September 29, 1995

EPA-SAB-RAC-95-023

Honorable Carol M. Browner
Administrator
U.S. Environmental Protection Agency
401 M Street, SW
Washington, DC 20460

Re: Review of Technical Aspects of the Office of Radiation and Indoor Air's (ORIA) Technical Support Document (TSD) for the Development of Radionuclide Cleanup Levels for Soil

Dear Ms. Browner:

This report was developed by the Radionuclide Cleanup Standards Subcommittee (RCSS, also referred to as "the Subcommittee"), an ad-hoc subcommittee formed by the Radiation Advisory Committee (RAC) of the Science Advisory Board (SAB), in response to a request to review the technical aspects of the Agency's radionuclide cleanup levels for soils. The RCSS met on October 27 and 28, 1994, January 26 and 27, 1995, March 27, 1995 (teleconference), and May 23 and 24, 1995.

The enclosed report addresses the Charge and elaborates upon those technical aspects and issues which the Subcommittee believes would be most likely to require the attention of the Agency in order to provide the most comprehensive and technically-supportable basis for the radionuclide cleanup standards. This letter summarizes the report and highlights the most significant findings and recommendations associated with major elements of the Charge. The Charge elements and the Subcommittee's major responses follow:

Charge 1: Are the methodologies used by ORIA in the following areas acceptable for providing a technical basis for writing a cleanup standard: (a) methodology for evaluating source terms for radioactively contaminated sites, (b) methodology for modeling transport to people, and (c) methodology for estimating risk to individuals and populations?

Response: The framework for the overall approach taken by ORIA in the draft TSD represents a creative and reasonable approach to addressing risk reduction/cost tradeoffs in the soil cleanup standards for radionuclides. however, a major concern of the Subcommittee is that source term information appears weak even at the most well-defined sites, and ORIA had to be quite inventive at some sites. Recognizing that consistent site-wide data are limited, the Subcommittee commends ORIA for making good use of the available data and for its continuing efforts to work with other agencies to improve the site-related data base and to ensure appropriate utilization of the information collected. Still needed are estimates of uncertainties for contaminated soil volumes. Also, for many reference sites, it appears that the radionuclide selections were not sufficiently inclusive.

Charge 2: Are the assumptions and modeled pathways reasonable and suitable for assessing risk at radioactively contaminated sites: a) for the combined residential/agricultural land use scenario, and b) for the industrial-commercial scenario?;

Response: The Subcommittee is satisfied that ORIA's choice of an on-site residential scenario and a commercial-industrial scenario for thorough analysis is adequate for estimation of reasonable maximum individual exposures and risks from sites that have been cleaned up to a specified level of contamination. While other land-use scenarios are also plausible, they are not likely to produce substantially higher estimates of population risks when subsequently applied to estimate the number of cancer cases avoided at sites. However, the Subcommittee recommends that ORIA consider adding qualitative discussions of the likely magnitude of risks related to recreational and off-site residential neighbor scenarios.

Charge 3: Is RESRAD v. 5.19 suitable for modeling radiation risks to individuals at radioactively contaminated sites?

Response: The Subcommittee finds that the initial screening and selection of candidate models for further evaluation as reported in the TSD was conducted in a reasonable, sound and thorough manner, using appropriate criteria for model selection. Furthermore, the Subcommittee concurs with ORIA's decision in its selection of RESRAD as a reasonable transport model code for use at the current time. However, while it incorporates some conservative assumptions in its formulations, RESRAD itself may not necessarily provide conservative risk estimates if inappropriate parameter values are selected for the modeling input.

Because the Subcommittee has not evaluated the default values in the RESRAD code, nor the full parameter set used for each reference site, it is unable to assess fully the extent to which the model results can be considered to be conservative or bounding estimates of the true health effects associated with each level of cleanup. From a cursory review, several of the transport model parameter values appear to be inconsistent with values from the literature or known site characteristics; hence, the Subcommittee strongly urges EPA to obtain a thorough peer review of the default and site-specific parameter values used in the transport modeling.

In the face of constraints of time and resources for revisiting the pathway model definitions, ORIA should focus its efforts for any improvement of the definitions (i.e., underlying assumptions and adopted parameter values) on the dominant pathways, particularly external gamma radiation, radon inhalation, crop ingestion, and ingestion of groundwater, for selected reference sites. Ingestion of surface water should be investigated further to determine whether it might be an important pathway under any reasonable scenario.

Although the Subcommittee's specific Charge was to review scientific and technical aspects of the Agency's risk assessment methodology for sites with radioactive contamination, the Subcommittee's report also addresses some suggestions in areas that relate to policy choices and clearly go beyond a strict reading of the Charge. The most serious recurring problem for the Subcommittee was a difficulty in understanding how the Agency ultimately intends to take into account the interrelationships among its proposed regulatory actions to deal with soil cleanup, aquifer cleanup, cleanup of structures, and waste disposal. Although recognizing that the Agency is currently constrained by statutes or regulations to address these aspects as isolated problems, in truth, these problems are integrated, not isolated. The Subcommittee had difficulty commenting on some of the technical aspects of ORIA's TSD for soil cleanup, without having had the benefit of viewing it in the context of its proposed use. The current fragmentation of the peer review process leads to difficulties in reviewing estimates of overall exposures, risks and benefits. More scientifically robust estimates of overall exposure and risk would be derived from an integrated analysis.

Finally, the Subcommittee compliments the ORIA staff on its thorough documentation and forthcoming approach during the review. Compiling the information necessary to undertake this technical support document was obviously a formidable task, and ORIA's ability to organize, present and make use of this scattered information of variable quality is commendable. The Subcommittee also appreciated ORIA's prompt and thorough responses to many of the Subcommittee's technical comments in writing and in presentations, and that an excellent working relationship was established and maintained throughout the review process.

The RAC and its Subcommittee appreciates the opportunity to provide this report to you. We look forward to your response to this report, in general, and to the comments and recommendations in this letter, in particular.

Sincerely,

Genevieve M. Matanoski

Dr. Genevieve M. Matanoski, Chair
Science Advisory Board

James E Watson

Dr. James E. Watson, Jr., Chair
Radiation Advisory Committee
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NOTICE

This report has been written as a part of the activities of the Science Advisory Board, a public advisory group providing extramural scientific information and advice to the Administrator and other officials of the Environmental Protection Agency. The Board is structured to provide a balanced, expert assessment of scientific matters related to problems facing the Agency. This report has not been reviewed for approval by the Agency; hence, the comments of this report do not necessarily reflect the views and policies of the Environmental Protection Agency or of other Federal agencies. Any mention of trade names or commercial products does not constitute endorsement or recommendation for use.

ABSTRACT

The EPA Science Advisory Board's (SAB) Radiation Advisory Committee (RAC)/Radionuclide Cleanup Standards Subcommittee (RCSS) reviewed the Office of Radiation and Indoor Air's (ORIA) "Technical Support Document (TSD) for the Development of Radionuclide Cleanup Levels for Soil" (9/94). The RCSS supports ORIA's approach of defining a generic site to compare environmental pathway models; perform sensitivity/uncertainty analyses; and generate generic tables of cleanup soil concentrations for different land-use scenarios. ORIA defined reference facilities to represent sites and derived site-specific risk factors to estimate soil remediation volumes and health effects averted under each of the scenarios, for a range of cancer incidence and radiation dose cleanup goals. The RCSS was concerned that source term information appeared weak and the radionuclide selections were not sufficiently inclusive. The RCSS emphasized the need to estimate uncertainties for contaminated soil volumes. The RCSS concluded that the screening and selection of candidate transport models for risk assessment were sound, and concurred with the use of RESRAD. The RCSS evaluated neither the default values in RESRAD, nor the parameter set used for each reference site, and therefore was unable to assess whether the model results are bounding estimates of the risks for each level of cleanup. The RCSS recommended that EPA improve its definitions of the dominant pathways. The RCSS commended the EPA's sensitivity and uncertainty analyses, but felt that the TSD did not adequately convey the magnitude of the uncertainties in soil volumes requiring remediation and cancers averted by remediation.

Key Words: Cleanup Standards, Environmental Radiation, Nuclear Facilities, Environmental Quality, Radionuclide Transport, Radionuclide Cleanup

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1. EXECUTIVE SUMMARY

At the request of the Environmental Protection Agency (EPA), Office of Radiation and Indoor Air (ORIA), the Science Advisory Board (SAB), through the Radionuclide Cleanup Standards Subcommittee (RCSS, or the Subcommittee) of the Radiation Advisory Committee (RAC), has reviewed the Agency's September 1994 draft report titled "Technical Support Document for the Development of Radionuclide Cleanup Levels for Soil" (hereinafter called the "TSD"). The Subcommittee has responded to three specific questions posed by ORIA and has also provided additional comments and suggestions.[1]

The EPA ORIA staff is to be complimented on its thorough documentation and forthcoming approach to working with the Subcommittee. Compiling the information necessary to undertake this technical support document was obviously a formidable task, and ORIA's ability to organize, present and make use of this scattered information of variable quality is commendable. The Subcommittee also appreciated ORIA's prompt and thorough responses to many of the Subcommittee's technical comments in writing and in presentations.[2] The Subcommittee notes that an excellent working relationship was established and maintained throughout the review process.

1.1 General Findings

Because the TSD presented soil cleanup and related cleanup problems separately, a recurring problem for the Subcommittee was a difficulty in understanding how the Agency ultimately intended to take into account the interrelationships among its proposed regulatory actions to deal with soil cleanup, aquifer cleanup, cleanup of structures, disposal of radioactive waste generated by the cleanup, and recycle and reuse of materials and equipment after cleanup. Interactions among these five activities is expected to have a substantial effect on the estimated costs and benefits of the proposed standard, and hence should be discussed in the TSD, at least in a qualitative sense. Although recognizing that the Agency is currently constrained by statutes or regulations to address these aspects as isolated problems, in truth, these problems are integrated, not isolated. The Subcommittee had difficulty commenting on some of the technical aspects of the TSD for soil cleanup, without having had the benefit of viewing it in the context of its proposed use. The current fragmentation of the peer review process leads to difficulties in

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[1] Numbers associated with each finding or recommendation indicate the section of this review report from which it derived, i.e., discussion supporting finding 8.8.a is found in section 8.8.

[2] In fact, many of the issues raised in this report were adequately addressed by ORIA while the report was being prepared (e.g., through memoranda and presentations by ORIA staff members at RCSS meetings. However, although acknowledging this fact, the Subcommittee has retained its comments on the contents of the document under review.

reviewing estimates of overall exposures, risks and benefits. More scientifically robust estimates of overall exposure and risk would be derived from an integrated analysis. (8.8.a)

1.2 Response to Issue #l

Issue #1. Are the methodologies used by ORIA in the following areas acceptable for providing a technical basis for writing a cleanup standard:

a) methodology for evaluating source terms for radioactively contaminated sites?
b) methodology for modeling transport to people? and
c) methodology for estimating risk to individuals and populations?

1.2.1 Findings on Evaluation of Source Terms

The overall approach taken by ORIA in the draft TSD represents a creative and reasonable approach to addressing risk reduction/cost tradeoffs in the soil cleanup standards for radionuclides. Specifically, the Office has defined a generic site as a basis for comparing the behavior of several different environmental pathway models; for carrying out sensitivity and uncertainty analyses; and for generating preliminary generic tables of cleanup soil concentrations for different land-use scenarios. A suite of reference facilities was defined by ORIA to represent the full spectrum of sites to be covered by the proposed rule, and site-specific risk factors were calculated and employed to develop estimates of total soil remediation volumes and health effects averted under each of the scenarios, for a range of cleanup goals stated in terms of lifetime cancer incidence risk or annual radiation dose. Commendable also are ORIA's efforts to collect information on radioactive sites, to construct the source terms for these sites, to test the sensitivity of its assumptions, and to analyze the uncertainty of its results. (3.1.a)

A major concern of the Subcommittee is that source term information appears weak even at the most well-defined sites, and ORIA had to be quite inventive at some sites. Recognizing that consistent site-wide data are limited, the Subcommittee commends ORIA for making good use of the available data and for its continuing efforts to work with other agencies to obtain site-related data and to ensure appropriate utilization of the information collected. Still needed are quantitative estimates of the uncertainties for the contaminated soil volumes. Also, for many reference sites, it appears that the radionuclide selections were not sufficiently inclusive. Most of these sites contain multiple radionuclides, and different combinations may affect the calculated waste volumes and allowable concentrations substantially. Specifying the chemical and physical forms of specific radionuclides in the individual reference sites would be desirable not only for predicting leaching rates and transport rates in groundwater, but also for those cases in which the form significantly affects human intake, as in the case of uranium at Fernald or in munitions at Department of Defense facilities. (3.3.a)

For example, additional discussion is needed in the TSD to justify the use of selected reference sites to represent other sites in their categories that have different types of radionuclides, and different hydrological, geological and meteorological conditions. The Subcommittee was skeptical of the appropriateness of using the Oak Ridge Reservation to represent the five major DOE facilities involved with diversified weapons research and development activities, given that the total volume of contaminated soil in this category is dominated by the Los Alamos National Laboratory site. (3.2.b)

1.2.2 Findings on Modeling Transport to People

With the exception of a surface water runoff and erosion pathway with subsequent potential for drinking water and fish consumption exposures, ORIA has generally included in the RESRAD analysis all the pathways that are likely to be important for radionuclides in inorganic forms. (Direct and indirect dermal absorption pathways might be important for organic chemicals and mixed wastes.) (4.3.b)

In the face of constraints of time and resources for revisiting the definitions of the pathways, ORIA should focus its efforts for any improvement of the definitions (i.e., underlying assumptions and adopted parameter values) on the dominant pathways, particularly external gamma radiation, radon inhalation, crop ingestion and ingestion of groundwater (especially the population risk assumptions). Ingestion of surface water should be investigated further to determine whether it might be a dominant pathway under any widely prevalent conditions. (4.3.a)

