna (1982).

26. Moore, R. E., Baes, C. F., III, McDowell-Boyer, L. M., Watson, A. P., Hoffman, F. O., Pleasant, J. C., and Miller, C. W. AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Atmospheric Releases of Radionuclides, EPA 520/1-79-009, Enviromental Protection Agency, Washington, D. C. (1979).

27. Simmonds, J. R., Linsley, G. S., and Jones, J. A. A General Model for the Transfer of Radioactive Materials in Terrestrial Food Chains, NRPB Report R89, National Radiological Protection Board, Chilton, Didcot, Oxon, United Kingdom (1979).

28. U. S. Nuclear Regulatory Commission (USNRC). 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, Regulatory Guide 1.109, rev. 1, October 1, USNRC, Washington, D. C. (1977).

29. Buckner, M. R. (ed.). Proceedings of the First SRL Model Validation Workshop, November 19-20, 1980 at Hilton Head, South Carolina, DP-1597 (1981).

30. Lindackers, H. H., and Bonnenberg, H. (organizers). Proceedings of a Workshop on Accuracy in Dose Calculations for Radionuclides Released to the Environment, September 10-14, 1979, Aachen, Federal Republic of Germany, GUW, Aldenhoven (1980).

31. Golden, J. C., Chandrasekaran, E. S., and Kahn, B. "Monitoring the critical radiation exposure pathways at a BWR nuclear power station," Health Phys. 42:777-788 (1982).

32. Voilleque, P. G., Kahn, B., Krieger, H. L., Montgomery, D. M., Keller, J. H., and Weiss, B. H. Evaluation of the Air-Vegetation-Milk Pathway for I-131 at the Quad Cities Nuclear Power Station, NUREG/CR-1600, USNRC, Washington, D. C. (1981).

33. Weiss, B. H., Voilleque, P. G., Keller, J. H., Kahn, B., Krieger, H. L., Martin, A., and Phillips, C. R. Detailed Measurement of I-131 in Air, Vegetation and Milk Around Three Operating Reactor Sites, NUREG-75/021, USNRC, Washington, D. C. (1975).

34. Weiss, B. H., and Keller, J. H. Enhancement of radioiodine concentration in milk due to wet processes in the environment. Paper 184 presented at the 1977 annual meeting of the International Radiological Protection Association, Paris, France (1977).

35. Kaye, S. V., Hoffman, F. O., McDowell-Boyer, L. M., and Baes, C. F., III. "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 (1982).

36. Helton, J. C., and Iman, R. L. "Sensitivity analysis of a model for the environmental movement of radionuclides," Health Phys. 42:565-584 (1982).

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

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

39. Garten, C. T., Jr. "Statistical uncertainties in predicting plutonium dose to lung and bone from contaminated soils," Health Phys. 39:99-102 (1980).

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

41. Schwarz, G., and Dunning, D. E. "Imprecision in estimates of dose from ingested Cs-137 due to variability in human biological characteristics," Health Phys. 43(5):631-645 (1982).

42. Cranwell, R. M., and Helton, J. C. "Uncertainty analysis for geologic disposal of radioactive waste," in Proceedings of the Symposium on Uncertainties Associated with the Regulation of the Geologic Disposal of High-Level Radioactive Waste, Gatlinburg, Tennessee, March 9-13, 1981 (D. C. Kocher, ed.) CONF-810372, pp. 131-144, NTIS, Springfield, Virginia (1982).

43. Iman, R. L., Davenport, J. M., and Ziegler, D. K. Latin Hypercube Sampling (Program User's Guide), SAND 79-1473, Sandia National Laboratory, Albuquerque, New Mexico (1980).

44. Iman, R. L., Conover, W. J., and Campbell, J. E. Risk Methodology for Geologic Disposal of Radioactive Waste: Small Sample Sensitivity Analysis Techniques for Computer Models, with an Application to Risk Assessment, NUREG/CR-1397, U. S. Nuclear Regulatory Commission, Washington, D. C. (1980).

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

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

47. Golomb, D., S. Batterman, and J. Bruhl, "Air quality screening model with long-term averaging," J. Air Poll. Control Assoc. 33(3): 215-219 (1983).

48. United Nations Scientific Committee on Effects of Atomic Radiation, Sources and Effects of Ionizing Radiation, UNSCEAR report, United Nations, New York, N. Y. (1977).

49. Kocher, D. C. (ed.). Proceedings of the Symposium on Uncertainties Associated with the Regulation of Geologic Disposal of High-Level Radioactive Waste, Gatlinburg, Tennessee, March 9-13, 1981, CONF-810372, NUREG/CP-0022, NTIS, Springfield, Virginia (1982).