Major comments on specific pathway models are as follows:

a) The <u>external radiation pathway</u> is relatively simple to define. With the possible exception of exposure time, ORIA's modeling of this pathway is generally acceptable. Of the three codes considered in detail in the generic base case study, RESRAD is the most realistic and flexible in its capabilities to account for the processes that govern exposure of individuals by this pathway. The assumptions and parameter values for the rural/residential and commercial/industrial scenarios which affect the dose from direct gamma radiation are reasonable and are, presumably, incorporated into the RESRAD model used in the analysis. The RESRAD equations for correcting for depth of contamination, cover thickness and source area are also reasonable. However, the Subcommittee recommends that ORIA be more explicit regarding its assumptions about the shielding factor for indoor exposures and be sure that the parameter value used is consistent with those assumptions. (5.2.a to 5.2.d)

b) With the generic qualification regarding non-uniformity of source term contamination and population mobility, the soil and food ingestion pathways appear to be consistent in form with currently available methods although the use of a larger default value for soil ingestion by an RME individual should be considered. The Subcommittee recommends an expansion of the discussion of the uncertainty and variability in the risk estimates generated under the simplified assumptions of these pathway models. Some of the default parameter values used by ORIA serve to underpredict exposure by this pathway. For example, growth of crops on contaminated soils will be a major pathway of concern for ^{90}Sr, ^{129}I, ^{99}Tc, and ^{137}Cs on sites with nutrient-poor and highly acidic soils. Consequently, ORIA should revise its soil-to-plant transfer factors for those sites at which radionuclide uptake by plants is expected to be higher than the default value. (5.3.a to 5.3.d)

c) For the particle inhalation pathway, the Subcommittee recommends that the TSD clearly specify all parameters used in calculations of dose and risk from inhalation of airborne radioactive dusts for the three identified exposure scenarios for cleanup workers, and indicate ranges of values as well as those assumed or adopted for the calculations. Although population density is addressed for all generic sites, the characteristics of the populations themselves do not seem to be addressed with respect to age, living behaviors or housing types. In addition, this pathway assumes that inhalation exposure can be described by the amount of dust expected to be in the air and that the concentration of radionuclides in the dust is the same as that in the soil under consideration. For a small site, much of the dust in the air even at the downwind edge of the site will come from unaffected regions upwind of the site, and the ORIA assumption could lead to substantial overestimates of risk via this pathway. Moreover, real atmospheric dust loadings vary substantially with site and depend on such factors as soil particle size distributions, vegetative cover, humidity, and precipitation patterns; such variations have not been taken into account in the modeling of the reference sites. (5.4.a to 5.4.e)

d) The methodology used for estimating population risks from exposure to radon indoors is reasonable, and the risk conversion factor used in this calculation previously has been reviewed by the RAC. The total uncertainty associated with individual and population risk estimates should be presented. Of concern is the orders of magnitude variability in the radon entry into individual homes and the large uncertainty that will exist in a single calculated value that is used to estimate the radon concentration in any home. The methodology used in RESRAD for estimating individual risks does not account for advective flow of radon into a home, and this omission can result in underestimation of indoor radon concentrations. (5.5.a to 5.5.h)

e) With regard to the groundwater pathway, the Subcommittee found aspects of some of the reference site models to be overly conservative, using parameter values that were internally inconsistent, or inconsistent with values from the literature or known site characteristics. Examples include hydraulic conductivities and gradients, well construction designs, groundwater withdrawal rates, and distribution coefficients (i.e., K_d values) for those long-lived radionuclides for which the chemical form is highly sensitive to the prevailing oxidation-reduction potential (e.g., ^{99}Tc, U, Pu). Additional work in the form of sensitivity analyses and peer review of parameter values is required before the importance of the drinking water pathway can be fully assessed, for those sites at which the contamination does not remain at the soil surface or within the unsaturated zone throughout the modeled period of time. (5.6.a to 5.6.c)

f) In the surface water pathway, contributions from particulate-phase radionuclides associated with soil that is eroded from the site as part of the rainfall-runoff process can be a significant source of contamination for surface water bodies when large portions of the watershed areas are contaminated. For these cases, the Agency should encourage and seek inclusion of soil erosion and particulate-phase radionuclides in the surface water modules of future models used for assessment of soil cleanup standards. (5.7.a)

1.2.3 Findings on Risk Estimation Methods

The set of risk coefficients used in the TSD for risk-based standards differs from that used for dose-based standards, and neither set incorporates the latest changes recommended by the ICRP or NCRP. This choice can lead to considerable confusion on the part of the reader. While the Subcommittee understands the difficulties ORIA faced in deciding how to reconcile its internally generated risk coefficients with those in RESRAD derived from Federal Guidance Reports Nos. 11 and 12, leaving the problem unresolved can lead the unwary user of the TSD to erroneous conclusions. Although the Subcommittee is not specifically recommending that ORIA undertake the magnitude of effort needed to produce a completely consistent document, it does make the following findings and recommendations:

a) The TSD should include a section specifically describing the dose conversion factors and slope factors used for each of the types of assessments. While this information can for the most part be gleaned from the various chapters, it is currently difficult to sort out. Deviations from published values should be adequately explained and the methodology for making adjustments described. (6.6.c)

b) The Subcommittee recommends that the TSD include a discussion of the proposed annual dose limit as it applies to cleanup. The selection of appropriate assumptions and parameter values for modeling RME exposures will depend upon whether the standard is developed as an annual dose limit for "any" member of the public or as an average annual measure of compliance with the long-term (lifetime) individual risk limit. If the annual limit is intended to serve as a surrogate for the lifetime risk limit, then the approach used in the TSD is generally acceptable and the importance of the recommendation for age-specific factors presented below is diminished. If, however, the goal of the standard is to assure that every individual is explicitly protected to the annual limit, then it will be important to consider the use of age-dependent dose factors in the TSD analysis. (6.6.a)

c) The Subcommittee recommends that EPA give consideration to adoption of the recent recommendations of the ICRP and NCRP that provide updated metabolic and dosimetric models and approaches for calculating age-dependent doses for the inhalation and ingestion of radionuclides for all members of the public. Adoption of the ICRP and NCRP approaches offers a procedural advantage to the EPA in that they have been extensively peer reviewed and are widely accepted. Use of these approaches would likely increase the technical acceptance of the standards by the scientific community. If alternative approaches are used in the TSD, then ORIA should explicitly explain its methodology and justify its reasons for departures from ICRP and NCRP recommendations. (6.6.b)

d) The Subcommittee recognizes that, by Presidential directive in 1987, EPA should use the exposure-to-dose conversion factors tabulated in Federal Guidance Reports Nos. 11 and 12 and their subsequent revisions. This guidance is based on a "linear extrapolation to zero" exposure-to-dose relationship from observed, but much higher, dose-effect studies. As noted in the TSD, the scientific community has been unable to come to a consensus on issues such as the possibility of threshold doses below which no effects occur, the validity of extrapolating curves from known high exposure effects to zero, and the possibility of hormesis (the concept that small doses of radiation may be beneficial to humans). The Subcommittee recommends that the uncertainties associated with extending risk analyses to very low radiation exposures in the absence of scientific consensus be reflected in the presentation of the final results. (6.6.d)

e) A comparison was made of EPA and NRC estimates of contaminated soil volumes and cancer risks associated with that soil, for commercial nuclear power reactors, research reactors, rare earth extraction facilities, and uranium fuel fabrication facilities. The EPA estimates of soil volume were significantly larger than NRC estimates, by factors up to 100. For three of these four types of facilities, the estimates of the number of fatalities associated with contaminated soil were generally comparable, within a factor of ten of each other. The exception was the case of the uranium fuel fabrication facility, for which mortality estimates in the TSD were higher than those calculated by the NRC by factors ranging from 2.5 to 100. The reasons for this difference are not clear and should be investigated by ORIA. (6.6.h)

1.3 Response to Issue #2 (Suitability of Modeled Pathways)

Issue #2. Are the assumptions and modeled pathways reasonable and suitable for assessing risk at radioactively contaminated sites:

a) for the combined residential / agricultural land use scenario? and
b) for the industrial / commercial scenario?

The Subcommittee is satisfied that ORIA's choice of an on-site residential scenario and a commercial-industrial scenario for thorough analysis is adequate for estimation of reasonable maximum individual exposures and risks from sites that have been cleaned up to a specified level of contamination. While other scenarios are also plausible, they are not likely to produce substantially higher estimates of maximum individual risks. However, the Subcommittee recommends that ORIA consider adding qualitative discussions of the likely magnitude of maximum individual risks related to recreational and off-site residential neighbor scenarios. (4.2.a)

1.4 Response to Issue #3 (Suitability of RESRAD)

Issue #3. Is RESRAD v. 5.19 suitable for modeling radiation risks to individuals at radioactively contaminated sites?

The Subcommittee finds that the initial screening and selection of candidate models for further evaluation as reported in the TSD was conducted in a reasonable, sound and thorough manner, using appropriate criteria for model selection. Furthermore, the Subcommittee concurs with ORIA's decision in its selection of RESRAD as a reasonable transport model code for use at the current time. However, while it incorporates some conservative assumptions in its formulations, RESRAD itself may not necessarily provide conservative risk estimates if inappropriate parameter values are selected for the modeling input. (5.1.a, 5.1.b)

Because the Subcommittee has not evaluated the default values in the RESRAD code, nor the full parameter set used for each reference site, it is unable to fully assess the extent to which the model results can be considered to be conservative or bounding estimates of the true health effects associated with each level of cleanup. From a cursory review, several of the transport model parameters appear to inconsistent with values from the literature or known site characteristics; hence, the Subcommittee strongly urges ORIA to obtain a thorough peer review of the default and site-specific parameters used in the transport modeling. As more information becomes available through site applications and studies, the Agency should re-evaluate this issue to determine whether RESRAD—including its default parameters—should be modified or replaced so as to ensure the maintenance of an appropriate balance between realistic prediction and reasonably conservative protection of public health. (5.1.c)

1.5 Findings on Sensitivity and Uncertainty Analyses

Overall, the Subcommittee commends ORIA on conducting sensitivity and uncertainty analyses for the risk and soil volume calculations and on its thoughtful discussions of the important assumptions and parameter value choices. However, because of limitations of the analyses, the reader is left without a sound appreciation for the magnitude of the overall uncertainties in soil volume requiring remediation and cancers averted via remediation. The generic sensitivity and uncertainty analyses are limited by the choice of parameter values that were varied, the need to select nominal values for all other values when varying one parameter, and the lack of analysis regarding model uncertainties. Although quantitative sensitivity analyses of the reference sites are conducted with respect to policy choices such as the target risk level, time horizons, and land-use scenarios, only a qualitative discussion of scientific uncertainties is offered. Thus, the true but unknown values for cancers averted and soil volumes to be remediated at each RME risk level may be quite different from those presented in the report. Given that policy decisions are to be made using the numerical results of the TSD models as one criterion, it is critical that the quantitative uncertainties about those results be disclosed and emphasized in the presentation. (7.7.a)

The importance of uncertainty analyses and communication of the results of those analyses to environmental risk managers was underscored by the EPA Administrator in a recent memorandum transmitting the EPA Risk Characterization Program to EPA staff (EPA, 1995).

Consistent with the spirit of that guidance, the Subcommittee recommends that ORIA improve its risk assessment and characterization in the TSD in the following areas:

a) Discussion in the TSD should clarify the purpose of the risk calculations: are these to be screening or bounding estimates, high end estimates (e.g., above the 90th or 95th percentile), or central tendency estimates of the true value (risk)? The objective then should guide the selection of appropriate parameter values used in pathway modeling. (7.7.b)

b) ORIA should, at a minimum, discuss qualitatively any biases in its estimates of cancers averted as well as biases in the soil volumes to be remediated. The positive correlation of these biases with any in the individual risk estimates should also be mentioned. (7.7.c)

c) To the extent possible, ORIA should provide best estimates of cancers averted total and fatal) and soil volumes to be remediated for the various proposed standards, as well as uncertainty ranges about each of those estimates, in addition to the nominal values currently provided. (7.7.d)

d) In the future, when EPA evaluates the need for cleanup at a specific site in response to the final cleanup standard, it should not only allow but also explicitly encourage quantitative uncertainty analyses in the site assessment. The level of detail in such uncertainty analyses should be commensurate with the stakes (potential for risk reduction and cleanup costs) revealed in a screening analysis. (7.7.e)

1.6 Comments on Issues at the Science/Policy Interface

The Subcommittee's charge was to review scientific and technical aspects of the methodology used by ORIA to model radiation risks to individuals at radioactively contaminated sites. The results of the risk assessments will be used by the Agency in making policy decisions on cleanup levels for soil. The review by the Subcommittee primarily focused on the evaluation of source terms, environmental transport, and estimation of risks. However, in the course of this work, some issues were identified that were outside the scope of the Subcommittee's charge. The following comments on these issues, which involve the interface of science and policy, are provided for consideration by the Agency.

Risk metric. EPA's decision to use lifetime risk corresponding to reasonable maximum exposure as a risk metric for the proposed standard is appropriate. Although another metric (such as the population risk attributable to a facility) could have been used, the formulation of the standard is consistent with other EPA waste management strategies. In its comparison of the risks of remedial activities to those avoided through the remediation, EPA uses population risk (cancer fatalities caused vs. cancer cases averted). While this choice is also reasonable, it bypasses the issue of potentially higher individual risks to some remediation workers. The Subcommittee recommends that EPA be more explicit about the reasons it chose to use an individual risk metric for the standard and a population risk metric for describing the costs and benefits of the standard. A brief mention of the potential for higher-than-average individual risks to remediation workers is also recommended. (8.8.c, 8.8.d)

Time horizons. By estimating risks over time horizons of 100, 1,000, and 10,000 years, EPA has also provided the cleanup standard decision-makers with a range of options for evaluating the benefits of the standard. However, the Subcommittee is concerned that estimates for risks incurred more than 100 years in the future are highly speculative given the uncertainties about the rate and direction of change in medical science and other technological and social arenas which could either reduce or accentuate the relative importance of cancer as a health risk in the future. The Subcommittee therefore recommends that EPA emphasize the increased uncertainty of its estimates for 1,000 and 10,000 years in comparison with those for 100 years. (8.8.e)

Inconsistency with existing EPA regulations. The analyses of risk associated with ^{226}Ra suggest that cleanup levels below 10^{-3} are probably not feasible because of the natural variation in the abundance of this isotope. For comparison, the cleanup standard for ^{226}Ra at uranium mill tailings sites is 5 pCi/g above background in the top 15 cm of soil and 15 pCi/g above background in deeper layers (UMTRCA regulation, 40 CFR 192), corresponding to lifetime risks of approximately 10^{-2}. This illustrates a lack of consistency between the lower risk levels being considered in the TSD and those corresponding to existing regulations dealing with radionuclide cleanup, such as the UMTRCA regulation. (8.8.f)

Potential applicability to Naturally Occurring Radioactive Material (NORM). The Subcommittee is not offering an opinion on the advisability of using the proposed cleanup standard as an ARAR (Applicable or Relevant and Appropriate Regulation) or as the precedent for any NORM regulations that might be proposed in the future. However, these possibilities should be noted in the TSD, and the costs and benefits of any such actions should be discussed in the Regulatory Impact Analysis. The Subcommittee notes that the TSD does not provide the technical basis for cleanup criteria for NORM because the analyses presented in the TSD do not address cancers averted and volumes of soil affected for sources of NORM. Also, the feasibility issue noted in the preceding recommendation would be pertinent because ^{226}Ra is one of the principal NORM radionuclides. (8.8.g)

2. INTRODUCTION

2.1 Overview of EPA Modeling Objectives and Strategy

The EPA is proposing regulations that would set standards for radiation doses from contaminated soil prior to release of the land for unrestricted public use. The proposed regulations are meant to apply to contaminated soil remaining after cleanup of high-risk ($\geq 10^{-2}$ lifetime cancer incidence risk level) contaminated soil and facilities. As such, the proposed regulations could affect large volumes of lower-risk ($< 10^{-6}$ to 10^{-2} risk level) contaminated soils at sites presently under the control of a Federal Agency and sites licensed by the U.S. Nuclear Regulatory Commission (NRC) or by an NRC Agreement State, that are to be released from those licenses or control. The regulations also have the potential to affect large volumes of soil at mining sites if the proposed cleanup standards extend to sites containing naturally-occurring radioactive materials (NORM).

The major objectives of the EPA report are to: (a) estimate the volume of soil that may require remediation at sites that fall within the scope of the proposed rule, and (b) estimate the number of potential radiogenic cancers averted as a result of the remediation of contaminated soil.

The approach taken by EPA is to consider typical scenarios and pathways through which individuals and populations may be exposed. Then, using mathematical models, cleanup levels are determined that will yield acceptable predicted risks. The modeling process consisted of development of selection criteria. Using these criteria, a model evaluation and selection was performed which resulted in the selection of the following models: RESRAD, PRESTO, and RAGS/HHEM. These models were then compared and tested using a hypothetical generic base case.

As part of the generic base case study, a limited sensitivity analysis was performed using RESRAD Version 5.19. The five parameters that were varied included area of the contaminated zone, thickness of the contaminated zone, infiltration rate, distribution coefficients, and thickness of the unsaturated zone. Also, as part of the generic base case study, a limited uncertainty analysis was performed using a modified version of RAGS/HHEM Part B models and Monte Carlo techniques. The base case study included evaluation of both individual impacts and impacts to an aggregate population. Individual impacts were estimated using RESRAD while population impacts were estimated using a population model developed by ORIA for the TSD analyses.

Based on results of the model comparisons for the generic base case site assessment, the RESRAD code was selected to analyze the set of reference sites. A result of the reference site analysis was the development of risk factors for each radionuclide. From the risk factors, one can estimate the volumes of soil requiring remediation in order to achieve the proposed standard's risk level. To estimate the volume of soil that may require remediation, the approach taken is to calculate the risk resulting from a reasonable maximum exposure (RME) to an individual either on the site and consuming locally-produced agricultural products, or working on the site in an industrial occupation. The risks are calculated for these individuals based on radioactive contamination levels remaining after remediation of the site. To calculate the RME risk to an individual at a given site, site data and generic data are used in conjunction with the risk assessment code RESRAD.

To estimate the numbers of potential fatal radiogenic cancers averted as a result of remediation of contaminated soil, risk levels calculated for individual exposures are combined with various population scenarios. The resulting numbers of cancers averted are given in terms of various individual risk levels ranging from $< 10^{-6}$ to 10^{-2}, as well as for various dose limits ranging from 0.1 to 100 mrem/yr, together with remediated soil volumes corresponding to each of these risk and dose levels.

2.2 Charge to the SAB

At the request of the Environmental Protection Agency (EPA), Office of Radiation and Indoor Air (ORIA), the Science Advisory Board (SAB), through the Radionuclide Cleanup Standards Subcommittee (RCSS, or the Subcommittee) of the Radiation Advisory Committee (RAC), has reviewed the Agency's September 1994 draft report titled "Technical Support Document for the Development of Radionuclide Cleanup Levels for Soil" (hereinafter called the "TSD"). The RSCC responded to the following three questions posed by EPA and has also provided additional comments and suggestions for its consideration.

Issue 1. Are the methodologies used by ORIA in the following areas acceptable for providing a technical basis for writing a cleanup standard: (a) methodology for evaluating source terms for radioactively contaminated sites, (b) methodology for modeling transport to people, and (c) methodology for estimating risk to individuals and populations?

Issue 2. Are the assumptions and modeled pathways reasonable and suitable for assessing risk at radioactively contaminated sites, (a) for the combined residential / agricultural land use scenario, and (b) for the industrial and commercial scenario?

Issue 3. Is RESRAD v. 5.19 suitable for modeling radiation risks to individuals at radioactively contaminated sites?

2.3 SAB Review Procedure

The primary review document is the Agency's September 1994 draft report titled "Technical Support Document for the Development of Radionuclide Cleanup Levels for Soil" (TSD), including its Appendices A-O (EPA, 1994a). The Subcommittee's review also benefited from the extensive documentation voluntarily provided by ORIA in support of the TSD.

The RCSS met on October 27-28, 1994, January 26-27, 1995, and May 23-24, 1995, at which time it was briefed by ORIA staff on specific aspects of the TSD, including revisions in progress since issuance of the September 1994 TSD draft. In addition, the RCSS conducted a teleconference on March 27, 1994.

The RCSS wishes to compliment the ORIA staff on its thorough documentation and forthcoming and candid approach to working with the RAC's Subcommittee. Compiling the information necessary to undertake this technical support document was obviously a formidable task, and ORIA's ability to organize, present and make use of this scattered information of variable quality is commendable. The Subcommittee also appreciated ORIA's prompt and thorough responses to many of the Subcommittee's technical comments in writing and in presentations.[3] The Subcommittee notes that an excellent working relationship was established and maintained throughout the review process.

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[3] In fact, many of the issues raised in this report were addressed by ORIA while the report was being prepared (e.g., memoranda to RCSS members and various presentations by EPA staff members at RCSS meetings. However, although acknowledging this fact, the Subcommittee has retained its comments on the contents of the document under review.

3. SOURCE TERM ISSUES

3.1 Introduction

To provide a technical and scientific basis for the tradeoffs between risk reduction and cost in the proposed soil cleanup standards for radionuclides, ORIA defined a generic site, reference sites, and several land-use scenarios. ORIA then coupled this information with a calculational model in order to estimate the total soil volume that would require remediation under each of the scenarios, for a range of cleanup goals stated in terms of soil specific activity required to achieve reductions to specific dose levels and lifetime cancer risk levels. Using the generic site , three calculational models were compared; of these, the RESRAD model was chosen to calculate site-specific dose or risk factors from which soil specific activity and clean-up volume were derived for each reference site and a particular dose or lifetime cancer risk level. Additional information was generated for cancer deaths averted by the remedial action, cancer risks to workers undertaking the cleanup activities, and radiation doses off-site using a population effects model devised by ORIA for this purpose.

Calculations for the generic site provide dose factors in mrem/yr per pCi/g for each radionuclide of interest so that the cleanup may be performed in terms of soil specific activity. Furthermore, a sensitivity analysis was performed using the generic site to address some of the issues regarding uncertainties in the choice of the dimensions, and the physicochemical characteristics of the site that relate to geologic processes such as radionuclide transport and leaching. Calculations for the reference sites, selected to represent the variety of facilities that have radioactively contaminated soils, also provide dose factors in mrem/yr per pCi/g for each radionuclide of interest. These suggest the soil volumes at specified dose or risk levels to guide the policy choice that cleanup to such levels would be a reasonable requirement to ensure the public's health and safety if the site were to be released for unrestricted or restricted use. The appropriateness of the generic site is confirmed if dose or risk factors at the reference sites are comparable. The appropriateness of the reference site approach is confirmed if the resultant calculations capture the universe of sites to be remediated under the proposed rule.

The source term for the generic site is contaminated soil at a density of 1.5 g/cm^3 in an area of 100 m x 100 m, from a flat surface to a depth of 2.0 m. Uniform water flow rates and distribution coefficients for radionuclides are specified. All radionuclides, natural or manmade, that could be expected to be present at a site to be decontaminated or decommissioned are considered if they have a half life longer than about 0.5 year or are short-lived progeny of such radionuclides. Each radionuclide is assumed to be uniformly distributed throughout the soil at a concentration of 1 pCi/g. The main questions about the generic site that relate to the source term concern the following: (a) site dimensions, (b) selection of radionuclides to be included in the model exercise, and (c) influence of the chemical and physical form of the radionuclides on their transport characteristics.

The source term for each individual reference site consists of a zone of specified area and thickness contaminated with one or more radionuclides; these radionuclides are subsequently redistributed throughout the environment over time as a function of various hydrologic parameter values and radionuclide distribution coefficients (TSD Tables 4-5 to 4-8). The radionuclides that have been identified and measured by soil monitoring, and that are considered to pose possible health risks, are assumed to be distributed uniformly throughout the soil in the vertical direction to a specified depth. The depth of contamination is either calculated, taken from the literature, or assumed to be 5 cm. Horizontally, the radionuclides are assumed to be distributed nonuniformly in accord with isopleths constructed on the basis of monitoring overflights or surface samples. The reference site is assumed to be a selected portion of an actual site that excludes structures and waste disposal areas, as well as locations considered uncontaminated. The resulting soil volumes are multiplied (when necessary) by a weighting factor so that the reference site represents all the facilities in that category.

The Subcommittee presents the following findings concerning the overall approach used by ORIA to define the source term:

3.1.a ***Finding*** The overall approach taken by ORIA in the draft TSD represents a creative and reasonable approach to addressing risk reduction/cost tradeoffs in the soil cleanup standards for radionuclides. Specifically, ORIA has defined a generic site as a basis for comparing the behavior of several different environmental pathway models; for carrying out sensitivity and uncertainty analyses; and for generating preliminary generic tables of cleanup soil concentrations for different land-use scenarios. A suite of reference facilities was defined by ORIA to represent the full spectrum of sites to be covered by the proposed rule, and site-specific risk factors were calculated and employed to develop estimates of total soil remediation volumes and health effects averted under each of the scenarios, for a range of cleanup goals stated in terms of lifetime cancer incidence risk or annual radiation dose. ORIA's efforts to collect information on radioactive sites, to construct the source terms for these sites, to test the sensitivity of its assumptions and to analyze the uncertainty of its results are also commendable.

3.1.b ***Finding***. With regard to the generic site source term, the list of radionuclides in Table 2.6 of the TSD appears to be complete and the associated data correct.

Evaluation of the TSD concerning the source terms for individual reference sites is complex due to the number of sites, the differences among them, and the assumptions made by the EPA in its analysis. Comments are presented below in five sections: selection of representative reference sites for each category (section 3.2), extent and completeness of the information obtained for reference sites (3.3), acceptability of modeling of source term information for reference sites (3.4), development of the distribution functions for soil volume vs. activity (3.5), and adequacy of the sensitivity and uncertainty analyses of source term parameters (3.6).

3.2 Selection of Representative Reference Sites

EPA conducted additional analyses on individual reference sites in order to: (a) validate risk factors (e.g., risk per residual pCi/g) calculated for the generic site, and (b) identify sites for which risk factors calculated for the generic site would be inappropriate.   Facilities were selected by EPA to represent the full spectrum of sites in order to derive estimates of the soil volume that would require remediation under each of the scenarios, for a range of cleanup goals stated in terms of dose or lifetime cancer risk. EPA correctly recognized that analyses of actual sites would provide more information than analyses of "representative sites," but that data and resources would be insufficient to undertake such a major task.

Although approving of the overall approach of defining the representative sites, the Subcommittee identified two general concerns about the selection process: a possible need to expand the number of reference sites to include extra-territorial sites, and skepticism about the selection of atypical sites to represent specific categories.

3.2.a ***Recommendation***. The Subcommittee noted that there was some ambiguity concerning the "universe" of sites to which the proposed regulations would apply. Specifically, the TSD should state whether the regulations will apply to Territories, trusteeships, and foreign bases of the U.S., as well as to cleanups on U.S. and foreign soils from accidents or incidents of U.S. Government responsibility (e.g., see Mandelker, 1990). To the extent that the regulations will apply to such cases, additional information is needed on the extent of radiological contamination at U.S. Government-controlled sites outside the fifty States. The volume and activity of the radionuclide source term can also be significantly impacted depending upon whether or not formerly decontaminated sites are "grandfathered" from the proposed cleanup standard.

3.2.b ***Recommendation***. The Subcommittee was skeptical of the appropriateness of using the Oak Ridge Reservation to represent the five major DOE facilities involved with diversified weapons research and development activities (Reference

Site VI), given that the total volume of contaminated soil in this category is dominated by the Los Alamos National Laboratory (LANL) site insofar at it alone accounts for 93% of the total volume. In fact, according to the DOE Integrated Data Base (IDB), LANL is second only to the Nevada Test Site in terms of total volume of contaminated soil among all DOE facilities and National Laboratories (TSD Table 1-3, p. 1-16). In general, additional discussion is needed in the TSD to justify the use of selected reference sites to represent other sites in their categories that have different types of radionuclides, and different hydrological, geological and meteorological conditions.

3.3 Completeness of Source Term Information for Reference Sites

Information regarding the types, concentrations, speciation and distributions of radionuclides in soils appears to be incomplete for many of the reference sites (see Appendices A and B of this report). Some specific areas of concern are as follows:

a) A significant weakness in the reference site analysis is the heavy dependence on aerial survey data in the projections of radionuclide concentrations in soil and depth of contamination. The aerial survey data are certainly valuable, particularly for sites where deposition is the major pathway for contamination. The estimates of concentrations and volumes of contaminated soil are dependent on assumptions about the depth and extent of contamination.

b) Consideration of subsurface soil contamination at reference sites is important in assessing cost and projected number of fatalities averted due to clean-up activities. For many of the reference sites, the contamination is assumed to be near the surface, in the top 5 to 15 cm. This assumption is reasonable for sites where the majority of the contamination is due to windblown materials but is not appropriate for sites such as Hanford where the contamination in some areas is at depths greater than 15 cm.

c) The analysis of several of the more complex reference sites such as Hanford included only one radionuclide when, in fact, several radionuclides may contribute to the risk associated with the site. For example, although EPA recognized that the Hanford site included radionuclides other than ^{137}Cs, not all were listed in the analysis (TSD, p. 4-39).

To a large extent, incomplete source-term inventories are understandable and indeed unavoidable, given the extensive data bases to be searched for 16 sites in a brief period, and the limited fraction of data collected to delineate soil cleanup volume compared to the bulk of the data collected for the distinctly different purposes of radiation protection or compliance monitoring at the active sites. It is recognized that a complete summary of environmental radionuclide levels and distributions is not necessary to achieve the purpose of the TSD. Nonetheless, efforts to improve the data set appears warranted for some of the sites, most notably Hanford Reservation, Fernald Environmental Management Project, Savannah River Site, and Oak Ridge Reservation (Reference Sites I, II, V, and VI, respectively).[4] The following actions are suggested to EPA regarding the TSD:

3.3.a ***Recommendation***. A major concern of the Subcommittee is that source term information appears weak even at the most well-defined sites, and ORIA had to be quite inventive at some sites. Recognizing that consistent site-wide data are limited, the Subcommittee commends ORIA for making good use of the available data and for its continuing efforts to work with other agencies to obtain site-related data and to ensure appropriate utilization of the information collected. However, quantitative estimates of the uncertainties for the contaminated soil volumes are still needed. The TSD should utilize more of the information available for the reference sites, including aerial radiological surveys and reports describing the presence of subsurface heterogeneities, by obtaining the cooperation of knowledgeable site staff.[5] For many reference sites, it appears that the radionuclide selections were not sufficiently inclusive, particularly for those cases in which aerial survey data were the primary source for the source term. Most of these sites contain multiple radionuclides, and different combinations may drastically affect the calculated waste volumes (by affecting assumptions about depth of contamination) and allowable concentrations. Specifying the chemical and physical forms of specific radionuclides in the individual reference sites would be desirable not only for predicting leaching rates and transport rates in groundwater (see section 5.6.3 of this report), but also for those cases in which the form significantly affects human intake, as in the case of uranium at Fernald or in munitions at Department of Defense facilities. Some radionuclides may have been removed from consideration because they are judged to be of minor impact, or because insufficient information is available. The TSD should present the criteria for such decisions, with more detailed explanations for specific reference sites

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[4] The Agency's subsequent approach to resolving this issue was described to the Subcommittee in Wolbarst (1995b;c).

[5] Responses by Agency staff indicate that pertinent data bases are being expanded (e.g., Wolbarst, 1995b,c).

3.3.b ***Recommendation.*** The values of remediation area and soil volume at many sites are not explicitly calculated, but are said to be obtained based on overflight maps for the former, and from extrapolation for the latter. More than one value is given for some sites. Clarification of the calculation procedure and resolution of the discrepancies are needed.

3.3.c ***Recommendation.*** The TSD should make clear for each reference site that the available data are adequate only to achieve the purposes of the calculation, i.e., estimating soil volume for remediation and providing a basis for comparing the factor of risk per radionuclide concentration in soil by various scenarios with the generic site risk factors.

3.3.d ***Recommendation***. The TSD should emphasize in the introduction or conclusion that this document would be inappropriate to use as a basis for identifying priority cleanup sites or for comparing with site decontamination and decommissioning program proposals because of the restricted scope of this document, i.e., additional monitoring would be needed for either of these two applications.

3.4 Modeling of Source Term Information for Reference Sites

By creating an artificial reference site, EPA weakens the reliability of their presentation, the reader's confidence in their conclusions, and future uses of the information in the TSD. Concerns are cited in Appendices A and B about creating composite sites and using radionuclide data taken from other sites. Such manipulations introduce the opinions of Agency staff and distance reference site values from reality. Nonetheless, the Subcommittee recognizes the time and data constraints faced by EPA in developing the TSD, and understands the use of the reference site approach as generally providing a fair representation of the source term.