50. Meyer, H. R. Personal communication to C. W. Miller (1983).

51. Tichler, J., and Benkovitz, C. Radioactive Materials Released from Nuclear Power Plants, Annual Report 1978, NUREG/CR-1497, BNL-NUREG-51192. U. S. Nuclear Regulatory Commission, Washington, D.C. (1981).

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

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

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

55. Brussermann, K. (ed.). Proceedings of the Workshop on Evaluation and Mitigation of Accidental Releases of Radioactivity — Identification of Uncertainties. Bonnenberg and Drescher, Aldenhoven, Federal Republic of Germany (1983, in press).

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

57. National Academy of Sciences, Committee on Biological Effects of Ionizing Radiation. The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, NAS, Washington, D.C. (1980).

58. Bates, R. "Critical appraisal of current toxicological approaches utilized in context of health risk evaluation," in Symposium on Health Risk Analysis, Gatlinburg, Tenn., October 27-30, 1980 (P. J. Walsh, ed.) Oak Ridge National Laboratory, Oak Ridge, Tennessee (1981).

59. Cohen, B. L. "Proposals on the use of the BEIR-III report in environmental assessments," Health Phys. 41(5):769-774 (1981).

60. National Council on Radiation Protection and Measurements. Review of the Current State of Radiation Protection Philosophy, NCRP Report No. 43, NCRP, Bethesda, Maryland (1975).

61. Berger, C. D., Lane, B. B., Cotter, S. J., Miller, C. W., and Glandon, S. R. Population Dose Estimation from a Hypothetical Release of 2.4 x 10^6 Curies of Noble Gases and 1 x 10^4 Curies of I-131 at the Three Mile Island Nuclear Station, Unit 2, ORNL/TM-7980, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1981).

62. Niemczyk, S. J., and McDowell-Boyer, L. M. Interim Source Term Assumptions for Emergency Planning and Equipment Qualification, NUREG/CR-2629, ORNL/TM-8274, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1982).

63. Cotter, S. J., Miller, C. W., Moore, R. E., and Little, C. A. Estimates of Dose Due to Noble Gas Releases from the Three Mile Island Incident Using the AIRDOS-EPA Computer Code, ORNL-5649, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1980).

64. Crick, M. J., and Linsley, G. S. An Assessment of the Radiological Impact of the Windscale Reactor Fire, October 1957, Report NRPB-R135, National Radiological Protection Board, Chilton, Didcot, Oxon, United Kingdom (1982).

65. Erb, J. C. "An assessment of the environmental transport of radioiodine in the air-grass-cow-milk pathway using reported environmental monitoring data," in Proceedings of the Health Physics Society 12th Midyear Topical Symposium, February 11-15, 1979, Williamsburg, Virginia (1979).

66. Nuclear Energy Agency. Comparison of Reactor Accident Consequence Models. Organization for Economic Co-operation and Development, Paris (1983, draft).

67. Schuttelkopf, H., and Pimpl, M. "Radioecological studies on plutonium and iodine-129 in the surroundings of the Karlsruhe reprocessing plant," in Environmental Migration of Long-Lived Radionuclides, IAEA-SM-257/100, pp. 395-414, International Atomic Energy Agency, Vienna (1982).

68. U. S. Environmental Protection Agency. 40 CFR Part 61, National Emission Standards for Hazardous Air Pollutants; Standards for Radionuclides, Federal Register 48:15076-91 (1983).

69. Bondietti, E. A. Unpublished data, Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1983).

70. Jonas, R., and Vogt, K. J. Untersuchungen zur Ermittlung der Ablagerungsgeschwindigkeit von Aerosolen auf Vegetation und andern Probenahmeflachen. Jul-1780. Kernforschungsanlage Julich GmbH, Federal Republic of Germany (1982).

71. U. S. Nuclear Regulatory Commission. Regulatory Guide 1.111, Revision 1, Methods for Estimating Atmospheric Transport and Dispersion of Gaseous Effluents in Routine Releases from Light-Water-Cooled Reactors, USNRC, Washington, D.C., 24 pp. (1977).

72. Miller, C. W., and Hoffman, F. O. "An analysis of NRC methods for estimating the effects of dry deposition in environmental radiological assessments," Nucl. Saf. 20:458-467 (1979).

73. U. S. Nuclear Regulatory Commission. Proceedings of the Workshop on Environmental Assessment held at National Bureau of Standards, Gaithersburg, Maryland, December 15-18, 1981, NUREG/CP-0025, USNRC, Washington, D. C. (1982).

74. Vanderploeg, H. A., Parzyck, D. C., Wilcox, W. H., Kercher, J. R., and Kaye, S. V. Bioaccumulation Factors for Radionuclides in Freshwater Biota, ORNL-5002, Oak Ridge National Laboratory, Oak Ridge, Tennessee (1975).