The main concerns among modeling assumptions for the source term are the selected thickness of the radioactively contaminated layer, the assumed uniformity of soil type and hydrology, and the thickness of the contaminated layer at large sites. Specific concerns are as follows:

a) Site-specific heterogeneities which could significantly affect the risk calculations, such as subsurface lenses and non-homogeneous concentrations of radioisotopes, have been ignored at some sites, notably Reference Sites II, IV, XVIII, and XX (based on Fernald, Weldon Spring, Cintichem, and the Apollo plant, respectively).

b) The depth of contamination in soils appeared to be arbitrarily assigned and, in some cases such as Reference Site II, contrary to actual known situations. A heavy reliance on radiological aerial survey results for modeling the horizontal distribution of radionuclides is understandable because of its convenience in providing isopleths to define soil zones of specific concentrations, but can be misleading because only near-surface originating gamma rays are measured. There appears to have been an over-reliance on aerial survey data to define radionuclide distributions for Reference Sites I, III and V, for which existing data from other information sources cast doubt on the TSD assumptions about the radionuclides present, and their aerial and vertical distributions.

c) The TSD specifically states that site characterization was limited to areas where soil was contaminated by spills, leaks, overflow or runoff from waste, or by windblown deposition (TSD, p. 4-3), on the assumption that buried waste would be remediated according to other sets of regulations (TSD, p. 4-3). Consequently, the isopleths need to be refined at many sites to address contaminated buildings, tanks, and waste disposal sites that are excluded from the analysis.

d) While it is recognized that the proposed cleanup standard would apply only to cleanup of soils at sites to be released to unrestricted public use, in many cases buildings and equipment are underlain by contaminated soils. It is unclear how the TSD deals with this aspect of the source term inventory.

Details of these and other concerns of Subcommittee members are provided in Appendices A and B of this report. The following recommendations are suggested for consideration by EPA:

3.4.a ***Recommendation.*** Separate modeling may be warranted for those sites with characteristics related to radionuclides, dimensions, or transport factors that are very different from the reference site intended to represent them. Differences found between the reference site and these individual case studies with respect to risk factor per radionuclide concentration or soil cleanup volume should be reported. (See recommendation 3.2.b)

3**.4.b** ***Recommendation***. Where source-term information for modeling risk factors per soil concentration and soil cleanup volumes is speculative, notably the thickness of contaminated soil and the variability of the radionuclide concentration, attempts should be made to improve the data base. If development of a more defensible source term is not feasible, the TSD should acknowledge this weak data-base and discuss its ramifications to the entire TSD presentation.

3.4.c ***Recommendation***. A frequently repeated statement in the TSD is that a reference site only partially represents the named site, without providing a detailed map that shows waste repositories and structures. The dimensions of the reference site for soil remediation should be specified as meticulously as possible. The possible impact of adjoining areas that contain high- or low-level waste repositories, contaminated structures, or contaminated groundwater should be acknowledged and discussed. Isopleths defining soil of a specific concentration need to be better defined at many sites that include buildings, tanks, and waste-disposal sites. The Agency should make clear that the contaminated soil defined by the radionuclide concentration isopleths for a given reference site is completely destined for soil cleanup, or that a specified fraction will be removed from consideration in its analysis of the reference site. Alternatively, EPA should indicate by estimates of uncertainty, why such boundaries can remain vague without materially affecting the results.

3.5 Development of Distribution Functions for Soil Volume vs. Soil Specific Activity

This section examines how the source term was developed and was used by EPA to estimate the volumes of soil that will need to be remediated. The process used by the Agency is summarized in Figure 3-1 which shows that the source term enters the calculation in two places: a) in the plot of soil volume as a function of the specific activity of the soil, and b) in the description of the contaminated zone used as input for the computerized environmental pathway exposure models. These will be discussed separately.

The development of reference sites in chapter 4 of the TSD has one major goal: to describe soil volume vs. soil specific activity (SSA) distributions or functions that will allow the conversion of an SSA (estimated from risk factors and risk level) into a volume of soil to be remediated. A second goal is to describe how the physical parameters used as input for the risk factor calculations (TSD Table 4-6) were estimated.

The mathematical representations described above were constructed from essentially two types of data: aerial surveys of gamma-emitting radionuclides, and soil sampling. Calculations of contaminated soil volumes assumed a fixed depth for the contaminated zone thickness, in which the radionuclide distribution was assumed to be continuous and uniform. Given the results described in Chapter 7 of the RCSS report, these assumptions mean that the accuracy of the estimated volumes of soil to be remediated is directly proportional to the accuracy with which the contaminated zone thickness is known.

The method used to convert the aerial survey data to a planar area (as opposed to a surface area) was a hand-drawn graphical conversion of the radioactivity isopleths to SSA isopleths followed by manual "pixel" counting. The resolution of the "pixels" of these converted maps is not discussed in the report, nor is the resolution of the aerial surveys. Therefore, the quality, precision, or accuracy of the data used cannot be properly evaluated. Furthermore, it is not possible to evaluate any degradation or loss of information , due to the methods used. It is surprising that the area calculations were done in this way, and that digital imaging techniques were not applied to this problem. The description of the sources of error for these analyses is highly speculative, confusing, and brings out problems that are not necessarily relevant to the discussion at hand. The section on errors (TSD, pp 4-12, 4-13) for the aerial surveys is neither useful nor informative, given that it is impossible to gauge the magnitude of the uncertainties in the data and area conversion technique from the information presented.

Several methods were then used to build the volume vs. SSA curves:

a) curve-fitting of data points with various mathematical functions, followed by pair-wise curve-fitting with the function selected for interpolation;
b) distributions obtained from various sources;
c) creation of a distribution using averaged distribution parameters from other reference sites; and
d) extrapolation to "zero activity above background" using curve fitting from the last pair of data points.

These mathematical formulae, and the methods used to derive them, are not presented in any detail (except for reference site II) in the TSD but were evaluated based on a technical addendum provided by Agency staff to the Subcommittee (Wolbarst, 1995a). The formula selection was done by least-squares two-parameter curve-fitting to linear, logarithmic, exponential, and power functions, followed by calculation of the Pearson product-moment correlation coefficient, and selection of the function with the highest absolute value for the correlation coefficient. These formulae, rather than the graphical representation of the data presented throughout TSD Chapter 4, were used to calculate the volumes of soil to be remediated.

The choice of exponential and power functions seems justified as providing the best fit although the high degree of correlation obtained by the linear regression procedure employed (the data were transformed log-log or log-linear for the curve-fitting) should not be taken to imply any degree of accuracy for the data, but rather as an indication of the precision of the computation (Glantz and Slinker, 1990). In general, the methods used are reasonable and sound, with the exception of the distribution constructed for Reference Site V (which is based on the Savannah River Site). The creation of a distribution for Reference Site V using distribution parameters from another reference site is a guess at best, given the absence of site-specific data.

A critical parameter for the determination of the volume of soil to be remediated, and for the reference site-specific risk factors, is the thickness of the contaminated zone. In most cases, it is assumed to be 5 cm because this is the minimum depth that can be excavated and removed. Other depth values are estimated based on DOE or NRC volumes divided by the total planar area of the site. It is impossible to tell whether the assumed and/or calculated depths overestimate or underestimate the true but unknown values based on the information presented in the report. The Subcommittee also notes that some estimates developed by the NRC (see Appendix A) of soil volumes to be remediated, differ from those in the TSD by as much as two orders of magnitude.

The source term input parameters used for the risk factor calculations are presented in TSD Table 4-6. The uncertainties involved in the areas or contaminated zone thicknesses presented cannot be evaluated. Although the TSD states that the volumes represent a conservative estimate or upper bound, this statement is not justified by the information or data presented. The degree of conservatism claimed for these parameters seems to be more a matter of belief or opinion of the Agency staff, rather than substantiable scientific fact based on the data presented.

The Subcommittee makes the following recommendation concerning errors and uncertainties in the source term definitions:

3.5.a ***Recommendation***. The method used to derive the mathematical expressions utilized to convert soil specific activity to a cleanup volume, and the formulas themselves, should be presented explicitly in a technical Appendix to the TSD.

3.6 Sensitivity and Uncertainty Analysis of Source Term Parameters

The sensitivity analysis in the TSD for the generic site suggests that the factors for risk per soil radionuclide concentration in various exposure scenarios are not highly sensitive to site definition. The aspect of the generic site which leads to the greatest variability of this factor for relatively insoluble radionuclides is the contaminated soil thickness. Only for soluble radionuclides do other site parameters have a significant influence on the risk factor. Indeed, for reference sites II, IV, VI, X, and XIII, where U and ^{99}Tc are dominant (that is, 5 out of 16 sites), the uncertainties in the risk factors for these radionuclides as a result of uncertainties in the transport parameters such as K_d value, are such that either no remediation or remediation of all contaminated soils are possibilities, based upon the site-specific parameters. In these cases, the reference site approach does not capture adequately the real universe of sites.

Including a section on sensitivities and uncertainties of soil cleanup volumes for reference sites is highly beneficial, but the results focus on the impact of varying modeling assumptions, which are only one aspect of the uncertainty. The extent to which the utilized radionuclide concentrations and distributions may differ from actual values, and the extent to which the created reference sites may differ from the median of each category, cause much larger uncertainty.

No evaluation was made of the uncertainty introduced by the assumption that soil removal is the only remediation technique to be used. Possible alternatives are radionuclide removal from the soil, covering or stabilizing (e.g.,vitrifying) soil, or maintaining control as part of a waste repository or elevated-contamination area. This omission is compounded further by the fact that the draft proposed rule explicitly mandates that buildings, aquifers, and surface waters be included in the pathway modeling for the risk calculations, to meet the determined risk level or ensure that the Safe Drinking Water Act Maximum Contaminant Levels (MCL) are met.

A problem of circularity occurs in the uncertainty analysis presented in TSD Section 6.1. Uncertainties in the soil volume vs. activity curves are disregarded due to lack of sufficient information to quantify them. Yet these curves are used to estimate the area and thickness parameters for input to RESRAD to calculate site-specific risk factors. The sensitivity analysis in TSD Chapter 3 shows that the major sources of uncertainty in this calculation are the contaminated zone thickness followed by the contaminated zone area. These calculated risk factors are themselves another source of uncertainty. Finally, these risk factors are converted to volumes of soil to be remediated using those curves whose uncertainties were disregarded in the first place. This discussion requires clarification, and estimates of uncertainty for all parameters need to be presented in a logical and scientifically sound manner.

The following can be concluded:

3.6.a ***Finding.*** The generic site seems to be generally applicable to most cases under study. Exceptions are cases where the "water-pathway dependent radionuclides" are dominant (TSD Table 3-20, p. 3-73). For these cases, accurate site-specific data will be necessary to describe the site and to calculate the site-specific risk factors needed to determine the levels of remediation. In addition, the sensitivity of modeling radionuclide migration into the groundwater to assumptions about mobilization by complexation with non-radioactive materials, such as organic solvents, should be examined (e.g., by varying the K_d parameter).

3.**6.b** ***Finding***. For risk calculations involving insoluble radionuclides, the sensitivity analysis indicates that the uncertainty of the result will be driven by the uncertainty in the thickness of the contaminated zone (although this conclusion may depend upon the dominant pathway for exposure, or on other site characteristics such as mobilization by complexation with non-radioactive species).

3.6.c ***Finding***. The soil volumes that need remediation for a 10^{-4} risk level cleanup may represent an upper bound given the fact that such a cleanup level for the case of such nuclides as ^{226}Ra and ^{232}Th may require achieving soil specific activities that cannot be distinguished from background at some sites.

3.6.d ***Recommendation***. The discussion and presentation of sources of errors and uncertainties, as they pertain to the source term, are limited to an acknowledgment that they exist. It is impossible to gauge their magnitude from the information presented in the TSD. Uncertainties should be estimated for the contaminated soil volumes listed in the DOE Integrated Data Base and for sites under control of Federal Agencies and sites licensed by the NRC or NRC Agreement States. A well-defined and scientifically credible set of uncertainty ranges should be included in TSD Tables 4-5 to 4-8 for all key input parameters used in the risk factor and volume calculations. In addition, nonquantifiable components of the overall uncertainty should be identified and discussed as to their likelihood of exceeding the quantified components of the uncertainty.

3.6.e ***Recommendation***. The TSD analyses should make it obvious to the reader that estimates of excavated soil quantities and associated transportation risks correctly include the application of a bulking factor of 130% applied to in-situ volumes.

3.6.f ***Recommendation.*** The circular argument regarding uncertainties in the source term in TSD Section 6.1 should be clarified, and these uncertainties should be quantified in a reasonable scientifically justifiable manner.

4. SCENARIOS AND PATHWAYS USED IN EPA'S ANALYSIS

4.1 Overview of Approach

In health risk assessments, scenarios are descriptions of the conditions under which persons could be exposed to hazardous materials. In the case of regulatory risk assessments, the scenarios are necessarily speculative because the risks to be reduced by the regulations have not yet been incurred and the future conditions of exposure are in doubt. Faced with this necessity to speculate, three courses of action are possible:

a) attempt to characterize exposures as realistically as possible, taking into account current conditions and the reasonable persistence or predictable change in those conditions;

b) use "conservative" assumptions about the conditions of exposure, i.e., those that will tend to overestimate risk so that regulations based on the assessment will be biased in the direction of the protection of health; or

c) analyze a range of scenarios to characterize the uncertainties and variabilities inherent in assessments of future risks.

In its assessment of the costs and benefits of various proposed cleanup standards for radioactively contaminated sites, EPA has adopted features of all three courses. For example, in its characterization of future occupancy patterns on or near the sites, EPA has postulated the possibility of on-site residential occupancy by persons who also consume some home-grown produce and livestock products, a scenario that tends to maximize individual risk, but it also uses plausible population density estimates for various scenarios, leading toward more reasonable population risk estimates. EPA also uses variants on most of its assumptions to test the sensitivity of the results to the assumptions and presents uncertainty analyses that have both quantitative and qualitative components.

The Subcommittee divided its comments on EPA's scenarios into three sections, as follows:

a) Section 4.2 on *Land Use* discusses the assumptions regarding who will live or work on or near the sites in the future and how they will conduct their lives.

b) Section 4.3 on *Selection of Pathways for Analysis* discusses the assumptions regarding how people on or near the site will become exposed through air, soil, groundwater, surface water, and direct gamma exposure.

c) Section 8.3 on *Time Horizon* discusses the span of years included in the analysis, especially as it relates to the calculation of population risk (number of cancers avoided by implementing the standards). To emphasize that this is more of a policy issue than a technical issue, and hence outside the scope of its charge, the Subcommittee has chosen to present its comments on this issue in a separate chapter of this report.

4.2 Land Use

EPA has chosen to develop a standard that would apply to sites with essentially unrestricted access and therefore has focused on scenarios that feature on-site occupancy for either residential or occupational (commercial-industrial) land use. The residential use scenarios include an option for agricultural use by the residents. Although EPA acknowledges that some sites (or portions of sites) may be managed with various restrictions on access or use, it has left open the question of what standards would apply to such areas. That question may be handled partly through EPA's standards for radioactive waste disposal, which could be used to manage the risks of those restricted access sites by treating them as waste disposal areas. However, the waste disposal regulations may only apply to engineered facilities, not to areas with only access or use restrictions.

Given EPA's decision to assume unlimited access, its overall choice of scenarios is appropriate. If its standards are designed to be protective for unlimited access, then future residential or commercial/industrial use is possible. The actual future use will depend on the suitability of the site for various activities, based on factors other than the past history of radioactive contamination and cleanup. Some sites may be unsuitable for any intensive human use even if residual radioactivity were not an issue. It is also possible that risk managers will decide to place restrictions on access rather than require expensive and potentially ecologically damaging soil removals at some sites. In those cases, the exposure of site neighbors to windblown dust, migrating groundwater or surface water runoff, and "shine" (direct exposure to gamma radiation at a distance) should be considered. While the Subcommittee is not prepared to recommend inclusion of an off-site residential neighbor scenario in EPA's analysis, it suggests that EPA consider adding a qualitative discussion of the influence of such a scenario.

All of the residential land-use scenarios involve food production and consumption. Food production, treatment, and consumption patterns evolve considerably over time. For example, rates of beef and chicken consumption are much different now than they were 30 years ago. The Subcommittee therefore recommends that, in transmitting its findings to the EPA group responsible for proposing the cleanup standard, the technical support group should emphasize the additional uncertainty of its estimates for 1,000 and 10,000 years in comparison with those for 100 years (see also Section 8.4 of this review document).

EPA has not explicitly considered a recreational scenario, in which the land would be used mostly for outdoor activities on an occasional basis. Although such a use would result in lower rates of occupancy (hours per lifetime), it could also introduce new pathways of exposure (e.g., consumption of local wild game, fish, fruit, and mushrooms). It is possible that persons living on one part of a large site might participate in outdoor recreational activities on another part of that site. Whether that practice would increase or decrease exposure relative to a person who spent most of his or her time at home is difficult to predict. While the Subcommittee is not prepared to recommend inclusion of a recreational scenario in EPA's analysis, it suggests that EPA consider adding a qualitative discussion of the influence of such a scenario.

In summary, the Subcommittee makes the following recommendation:

4.2a ***Finding.*** The Subcommittee is satisfied that EPA's choice of an on-site residential scenario and a commercial-industrial scenario for thorough analysis is adequate for estimation of reasonable maximum exposures and risks from sites that have been cleaned up to a specified level of contamination. While other scenarios are also plausible, they are not likely to produce substantially higher estimates of population risks.

4.2.b ***Recommendation***. Although the Subcommittee does not propose adding a recreational scenario to the analysis, it does recommend that EPA consider adding qualitative discussions of the influence of recreational and off-site residential neighbor scenarios on the risk estimates.

4.3 Selection of Pathways for Analysis

4.3.1 Pathways Included in the EPA Analysis

Pathways are mechanisms by which individuals may be exposed to radioactivity from a contaminated site. The list of pathways considered by the EPA in its analysis are the standard generic pathways typically considered for regulatory compliance calculations. EPA's analysis of the reasonable maximum exposure (RME) to future users of a site subject to the cleanup rule includes consideration of the following pathways:

a) direct external radiation from photon-emitting radionuclides in the soil,
b) inhalation of resuspended dust containing radionuclides,
c) inhalation of radon and radon decay products via diffusion into buildings or the atmosphere from radium-containing soils,
d) ingestion of groundwater to which radionuclides have migrated from soil
e) ingestion of soil containing radionuclides,
f) ingestion of produce grown in soil containing radionuclides,
g) ingestion of beef from cattle raised on-site,
h) ingestion of milk from milk cows raised on-site, and
i) ingestion of fish from surface waters that receive runoff from soils containing radionuclides.

Only a subset of the pathways is used for some future occupancy assumptions.

EPA's analysis of cumulative population risks (cancer fatalities avoided by reducing RME risk from 10^{-2} to the target risk level) considers only the following pathways:

a) direct external radiation from photon-emitting radionuclides in the soil,
b) inhalation of resuspended dust containing radionuclides,
c) inhalation of radon and radon decay products via diffusion into buildings or the atmosphere from radium-containing soils,
d) ingestion of groundwater to which radionuclides have migrated from soil, and
e) ingestion of produce grown in soil containing radionuclides.

In terms of risk assessments to the potentially exposed individuals and populations, the parameter values selected for each of these pathways must be scrutinized and evaluated. The Subcommittee recommends that this evaluation should focus on the following issues:

a) Is the purpose of the risk calculations to derive screening or bounding estimates, or to provide the best estimate of the true value (risk)? The objective then should guide the selection of appropriate parameter values used in pathway modeling.

b) To what extent are the parameter values selected for the reference pathways sufficiently conservative for the real pathways that could occur at real sites?

Additional comments on these aspects of the pathway modeling are provided in Section 5 of this report.

4.3.2 Pathways Omitted from the EPA Analysis

Provided one accepts EPA's choice to focus on a population at risk that would occupy the site after cleanup, the list of pathways considered for the RME risk is reasonably complete and probably includes the pathways of most importance from the perspective of cancer risk. EPA provides some explanation of its decision to exclude such pathways as dermal exposure to soil containing radionuclides (i.e., the relatively low dermal absorption of radionuclide compounds in comparison with other pathways, which is not always true for lipophilic compounds) and volatilization of radionuclides from soil or domestic water (which is probably minor for all radionuclides except radon, tritiated water vapor, and carbon-14 dioxide, each of which is treated specially in the analysis).

One omission that is potentially important is the possibility of ingesting surface water that receives runoff from soils containing radionuclides. This omission is curious because the fish pathway is included (by assuming that groundwater infiltrates into surface water) yet drinking water from surface sources is not. EPA dismisses this pathway by asserting that concentrations in groundwater would be higher than those in surface water at virtually any site (with the modeling assumptions used) and that "human ingestion of surface water is not considered a reasonable scenario in most instances." However, given the wide use of surface water for drinking water supplies and the better quality of surface water relative to groundwater in some environments, these arguments are not particularly strong. Moreover, radionuclides deposited on soils can move to surface water not only by dissolution into runoff water but also by overland transport of particles eroded from the soil. The Chernobyl experience shows that contamination of surface-water reservoirs occurs mainly by water runoff from the soil surface and not by groundwater infiltration (e.g., see discussion in section 5.7 of this report); hence, it is important not to neglect this pathway.

Groundwater and surface water pathways may also be modified by diversion of a portion of such waters and their associated sediment loads (in the case of surface waters) into combined sewer and storm sewer systems, the effect of which has not been addressed in the TSD analysis.

Another omission that may be important for specific sites is the recreational scenario and related pathways. None of the modeled scenarios contain exposure pathways due to recreational activities. People living in a residential or agricultural area (farms) will have recreational activities in that area, such as fishing in a nearby lake, hiking, and hunting. Additional exposure pathways such as consumption of wild game, mushrooms, wild fruit, and locally caught fish, should be discussed and qualitatively considered in any site-specific assessment as appropriate.

With respect to the lack of concurrence between the list of pathways used for the population impact analysis and the RME risk assessment, EPA states that the pathways omitted from the latter do not contribute substantially to total cancer fatalities, with the possible exception of soil ingestion for nuclides such as ^{239}Pu that do not involve external gamma radiation or radon and do not migrate rapidly to groundwater or crops. Given that other radiological risk assessments have yielded similar conclusions (NRC 1994a; DOE 1994a), this focus is probably justified, at least for the sites most important to the overall impact analysis.

Because the on-site occupancy scenarios will almost always produce higher RME risks than a scenario in which off-site exposures are considered, the lack of off-site transport scenarios in the RME risk analysis of reference sites is probably justified. In future assessments of real sites, however, the following possibilities should be considered:

a) groundwater may not be used for drinking water on-site but affected aquifers may be utilized for off-site populations; and

b) agricultural use of the site may not be contemplated, but dust transported downwind could affect off-site agriculture, which might supply food for on-site residents.

For the population risk analysis, EPA asserts that its methods for estimating cancer fatalities have features that compensate for the distribution of impacts on-site and off-site. In particular, EPA uses the radionuclide inventory as its principal source term for the population impact analysis and does not assume that the groundwater containing radionuclides would be consumed only by on-site occupants. For example, EPA assumes that one half of all the groundwater originating at the site will be withdrawn for use and that 1% of that will be ingested as drinking water. With this assumption and the linear relationship between intake and risk, the number of fatal cancers averted would not be affected by the fraction of the water used by on-site versus off-site populations (except for decay in transit to off-site populations). Similarly, dust containing radionuclides could be inhaled by either on-site or off-site populations and identical cancer burdens would be calculated if the total amount of dust inhaled is assumed to be constant. Moreover, EPA should provide the rationale for its assumptions regarding groundwater withdrawal; the assumption that 50% of the groundwater will be withdrawn for residential use may be reasonable but is difficult to support.

4.3.3 Duration of Exposure

In every case, the person at risk is assumed to be exposed to soil with uniform concentrations of radionuclides for a defined period and duration of exposure, and not exposed at all for other times. Real sites will be characterized by non-uniform distributions of radionuclides. The lack of uniformity in some cases could lead to significantly erroneous risk calculations if the proper statistic for the distribution of concentrations is not used. While the average concentration over an area may be satisfactory for many pathways, it may not be if a real person's mobility were substantially greater than the scale of disuniformity. Generally but not always, more accurate consideration of disuniformity and mobility would lead to lower calculated risk levels for the reasonable maximum exposure. On the other hand, on large sites the time spent "away from home" may be spent at other locations on site, which could lead to residual risks larger than assessed. The net effect of these two simplifications cannot be predicted without knowing more about the true distributions of radionuclides.

4.3.4 Findings and Recommendations Concerning Pathways

4.3.a ***Recommendation***. In the face of constraints of time and resources for revisiting the definitions of the pathways, EPA should focus its efforts for any improvement of the pathway definitions (i.e., underlying assumptions and adopted parameter values) on the dominant pathways, particularly external gamma radiation, radon inhalation, crop ingestion and ingestion of groundwater (especially the population risk assumptions). Ingestion of surface water should be investigated further to determine whether this pathway might be dominant under any widely prevalent conditions.

4.3.b ***Finding.*** With the exception of a surface water runoff and erosion pathway with subsequent potential for drinking water and fish consumption exposures, EPA has generally included in the RESRAD analysis all the pathways that are likely to be important for radionuclides in inorganic forms. (Direct and indirect dermal absorption pathways might be important for organic chemicals and mixed wastes.)

4.3.c ***Finding.*** In assessing compliance with the proposed standard at real sites, consideration should be given to the non-uniformity of soil concentrations, even after site cleanup, as they might affect the risk to persons who do not remain at the same geographical point continuously.

4.3.d ***Recommendation***. ORIA needs to identify the purpose of the risk calculations as being either the derivation of screening or bounding estimates, or the provision of the best estimate of the true value (risk ?). The chosen objective should then guide the selection of the appropriate parameter values to be used in pathway modeling.

4.3.e ***Recommendation***. ORIA should determine if the parameter values selected for the reference pathways are sufficiently conservative for the real pathways that could occur at real sites.

4.3.f ***Recommendation.*** In future assessments of real sites, ORIA should take into account the possibility that groundwater on-site may not be used as a source of drinking water, but that affected aquifers may be utilized by off-site populations.

4.3.g ***Recommendation.*** Although agricultural use of a site may not be contemplated in the current scenario future analyses should consider the possibility that dust, transported downwind, could affect off-site agriculture which might supply food for on-site residents.

5. METHODOLOGIES FOR MODELING TRANSPORT TO PEOPLE

Section 5.1 reviews the Agency's rationale for selecting RESRAD for modeling exposure to on-site individuals, and a a population model developed by EPA for the TSD for modeling exposure to populations. Sections 5.2 through 5.7 provide Subcommittee comments on details of the model formulations used as part of these codes to simulate transport by the selected pathways, including the choice of variables to be included and appropriateness of selected parameter values that were used by EPA.

5.1 Evaluation of Candidate Models for Pathway Modeling

5.1.1 Model Selection Criteria and Process

Once the desired exposure scenarios and pathways were determined for the assessment, EPA sought to identify a set of candidate models for predicting exposure and risk, and to evaluate which model or models were most appropriate for the task. The models considered needed to be multimedia, considering the full range of direct, soil, air, surface water, ground water and food ingestion pathways. In particular, the models needed to address the following exposure pathways:

a) external radiation exposure;
b) soil ingestion;
c) plant, meat and milk ingestion;
d) inhalation of volatiles and fugitive dusts;
e) migration of radionuclides to ground water; and
f) ingestion of contaminated drinking water.

In addition, it was desired that the models be capable of assessing the exposure and risk from radon.

A set of 21 potential models was identified by EPA through review of scientific literature, EPA reports and databases, and discussions with project staff. The models were evaluated according to their ability to address the multiple exposure pathways identified above, and based on the following model performance criteria:

a) level of validation and peer-review, outside and within EPA;
b) availability and accessibility of the computer code;
c) extent to which the computer code was user friendly; and
d) the amount of site data required to implement the model.

Based on this evaluation, the DOE RESRAD code was chosen for the modeling of individual risks at reference radiation sites, and two additional codes, EPA's PRESTO-CPG and a modified version of EPA's RAGS/HHEM Part B, were chosen for further comparative studies. The RESRAD code covers all exposure pathways of interest, has been subject to validation and peer review studies, has an available and user-friendly computer code with a moderate amount of required site data, and is being used by DOE for site evaluation studies. The PRESTO-EPA-CPG model has similar capabilities (although it does not compute exposure and risk due to radon) and has undergone previous review by the EPA SAB for use in the evaluation of low-level radioactive waste disposal rules (Federal Guidance Report No. 11, (EPA, 1988)) (EPA, 1987). The RAGS/HHEM Part B model is a simple, conservative screening model based on unit transport and exposure factors. While it is not computerized, and has not undergone validation and peer review by the SAB, it is believed to provide a conservative baseline for comparison of the more realistic process models. The Subcommittee finds that the initial screening and selection of candidate models for further evaluation as reported in the TSD was conducted in a reasonable, sound and thorough manner, using appropriate criteria for model selection.

The comparative evaluations of RESRAD, PRESTO and RAGS/HHEM included a detailed comparison of model formulations and assumptions for the various exposure pathways, and application of each to a generic site for calculation of unit risk factors for the radionuclides of interest. The latter comparisons indicated significant differences between the models for many of the radionuclides. Differences were identified both in the overall unit risks and in the allocation of this risk to the different exposure pathways. A number of reasons for these differences were identified; principal among these are the different treatment of source-term depletion and progeny ingrowth by the models, and differences in the methods for computing leachate flows and concentrations to the ground water. The RAGS/HHEM model assumes an infinite, nondepleting source, and PRESTO is limited in its treatment of radioactive ingrowth when performing source calculations for a series of ground water sources. RESRAD is more conservative than PRESTO in assigning radionuclide concentrations to leachate; the former assumes equilibrium while the latter considers the contact time between the infiltrating water and the waste.

As a result of these differences, when the models are applied to a generic site at which the ground water exposure pathway is important, RAGS/HHEM estimates the highest risk for a given waste concentration, PRESTO the lowest, and RESRAD an intermediate value, although sometimes much more conservative than PRESTO. Differences between the models are greatest for radionuclides with low K_d values (low adsorption onto soils) and short half lives, such as tritium; the models provide similar estimates for highly adsorbing, slowly decaying radionuclides. However, significant differences in unit risk are present even for some radionuclides with very long half-lives, such as ^{232}Th. Other differences in model predictions are ascribed to process and exposure assumptions such as inclusion of erosion for surface waters (included in PRESTO, but not in RESRAD) and corrections for time spent indoors (included in RESRAD, but not in the other models).

The significant differences in the predicted unit risk for the three models evaluated are somewhat disturbing, though not surprising. Similar, order-of-magnitude differences between RESRAD and six other radiation exposure pathway models were found in a benchmarking study conducted by Argonne National Laboratory (Faillace *et al*., 1994). These arise from fundamental differences in pathway representations and assumptions, as well as possible errors in input databases. Presentation and acknowledgment of such differences for selected radionuclides provides an honest appraisal of the current state-of-the-art for radiation exposure modeling.

5.1.2 Comments on Selection of Models and Their General Limitations

RESRAD and the other models considered by the Agency are reflective of the state-of-the-art for comprehensive integrated multimedia/multipathway models for exposure and risk. These models include simplified modules for many of the pathways and exposure scenarios they consider. In some cases, these modules are similar to models that could be used in site-specific applications focusing on a specific pathway. In other cases, they are more idealized and simplified.

RESRAD is particularly well-suited for the application to which it is applied by EPA—determining on-site risks from residual radionuclides in soils—because this is the sole focus of the model. The Agency summarizes its basis for selection of RESRAD for the individual unit risk calculations as follows (TSD, bottom of p 3-29):

> "The results calculated by the three models are similar for many of the radionuclides. The most significant changes are caused by the decay and ingrowth corrections, which at this time only RESRAD applies to all of the radionuclides. The most significant change for a single radionuclide is H-3, and this is caused by a combination of slow leach rate and short half-life for the PRESTO calculation. RESRAD was selected for performing the calculations for the reference sites because it calculates a more conservative result for H-3 and includes corrections for ingrowth and decay of principle (sic) radionuclides."

While the Subcommittee believes the reasons for selecting RESRAD are somewhat more extensive than those described above (including its use by DOE, user friendliness, and full coverage of on-site exposure pathways), the comparative evaluation appears to result in a reasonable and appropriate selection of the RESRAD model for use at the present time. However, because of the highly simplified and idealized conceptual models for the specific pathways, many of the important processes which can affect radionuclide fate and transport are neglected. The generic processes included in these codes may not always represent the most important processes that will dominate the actual dose and risk to real individuals exposed at a specific location (e.g., see discussion of Chernobyl studies in section 5.3.3 and 5.7 of this report). Even in the case when all real processes are included, the generic (default) parameter values might not be appropriate for a real site. Consequently, RESRAD, as configured for the TSD analyses, is appropriate only to provide an approximate generic assessment for highly aggregated studies in which results from several individual studies are pooled together and in which it is understood that only the results *taken as a whole* are intended to represent the universe of real sites. Thus, RESRAD may not be appropriate for analyzing a specific site's compliance with the standard for complex site conditions. Furthermore, from a cursory review, several of the transport model parameters appear to be internally inconsistent, or inconsistent with literature values or with known site characteristics (e.g., see section 5.6); hence, the Subcommittee strongly urges EPA to obtain a thorough peer review of the default and site-specific parameters used in the transport modeling. As more information becomes available through site applications and studies, the Agency should revisit this issue to determine whether RESRAD should be modified or replaced so as to ensure that the appropriate balance is maintained between realistic prediction and reasonably conservative protection of public health.

RESRAD is user-friendly and quite flexible for input data. However, the current version of this model is limited by the absence of a transport module to on-site analysis; other models may have to be used to determine risk at off-site locations. In some cases, the calculated on-site risk may not be significantly different from the risk at an off-site location if, for example, the half-life of the radionuclide is very long, the location under consideration is not very far from the site, and/or the transport process is not very rapid. The difference in risks can be very significant for other cases. Because the Agency does not rule out "institutional control" (i.e., imposition of "restricted access" to a contaminated site) as a realistic remediation measure for some sites, the need for the model to be able to assess off-site risk becomes obvious. It is unlikely that people who live near, but not on, the soil with residual radioactivity would be more at risk than equivalent on-site residents, but the ability to calculate off-site risks for such a residential neighbor scenario would allow the analysis of a strategy where affected soils might be allowed to have higher concentrations if restricted from residential use. For example, a site with commercial / industrial use might still pose more risk to residential neighbors than to on-site workers. The Subcommittee recommends

that EPA supplement RESRAD to include the capability for calculation of off-site risks, if the Agency chooses to continue using this model for this task.

The basis for the selection of a methodology for the population dose model was not discussed extensively in the TSD text. The population model used for the TSD analyses, (hereafter called the CU-POP model, for "cleanup population" model, as requested by the Agency), was developed by the Agency and is based loosely on the RAGS/HHEM model in that it employs simple bounding algorithms and assumptions. However, the CU-POP model differs fundamentally from RAGS/HHEM in a number of respects. It explicitly accounts for radioactive decay, progeny ingrowth, and source depletion by leaching of the soil as a function of time, as well as decay and ingrowth during transport through the unsaturated zone; in contrast, RAGS/HHEM assumes a non-depleting and non-decaying source term. Appendix C of the TSD describes RAGS/HHEM, and Appendix E describes the CU-POP model. The TSD does not discuss the advantages and limitations of alternative methodologies for calculating population effects.

5.1.3 Findings and Recommendations Concerning Model Selection

5.1.a ***Finding.*** The Subcommittee finds that the initial screening and selection of candidate models for further evaluation as reported in the TSD was conducted in a reasonable, sound and thorough manner, using appropriate criteria for model selection.

5.1.b ***Finding***. The Subcommittee concurs with the Agency's decision in its selection of RESRAD as a reasonable transport model for use at the current time. However, while it incorporates some conservative assumptions in its formulations, RESRAD itself may not necessarily provide conservative risk estimates if inappropriate parameter values are selected for the modeling input.

5.1.c ***Recommendation.*** Because the Subcommittee has not evaluated the default values in the RESRAD code, nor the full parameter set used for each reference site, it is unable to fully assess the extent to which the model results can be considered to be conservative or bounding estimates of the true health effects associated with each level of cleanup. From a cursory review, several of the transport model parameters appear to be inappropriate (e.g., see section 5.6 of this report); hence, the Subcommittee strongly urges EPA to obtain a thorough peer review of the default and site-specific parameters used in the transport modeling.

5.1.d ***Recommendation.*** Recognizing the generic and idealized nature of the RESRAD model and its limitation to on-site exposure scenarios, the code may not be appropriate to use in many of the site-specific applications where specific processes or exposure scenarios not included in the model are present and important. The Agency should make readers aware that, for these site-specific applications, there should not be an attempt to force sites to fit the model (as there may be a tendency to do simply because RESRAD is the model upon which the national regulations are based); rather, the Agency should emphasize that models and estimation procedures appropriate for site conditions are needed.

5.1.e ***Recommendation.*** The Subcommittee believes that the population dose model developed by the Agency for the TSD is generally adequate but could be improved through further consideration of off-site risks and the use of better site-specific transport parameters.

5.1.f ***Recommendation.*** The Agency should provide a more extensive summary of the validation and model comparison studies that have been conducted for RESRAD. These studies may suggest that very significant differences often occur between model predictions and observed data, or between the predictions of alternative models, when they are applied to specific sites or contamination incidences. This, however, provides an honest appraisal of the state-of-the-art of such modeling and a good sense of the significant uncertainty associated with the application of general models to specific cases with varying conditions and highly uncertain input parameters.

5.1.g ***Recommendation***. As more information becomes available through site applications and studies, the Agency should revisit this issue to determine whether RESRAD should be modified or replaced so as to ensure that the appropriate balance is maintained between realistic prediction and reasonably conservative protection of public health.

5.2 Pathway Models for Direct External Exposure

Models, assumptions, parameter values, and results of model runs related to direct gamma radiation exposure were reviewed. In general, direct gamma radiation doses and risks were appropriately determined for both the Reasonable Maximum Exposure (RME) for individuals and the Population Exposures. Problems inherent in the application of the results of the generic site assessment involving simple, convenient contamination patterns to real

sites with complex contamination profiles are not discussed in this subcommittee review report.

5.2.1 Modeling Risks to Individuals

5.2.1.1 Exposure correction factors

The three principal exposure scenarios for the Reasonable Maximum Exposure (RME) to individuals include a direct external gamma exposure pathway in which assumptions regarding exposure duration, time and shielding factors differ for each scenario. Two of the three pathway models (RESRAD and RAGS/HHEM) include correction factors for shielding for indoor exposure vs. outdoor exposure as well as for exposure time, frequency and duration. The third model (PRESTO) corrects only for exposure duration. RESRAD also takes into account other factors which affect external exposure from contaminated soil such as radioactive decay and progeny ingrowth, geometry of the contaminated soil, depletion of the contaminated soil by environmental processes and shielding by clean cover. RAGS/HHEM does not account for these processes; PRESTO corrects for depletion, geometry and decay (as well as ingrowth in the version used). Therefore, among these three models, RESRAD is the most realistic in its mathematical formulation of this pathway.

One issue regards how the shielding factor for indoor exposure should be applied. In the case of deposition of radioactive materials around a structure after it has been built, much of the "shielding" effect comes from the simple fact that a person in the middle of a house is several meters away from any radioactive materials, independent of any attenuation of radiation by building materials. If a one-story house is built on undisturbed soil that has already been affected by radionuclides, then the distance effect largely disappears and is replaced by whatever attenuation is provided by any slab or flooring. The shielding factor may thus depend on the depth of soil removed in constructing the foundation and basement, if any, and the depth of radioactive concentration assumed.

5.2.1.2 Dose conversion factors and slope factors

All three codes use EPA-approved dose conversion factors (DCFs) taken from Federal Guidance Report No. 12 (EPA, 1993a). The TSD is very confusing with regard to the slope factors used to estimate risks. Several versions of the Health Effects Assessment Summary (HEAS) Tables are referenced, but these references are apparently incorrect. In fact, the values used and tabulated in TSD Appendix Table B-1 are from a personal communication with Jerry Puskin of EPA/ORIA. This creates confusion. The source and derivation of slope factors should be clarified.

Because the dose conversion factors and slope factors used in the models are EPA-approved values, they were not reviewed in depth. However, Federal Guidance Report No. 12 (EPA, 1993a) values were similar to the DOE values used in RESRAD. Selected slope factors from the 1993 HEAS Tables (EPA, 1993b) were compared to the slope factors in Federal Guidance Report No. 12 (EPA, 1993a), and were found to be similar after appropriate conversion factors were applied. As noted in the TSD, the uncertainty in the slope factors for external gamma radiation supports a lower bound of zero and an upper bound of approximately three times the nominal value.

All three codes use the same slope and dose conversion factors, and thus the differences in results should be due only to the way the models treat the other factors. For the generic site, under the rural/residential scenario, the calculated unit risk estimates for most nuclides for which external exposure accounts for 95% or more of the total risk show little difference among the models. This indicates that the processes neglected by PRESTO and RAGS/HHEM are not significant for the generic site. However, Table 3-10 is somewhat confusing in this regard. For example, the risk factors (presumably in risk per pCi/g although units are not given in the table) for ^{57}Co for RESRAD, PRESTO, and RAGS/HHEM are 6.25 x 10^{-6}, 2.44 x10^{-6}, and 6.40 x 10^{-6} respectively. External gamma radiation accounts for 98% of the dose from ^{57}Co which has a half-life of 270 days. Because both RESRAD and PRESTO account for decay, the risk factors would be expected to be similar. According to the technical document, RAGS/HHEM does not correct for decay; therefore, the risk factor would be expected to be higher. This is obviously not the case. The risk factors calculated by RAGS/HHEM and RESRAD are similar, and the risk factor calculated by PRESTO is lower by a factor of 2.5. The comment in the table notes that the risk factor calculated by PRESTO is "higher" (sic, table entry should read "lower"; this erroneous wording is common throughout Table 3-10) due to the short half-life of ^{57}Co. In this case, the lower risk factor is due to the fact that PRESTO does not calculate the risk at the beginning of the simulation but only at the end of the first year. This would result in underestimating the risk because only 40% of the original ^{57}Co remains after the first year.

5.2.1.3 Generic site assumptions and parameter values

The principal assumptions for the generic site which affect direct external gamma radiation include a homogeneous, infinite depth of contamination and 10,000 m^2 area.[6] These are obviously base case values and are not applicable for most contaminated sites. For most real sites, the depth of contamination would be variable and the concentrations not homogeneous. RESRAD has the capability of taking into account variability in depth and areal extent of contamination and thus can be used to derive more realistic risk estimates. RAGS/HHEM does not have this flexibility, making it less useful for analysis of real situations.

The assumptions and parameter values for the rural/residential and commercial/industrial scenarios which affect the dose from direct gamma radiation are reasonable and are, presumably,

[6] The generic site is described as uniformly contaminated to a depth of 2 m. For the purpose of estimating direct gamma dose, this depth is equivalent to “infinite depth” as used in Federal Guidance Report No. 12 (EPA, 1993a).

incorporated into the RESRAD model used in the analysis. The RESRAD equations for correcting for depth of contamination, cover thickness and source area are also reasonable.

The TSD apparently has not considered the major impacts on direct radiation exposures by the site uses assumed in the exposure scenarios. Excavation for construction may move the contaminated soil layer, paving and buildings will attenuate gamma rays, and plowing may either increase or decrease the gamma-ray flux into air. The TSD should call attention to the site-specific nature of direct radiation exposure for each scenario.

5.2.1.4 Sensitivity and uncertainty analyses

The factors with significant impact on direct gamma radiation exposure which were considered in the RESRAD sensitivity analysis include the area of the contaminated zone and the thickness of the contaminated zone. Infiltration rate, which was also considered in the sensitivity analysis, affects depletion of the contaminated zone which also impacts direct gamma radiation exposure. Other site or scenario factors affecting direct gamma radiation risk such as duration of exposure were not considered. As would be expected, for nuclides with greater than 95% of the risk due to direct gamma exposure, the area of the contamination affected the risk by less than a factor of two. The contaminated zone thickness affected the risk by up to a factor of five for a 2 cm thickness or less than two for a thickness of 10 cm.

The uncertainty analysis for the RAGS/HHEM model considered a greater number of variables using distribution functions rather than discrete values. However, from the output, it was not obvious which parameters had the greatest influence on the final results.

Additional concerns relating to sensitivity results are found in section 7 of this report.

5.2.2 Population Exposures

Population exposure estimates for all sites were determined using the CU-POP model, which assumes that the population impact is proportional to the total radioactivity at the site.

The equations for calculating dose due to direct gamma radiation are based on the soil concentration, not total activity. The equations given in the appendix of the document include a correction for depth of contamination and depletion due to physical decay and leaching but not for ingrowth of progeny. The tables in the TSD indicate that, at least in the case of ^{230}Th and ^{232}Th where the principal health effect is due to the progeny, ingrowth is included.

The equations and assumptions for risk due to direct radiation exposure are reasonable and contain the appropriate corrections for depth. However, the implication in the TSD, with regard to population dose, is that the total inventory of the radionuclide under consideration at the site is the basis for the population dose calculations (TSD, p. 5-11). In addition, the TSD states that the direct gamma population dose calculation is based on the dose conversion factors from Federal Guidance Report No. 12 for an "infinitely thick" contamination depth (TSD p. 2-52). In fact, while this assumption was appropriately applied in the generic site calculation, the population doses for the reference sites were calculated using dose conversion factors from Federal Guidance Report No. 12 which more accurately reflected the assumed depth of contamination for each specific site. This is a reasonable approach to the calculation, but the method used is not obvious from the information presented in the TSD.

5.2.3 Cleanup Worker Exposures

The estimated direct radiation doses to workers for the reference sites are based on the volume of material to be handled and the radionuclide concentration. This is a reasonable approach.

5.2.4 Off-Site Doses Due to Cleanup Activities

Because direct radiation exposure decreases rapidly with distance from the source, off-site doses due to cleanup activities would be confined to individuals in proximity to transport vehicles such as rail cars. The total amount of time any individual or population would be exposed under these conditions is very small, and hence this pathway is not significant, as noted in the TSD.

5.2.5 Findings/Recommendations Concerning the External Radiation Pathway Model

5.2.a ***Finding.*** The external radiation pathway is relatively simple to define. With the possible exception of exposure time (see section 4.3.3), the EPA's modeling of this pathway is generally acceptable. Of the three codes considered in detail in the generic base case study, RESRAD is the most realistic and flexible in its capabilities for accounting for the processes that govern exposure of individuals by this pathway. The assumptions and parameter values for the rural/residential and commercial/industrial scenarios which affect the dose from direct gamma radiation are reasonable and are, presumably, incorporated into the RESRAD model used in the analysis. The RESRAD equations for correcting for depth of contamination, cover thickness and source area are also reasonable.

Areas in which the EPA could improve its analysis and presentation of results are as follows:

5.2.b ***Recommendation***. EPA be more explicit regarding its assumptions about the shielding factor for indoor exposures and be sure that the parameter value used is consistent with those assumptions. The assumed shielding factor for indoor exposure to external gamma radiation does not apply to buildings which have no contamination underneath the structure.

5.2.c ***Recommendation.*** The sensitivity analysis for contaminated zone thickness showed a significant anomaly which should be further investigated. The ratio of calculated soil concentration to base case soil concentration at a 1×10^{-4} risk level, as a function of contaminated zone thickness, shows a value two orders of magnitude higher for ^{230}Th than for any other nuclide. The reasons for such a drastic difference in risk for the same total radioactivity concentration are not obvious and should be explained.

5.2.d ***Finding.*** The uncertainty analysis for the RAGS/HHEM model considered a greater number of variables using distribution functions rather than discrete values. However, from the output, it was not obvious which parameters had the greatest influence on the final results.

5.3 Pathway Models for Ingestion of Soil and Food

5.3.1 Overview Comments

The produce ingestion pathway is somewhat simplified with respect to other multipathway risk assessment models currently in use. It employs two generic groups of produce: fruits, vegetables, and grain are lumped together while leafy vegetables are treated differently. In other models, root vegetables may be treated separately, or a list of specific food crops may be analyzed. The EPA document states that certain corrections were applied for fruits and non-leafy vegetables, but does not make it clear whether that correction was applied to grains as well. The EPA pathway assumptions share with many other systems the need to make many simplifying assumptions for very complex phenomena. The most critical issues, however, probably have more to do with the details of the occupancy scenarios as discussed above. In particular, the amount of locally grown foods as a percentage of the total diet will be highly variable from site to site, from person to person for a given site, and among different specific commodities. This variability has generally been handled by making conservative assumptions, which may lead to substantial overestimates of population risk via this pathway and perhaps to overestimates of RME risks.

The meat and milk pathways are similar in strengths and weaknesses to the produce pathways. As with many other multipathway models, meat is represented by beef. Some multipathway models attempt to consider chicken and eggs. Again, this pathway is unlikely to be important for many sites and, if so, any conservatism is correspondingly less important. It is possible that omission of poultry-related pathways could underestimate risks, although it seems unlikely that including them would move the livestock pathway into prominence. Note that dietary patterns have changed remarkably in the last 30 years and projections of food pathway risks even to 100 years should be viewed with skepticism.

The fish ingestion pathway is based on transfer factors from surface water to finfish and shellfish, after applying a groundwater to surface water seepage model to estimate concentrations of radionuclides in water from soil sources. A significant shortcoming of this pathway is the omission of a runoff pathway, which will probably be more important than the seepage pathway for any radionuclide that binds to soil yet can be liberated to the food chain from sediments. The runoff pathway is clearly more important than the seepage pathway for certain pesticides and highly lipophilic organic chemicals (Kellogg *et al*., 1992); EPA should probably conduct a sensitivity analysis to determine whether any radionuclides behave similarly. However, the overall importance of this omission will likely also be small because of the negligible contribution of the fish ingestion pathway in the current risk assessment.

In the sections that follow, comments are provided on specific parameter values used in the EPA analysis of the soil and food ingestion pathway.

5.3.2 Soil Ingestion Rates

Table 2-1 of the TSD gives the standard default parameter values used for each land-use scenario. The soil ingestion rate of 50 mg/day for commercial/industrial usage of the land is suitable only for office industrial activities. Other industrial activities (e.g., construction, surface mining) may require outside work or can produce high rates of resuspension of surface soil in the atmosphere, potentially leading to more deposition on skin and food with the opportunity for greater rates of soil ingestion. Therefore, a higher rate of soil ingestion is recommended as a more appropriate default value for this scenario. The exact value to be used, however, depends upon the overall intent or objective of the proposed annual dose limit as it applies to cleanup (see recommendation 6.6.a).

For children and adults affected with *pica*, the soil ingestion rate may be as high as 1 to 5 g/day. The highest values to be found in the literature are at 10 g/day. These are extreme values, which are not reasonable default values for the RME individual. Also, the population average soil ingestion rate is inappropriate for the RME individual who is defined as a maximally exposed individual. The EPA standard soil ingestion rate assumed for children is 200 mg/day.

5.3.3 Soil-to-Plant Transfer Factors for Nutrient-Poor Soils

The Savannah River Site is an example of a site with nutrient-poor soils that exhibit very high soil-to-plant transfer rates for ^{137}Cs and ^{90}Sr (Whicker and Hinton, 1993). Therefore, plant contamination will be unusually high at this site and the ingestion pathway is expected to dominate over other pathways of exposure. Chapter 4 of the TSD describes Reference Site I, based on a description of Hanford Reservation contamination, and Reference Site V, based on contamination reported for Savannah River Site. Both of these sites have ^{137}Cs as a major contaminant. The difference between these two sites, as stated in the reference site descriptions, is found mainly in the fractional distribution of ^{137}Cs in soil. No consideration is given to differences in the soil properties between these two sites. These properties would influence the uptake of ^{137}Cs by vegetation. For the Savannah River Site as well as for the Hanford Reservation, it is assumed in the TSD that the external exposure pathway would be dominant for any of the modeled scenarios. This assumption would not be valid for these sites if agricultural activities were to exist.

For a given radionuclide, the relative contribution to human health risk from different exposure pathways can vary considerably from site to site. Therefore, a model which incorporates only a limited number of generic pathways may be useful for one site, but may be inappropriate for another. For ^{137}Cs+D, the results presented in TSD Tables 2-8, 3-1, and 3-4 (for the rural residential exposure scenario) suggest that external radiation has a much larger contribution to the total dose than does the ingestion pathway. To the contrary, however, the ingestion pathway should dominate for any site with nutrient-poor surface soils because ^{137}Cs is rapidly recycled from such soils to plants grown on those soils (NCRP, 1993). Growth of crops on contaminated nutrient-poor or highly acidic soils will also be a major pathway of concern for sites at which ^{90}Sr, ^{129}I, and ^{99}Tc are present for the same reason. Eight of the sixteen reference sites described in the TSD contain one of these four radionuclides.

The distribution of fallout following the 1987 Chernobyl accident illustrates this point. Estimates made for southern Finland after the Chernobyl accident showed that exposure from ingestion of contaminated foodstuffs is equal to or greater than external exposure from contaminated surface soil (IAEA, 1995; Suomela *et al.*, 1991; Rahola *et al.*, 1991; Rantvaara, 1991). Figure 5-1 shows the relative contribution of ingestion and external exposure to the total dose for southern Finland. For one year and five years after the accident, the ingestion dose exceeds the external dose. When exposure is extended to a human lifetime, the contribution from external exposure becomes equal in importance to exposure from the ingestion of contaminated foods.

Furthermore, results in TSD Tables 3-1, 3-4, and 3-7 suggest that meat provides the largest contribution to the ingestion risk, and fish provides no contribution. The results for Finland showed a different situation, as can be seen in Figure 5-2. The figure shows the top three contributors to the ingestion dose delivered in the first year after the accident, in the first five years after the accident, and the lifetime ingestion dose. In the first year, milk was the major contributor, followed by fish, and beef. For the lifetime ingestion dose, fish is as important as milk, and beef is no longer one of the top three contributors to dose. Moreover, mushroom consumption becomes more important than beef, grain, fruits or leafy vegetables. None of the codes selected for use in the TSD include this pathway. Although fish and mushrooms may not be important at many U.S. sites, the possibility of atypical food sources dominating the exposure to humans should not be ignored when assessing exposures to members of critical population subgroups.

5.3.4 Leaching Rates for Anions

Table 3-1 of the TSD lists the ingestion of drinking water as the most important exposure pathway for ^{129}I and ^{99}Tc. At least for ^{129}I, this result is probably a consequence of the unrealistically low default value used by RESRAD for the ^{129}I distribution coefficient (K_d) used to model its rate of leaching from surface soil (a value of 1 is listed in Table 3-13, p. 3-43, of the TSD). Iodine-129 is expected to complex with organic matter at the soil surface, and thus be retained in this material; hence a higher K_d value is warranted for some sites. In this case, the risks associated with the ingestion pathways, particularly milk and meat, would probably dominate the total risk associated with this nuclide. Similarly, the risk associated with ^{99}Tc in surface soil should also be highly dependent on the chemical form of the radionuclide. Differences in the chemical form of ^{99}Tc were not taken into account in RESRAD. For both ^{129}I and ^{99}Tc, the food ingestion pathway should be more important than the groundwater pathway for a rural-residential scenario where the source of contamination is the surface soil.

5.3.5 Food Consumption Rates

In Chapter 3 of the TSD, a base case analysis was made of a Generic Test Site using the RESRAD code. The parameter values considered are given in Table 3-11 of the document. Explanation for the parameter values is provided on pages 3-48 to 3-50. However, the values used in the TSD calculations differ by a factor of two from those in the cited references (EPA 1989a; ANL 1993a) for the milk, meat, fish, and fruit-vegetable-grain groups of food. On page 3-49 of the TSD, EPA explains that it divides by 0.5 to account for the RESRAD contamination factor described on page 3-48. It then seems to apply its own contamination factor (0.25 for leafy vegetables). Our understanding is that RESRAD calculates contamination factors for plants, milk and meat internally (within the model structure). For the particular case of a contaminated area of 10,000 m^2, these factors are equal to 0.5. The intake rates are corrected (doubled) in order to eliminate the effect of the contamination factor. It is not clear why such a roundabout approach was taken; a more straightforward approach would have been to set the contamination factors to 1 (overriding the internal calculation) and then use the normal intake rates. In any case, better explanations are necessary.

The Agency may want to consider increasing the assumed intake value for freshwater fish. Superfund human health risk assessments typically assume a freshwater fish ingestion of about two fish meals a week, equivalent to a daily ingestion rate of about 54 g/day or approximately 20 kg/yr. This value is much higher than the 4.6 kg/yr value used for the generic calculations, which is based on a population average that includes people who do not eat fish. The fish intake rate for an RME individual should be increased to a reasonable maximum value.

5.3.6 Recommendations Concerning the Soil/Food Ingestion Pathway Model

5.3.a ***Recommendation***. With the generic qualification regarding non-uniformity of source term and population mobility (see recommendation 4.3.c), the soil ingestion pathway appears to be consistent in form with currently available methods. However, the rate of soil ingestion assumed for the industrial/commercial land-use scenario appears appropriate only for office (indoor) industrial activities; higher soil ingestion rates would be expected to occur for outdoor construction activities. Hence, the Subcommittee recommends the use of a larger default value for the industrial/commercial land-use scenario.

5.3.b ***Recommendation***. The produce, meat, milk and fish consumption pathways are simplified in comparison with state-of-the-art pathway analyses. Although the Subcommittee does not recommend that they be refined at this point, the uncertainty and variability in the risk estimates generated under the simplified assumptions should be discussed in greater detail.

5.3.c ***Recommendation***. The EPA needs to clarify whether the corrections applied to fruits and non-leafy vegetables were also applied to grains. A better explanation is also needed for the apparent discrepancies between recommended and default food ingestion values.

5.3.d ***Recommendation.*** Some of the default parameter values used by the EPA in its analysis of the food ingestion pathway lead to under-predictions of the exposure by this pathway. Specifically, the EPA should increase the fish intake value for RME individual, revise its soil-to-plant transfer factors for those sites at which radionuclide uptake by plants is expected to be higher than the default value, and consider using lower leaching rates (i.e., higher K_d values) for such nuclides as ^{129}I and ^{99}Tc for sites in which the contaminated soil contains a high organic content.

5.4 Pathway Model for Inhalation of Particles

5.4.1 General Comments

The inhalation of resuspended soil and dust is appropriately included as a potential exposure pathway for all three exposure scenarios (rural residential, commercial/industrial, and suburban). The methodology used to calculate risks from inhalation of radioactive dust at the generic site is not readily apparent from the report. Part of the confusion might be attributed to the inclusion of the comparison of the three computer codes, RESRAD, RAGS/HHEM and PRESTO-CPG, and from the use of different values for a number of the inhalation parameters. Supporting documents provide information about the models and the parameters used in the calculations, but justification for application to the cleanup conditions is lacking. Information provided about the three computer codes indicates that all use dosimetric models that have since been updated or revised. Relevant to the inhalation of radioactive particles is the fact that respiratory tract doses are calculated using the 1979 ICRP model (ICRP, 1979) which has recently been replaced (ICRP, 1994). The new ICRP model generally gives lower effective doses, such as by a factor of three, for the respiratory tract for many radionuclides. For other radionuclides, the new model gives about the same or somewhat higher effective dose values.

5.4.2 Default Parameter Values Used to Model Risks to Individuals

RESRAD (version 5.19) is used to model risks to on site individuals from inhaled particles. Selection of RESRAD followed comparative calculations with PRESTO-CPG and RAGS/HHEM codes. The following inhalation parameters are used in the risk calculations for individual members of the population:

Inhalation Rate (volume of air inhaled by an individual in one year): 7300 m^3, from HHEM Supplemental Guidance (EPA, 1991) (per Appendix C of the TSD). The rate is given as 20 m^3/day, 365 days per year. A value of 8000 m^3/yr was used in the example calculations comparing the three codes (page 2-55 of the TSD). Appendix C of the TSD also gives fractions of time spent indoors and outdoors for the three scenarios for use with the RAGS/HHEM code but there is no indication whether these were used in the RESRAD code for this report.

Mass Loading (quantity of contaminated dust contained in each m^3 of air): 1.5 $x10^{-6}$ g/m^3 for suburban exposures (EPA, 1994a); $2x10^{-4}$ g/m^3 for rural residential and commercial/industrial exposures (ANL, 1993b). The value given for suburban exposures appears to be low by at least an order of magnitude and should be verified. The residential release is in the range expected for dust particles less than about 10 μm.

Particle Size: Dose conversion factors used in RESRAD apply to dusts with an aerosol mean activity diameter (AMAD) of 1 micrometer.

Physiological Characteristics of Exposed Individual**:** RESRAD code is applicable to ICRP Reference Man (worker) (ICRP, 1975). It is not apparent that the model accommodates adjustments for members of the public.

Shielding Factor (ratio of quantity of dust in indoor air to the quantity of dust in outdoor air): 0.4 (ANL, 1993b)

5.4.3 Default Parameter Values Used to Model Risks to Populations

The following inhalation parameters are used in the risk calculations for populations: (see Table 3-24, page 3-82, of the TSD):

Inhalation Rate**:** 2400 m^3 (note: Inhalation dose conversion factors from Federal Guidance Report No. 11 (EPA, 1988) are used. These are based on calculations for Reference Man who works 2000 hours per year and breathes at a rate of 0.02 m^3/min. However, Appendix E gives the breathing rate used in the calculations as 8400 m^3. It is not specified whether committed dose equivalents per unit intake for lung or effective committed dose equivalents were calculated.)

Mass Loading**:** $50x10^{-6}$ g/m^3 (reference is given in Appendix E, page E-5, as NRC 92. It is probably NUREG/CR-5512 (NRC, 1992 in this report), but is not identified in the TSD reference list). However, TSD p. 2-55 gives a value of 100 µg/m^3, also referenced to NRC (1992); it is unclear which value was used in the analysis.

***Particle Size*:** Inhalation dose conversion factors from Federal Guidance Report No. 11 (EPA, 1988) are for aerosols with an AMAD of 1 micrometer.

***Physiological Characteristics*:** Federal Guidance Report No. 11 (EPA, 1988), page 11, says the derived guides in the report apply to Reference Man and application to other than normal occupational exposures should consider the effect of the different conditions of physiology, age, and sex on uptake and retention of radionuclides. This should also include different exposure conditions such as aerosol properties.

Slope Factors: Age-specific cancer risks for intake of radionuclides from EPA (1994b) were used.

The following inhalation parameters were used in the risk calculations for cleanup workers (see pages 5-17 of the TSD): inhalation rate, not given; mass loading, 400 µg/m^3; physiological characteristics, not given; slope factors, none used.

5.4.4 Updating RESRAD to Include the Revised ICRP Respiratory Tract Model

Although inhalation of radionuclides does not to appear to be the dominant exposure pathway at the reference sites using the methods apparently used in the draft report, it seems prudent to consider whether this would be the case if the inhalation risk calculations were more realistically based on the exposure conditions postulated for the three scenarios at the reference sites and if the risks were calculated for the general population and individuals in the populations rather than for Reference Man (a worker). There are some suggestions in the report that this was considered, but it is not clear whether parameters were adjusted for such factors as sex, age, level of physical activity, and body size. It appears that age-specific cancer risks were applied to non-age-specific estimates of radionuclide intakes. Calculations of risks to cleanup workers probably do not merit this criticism.

It was undoubtedly the intent of the draft report to calculate risks that could reasonably be applied to the general public, but the models available to EPA predated recent efforts to address exposures of the general public by the ICRP (ICRP, 1993a, 1994). Since the cleanup standard is to apply to the "reasonable maximum exposure" (RME) individual (page 2-3 of the TSD) , it should be ensured that the pathway model used to support the standard and eventually used to measure compliance with the standard, is the best available to calculate the RME. With respect to the inhalation of particles, the recently revised human respiratory tract model of the ICRP provides for calculating doses to different members of the population under various exposure conditions. Thus, using this model, the "maximum exposure that any individual is reasonably expected to receive at a site" (RME) can be identified.

For the above reasons, the Subcommittee recommends that RESRAD be updated to include the new ICRP human respiratory tract model (ICRP, 1994) which has been adopted by both the ICRP and the NCRP. If this would result in an unacceptable delay in completing the cleanup document, an attempt should be made to estimate the uncertainties resulting from applying models developed for Reference Man (workers) to the general population and to justify the approach. It may well be that the current effort results in conservative estimates of dose, but the impacts on cleanup should be estimated to determine just how conservative they are and the magnitude of associated cleanup costs. Table 5-1 that follows demonstrates that deposition of particles differs significantly among individuals in a population, depending upon age, sex and level of activity. The data are taken from ICRP Publication 66 (ICRP, 1994) which gives fractional deposition in the different regions of the respiratory tract. Only total deposition values are given in this table, although differences in regional deposition are frequently greater among members of the population than differences in total deposition and may have a greater impact on the doses received by sensitive tissues.

Depending upon the level of activity, deposition ranges from a low of 0.38 for a sleeping adult female to a high of 0.67 for a 1 yr old infant undertaking light exercise. This difference in deposition does not translate directly into a difference in dose that the two individuals would receive because of differences in the actual amounts inspired, organ and tissue sizes, etc.

5.4.5 Findings and Recommendations Concerning the Particle Inhalation Pathway

5.4.a ***Recommendation.*** The TSD should clearly specify all parameters used in calculations of dose and risk from inhalation of airborne radioactive dusts for the three identified exposure scenarios and indicate ranges of values as well as those assumed or adopted for the calculations. Although population density is addressed for all generic sites, the characteristics of the populations themselves do not seem to be addressed with respect to age, living behaviors or housing types. Some of these may be accounted for by the different scenarios, but this issue should be clarified. Analysis may conclude that none of these factors significantly affect the final results, but it should be documented.

5.4.b ***Recommendation*****.** Details of calculations of risk estimates given in TSD Table 5.9 should be described in the TSD (or, if already described, should be more clearly identified).

5.4.c ***Finding.*** The dust resuspension pathway as described in the EPA document assumes that the inhalation exposure can be described by the amount of dust expected to be in the air and that the concentration of radionuclides in the dust is the same as that in the soil under consideration. For a small site, much of the dust in the air even at the downwind edge of the site will come from unaffected regions upwind of the site, and the EPA assumption could lead to substantial overestimates of risk via this pathway. This difficulty could be

Table 5-1. Total respiratory tract deposition of 1 micrometer AMAD aerosols (Note 1)

	Breathing rate (m^3 / hr)	Fraction Deposited (Note 2)
Adult worker		
Nose breather	1.2	0.51
Mouth breather	1.2	0.34
Sleeping, nose breathing		
Adult male	0.45	0.39
Adult female	0.32	0.38
15 yr male	0.42	0.39
15 yr female	0.35	0.38
10 yr child	0.31	0.42
5 yr child	0.24	0.46
1 yr infant	0.15	0.54
3 mo infant	0.09	0.54
Sitting, nose breathing		
Adult male	0.54	0.42
Adult female	0.39	0.39
15 yr male	0.48	0.40
15 yr female	0.40	0.39
10 yr child	0.38	0.44
5 yr child	0.32	0.50
1 yr infant	0.22	0.61
Light exercise, nose breathing		
Adult male	1.50	0.53
Adult female	1.25	0.53
15 yr male	1.38	0.53
15 yr female	1.30	0.54
10 yr child	1.12	0.58
5 yr child	0.57	0.57
1 yr infant	0.35	0.67
3 mo infant	0.19	0.66

Heavy exercise, nose breathing		
Adult male	3.00	0.42
Adult female	2.70	0.42
15 yr male	2.92	0.41
15 yr female	2.57	0.41
10 yr child	2.03	0.43

Note 1. Values from ICRP 66 (ICRP, 1994)
Note 2. "Fraction deposited" is the fraction of activity present in a volume of ambient air before it is inspired, that is deposited.

corrected by adjusting either the dust loading or the concentrations downward depending on the size of the area under consideration. Moreover, real atmospheric dust loadings vary substantially with site and depend on such factors as soil particle size distributions, vegetative cover, humidity, and precipitation patterns.

5.4.d ***Finding.*** The dust resuspension pathway does not provide for differences in dust generation that might be expected in dry climates, where much of the soil may be exposed to wind and easily lofted, as opposed to moist climates, where vegetative cover and soil moisture may be effective in suppressing dust lofting over much of the year.

5.4.e ***Recommendation.*** The TSD is unclear whether the assumed dust loading is all respirable. EPA has stated orally that it is; this point should be clarified in the text and the appropriateness of the loading assumptions verified. In particular, the rationale for using different mass loading assumptions for individual and population risk calculations should be given.

5.4.f **Recommendation.** The risk models available to EPA predated recent efforts to address exposures of the general public by the ICRP (ICRP, 1993a, 1994). Consequently, the Subcommittee recommends that RESRAD be updated to include the new ICRP human respiratory tract model (ICRP, 1994) which has been adopted by both the ICRP and the NCRP. If this would result in an unacceptable delay in completing the cleanup document, an attempt should be made to estimate the uncertainties resulting from applying models developed for Reference Man (workers) to the general population and to justify the approach.

5.5 Pathway Model for Inhalation of Indoor Radon and Progeny

5.5.1 General Comments

The document emphasizes that "The inhalation of radon and its decay products is a major contributor to total exposure when radium isotopes are present in the soil." Methods used to estimate doses and risks from inhalation of indoor radon involve modeling the buildup of radon and radon progeny indoors and then multiplying these concentrations by a dose or risk conversion factor which relates airborne concentrations of radon progeny to dose and risk. The uncertainty in the estimated risks from indoor radon is associated with these components, that is the indoor radon concentration and the risk conversion factor. RESRAD is used to model indoor radon and radon progeny concentrations for calculating individual doses and risks, and CU-POP is used to calculate population doses and risks. PRESTO-CPG does not include a pathway for inhalation of radon.

5.5.2 Individual Risks

RESRAD, which is used to calculate individual risks, employs a diffusion model based on empirically derived constants to estimate the flux of radon into a home. As noted in the document, RESRAD does not account for advective flow. This omission can result in underestimation of indoor radon concentrations. However, of greater concern is the large variability in the radon entry into individual homes. As noted in the document, the ratio of the indoor radon concentration (pCi/L) to the soil ^{226}Ra concentration (pCi/g) for individual homes can range over several orders of magnitude. The document also appropriately notes that "The buildup of indoor radon is highly site-specific and cannot be reliably predicted for individual homes." Even when site-specific parameters are used in the model, there is likely to be a large uncertainty in the calculation of the indoor radon concentration in an individual home, and there would be variability among homes on the same site. A single value of the risk estimate does not reflect the uncertainty and variability in indoor radon concentrations. (It is also stated in the document that the variability in the ratio of indoor radon concentration to the soil ^{226}Ra concentration is likely to be relatively small when the parameter of interest is the ratio for large populations and long periods of time, which seems reasonable.)

The methodology presented in the RESRAD manual (DOE, 1993a) for calculating radon risk for a specified indoor radon concentration is different from the methodology normally used by the EPA (also used in the CU-POP model to calculate the population risk), which previously has been reviewed by the RAC (SAB, 1992). Exactly what was done using RESRAD is not clearly presented, but it appears that an effective dose equivalent was first calculated, and that this dose was multiplied by a risk conversion factor, which was determined largely from Japanese A-bomb survivors. This is not the recommended approach for estimating risks from radon exposures (ICRP, 1977)

The analysis of indoor radon appears to depend solely on the diffusion of radon from soil into a building. If so, the accounting for all significant transport mechanisms is incomplete. The dose contribution from buildings contaminated with radium is not addressed. Further, it is unclear how indoor radon can be modeled for all future structures in that assumptions about building characteristics of housing in the future can substantially influence the radon risk estimates.

5.5.3 Population Risks

The document points out the fact that CU-POP does not explicitly model the transport and buildup of indoor radon. Rather, the model uses an empirically determined relationship between indoor ^{222}Rn concentrations and ^{226}Ra concentrations in soil. A similarly derived relationship is used to estimate the ^{222}Rn concentration in outdoor air. This approach seems reasonable for estimating population exposures to radon because individual variations would tend to average out. The risk associated with the indoor radon concentration is computed assuming an equilibrium fraction of 0.5, an indoor occupancy factor of 0.6, and a risk conversion factor of 2.36×10^{-4} per Working Level Month (WLM). Depletion of ^{226}Ra in the soil by decay and leaching is considered when calculating future risks.

The risk conversion factor of 2.36×10^{-4} per WLM previously has been reviewed by the RAC (SAB, 1991). This value was determined by the EPA using the model recommended by the National Academy of Sciences Committee on the Biological Effects of Ionizing Radiation in their 1988 BEIR IV report (NAS, 1988), along with the adjustments for homes recommended by the National Academy of Sciences in their 1991 report titled "Comparative Dosimetry of Radon in Mines and Homes" (NAS, 1991). The International Commission on Radiological Protection has recommended use of a similar risk factor of 3×10^{-4} per WLM (ICRP, 1993b).

The methodology for calculating the risk corresponding to outdoor radon concentrations is not presented. The equilibrium fraction and risk coefficient for this application would differ from those applied to indoor radon concentrations.

5.5.4 Technical Feasibility

The TSD shows that, for the rural residential and suburban land use scenarios, when radon is considered, the radium soil concentration corresponding to a risk level of 10^{-4} or a dose of 15 mrem/yr is about an order of magnitude lower than the typical natural background soil concentrations of 0.2 to 4.2 pCi/g (presented on TSD p. 7-18). Consistent with this fact, the document notes that "As a general rule, notwithstanding the uncertainties in the risk factors, if the cleanup for a site containing ^{226}Ra is set at $1x10^{-4}$ to $1x10^{-3}$ or less and includes the potential for the buildup of indoor radon, virtually all of the soil contaminated above background will need to be remediated." Thus, some method would be needed in the proposed regulations to determine whether soil contamination exceeds background concentrations.

5.5.5 Findings and Recommendations Concerning the Radon Inhalation Pathway

5.5.a ***Finding.*** The methodology used for estimating population risks from exposure to radon indoors (for a specified radium soil concentration and population density) is reasonable, and the risk conversion factor used in this calculation previously has been reviewed by the RAC (SAB, 1991).

5.5.b ***Recommendation.*** The total uncertainty associated with individual and population risk estimates should be presented. Of great concern is the orders of magnitude variability in the radon entry into individual homes (as noted in the document), and the large uncertainty that will exist in a single calculated value that is used to estimate the radon concentration in any home. The implications of the expected large uncertainty in the individual risk estimate should be explored.

5.5.c ***Recommendation***. The methodology used in RESRAD for estimating individual risks should (but does not) account for advective flow of radon into a home. This omission can result in underestimation of indoor radon concentrations.

5.5.d ***Finding***. The document shows that for the rural residential and suburban land use scenarios, when radon is considered, the radium soil concentration corresponding to a risk level of 10^{-4} or a dose of 15 mrem/yr is about an order of magnitude lower than the typical natural background soil concentrations of 0.2 to 4 pCi/g. Thus, a key aspect of the proposed rule would be a method for determining whether soil contamination exceeds background concentrations.

5.5.e ***Recommendation.*** The methodology used in RESRAD for calculating radon risk for a specified indoor radon concentration is different from the methodology normally used by the EPA (also used in the CU-POP model to calculate the population risk), which previously has been reviewed by the RAC (SAB, 1991). The implications of these differences should be investigated.

5.5.f ***Recommendation.*** Consideration should be given to the implications of the fact, as noted in the TSD, that the buildup of indoor radon can be greatly diminished if a home is designed to preclude radon entry.

5.5.g ***Finding***. For a large site, site-related radon exposures may occur even when not "at home" or "at work." In the population risk analysis, the radon risks depend on a ratio method for estimating radon in indoor and outdoor air from radium concentrations in soil, which effectively assumes that a person is only affected by locally generated radon. This assumption may lead to a slight underestimation of impacts because it ignores the transport of site-related radon off-site. However, the off-site radon impacts would likely be much smaller than those of background radon.

5.5.h ***Recommendation***. The TSD produced estimates of soil remediation volumes and cancers avoided with and without consideration of radon at every site with radium in soil. Although not stated in the document, this option was apparently selected to enable the drafters of the cleanup rule to understand the difference in results if the radon issue was handled separately, for example by requiring radon remediation in any residences or places of employment built on the sites (or by assuming that radon-resistant construction will become standard everywhere). Although policy considerations may also have influenced this option, the differential treatment of radon seems to require further explanation in the TSD.

5.6 Pathway Model for Groundwater Transport and Ingestion

5.6.1 Effect of Depth of Contamination on Relative Importance of Groundwater Pathway

A critical issue not addressed in the TSD is the depth to which soil cleanup may be feasible and reasonable, i.e., at what depth does contamination become an aquifer cleanup problem instead of a soil cleanup problem? At various sites including Fernald, contamination has clearly penetrated below soil levels (e.g., see p. 4-44, first paragraph, of the TSD). If the surface soils at Fernald were "remediated" (i.e., removed), the main radiological risk would be from the deeper soils not considered in the calculations. One could conceive of situations in which this risk would exceed the risk posed by the original "unremediated" soils. Long-lived radionuclides with small distribution coefficient values (K_ds) would be the most likely candidates for this sort of behavior. By using unrealistically large K_d values (e.g., 1600 for uranium at Reference Site II based on Fernald), the potential risks associated with deeper contamination are ignored. The danger is that adequate risk reduction may not be achieved after remediation of shallow soils if deeper soil remains contaminated.

The Subcommittee recommends that the EPA conduct sensitivity analyses on K_d values, focusing on long-lived redox-sensitive radionuclides such as uranium and plutonium, in order to evaluate the extent to which the relative importance of the groundwater pathway is dependent upon this variable. A similar concern is expressed for the effect of non-radioactive mobilizing agents, such as organic solvents, which could also enhance the migration of otherwise relatively immobile radionuclides.

5.6.2 Hydrologic Parameter Values for the Generic Site Model

According to the RESRAD documentation (pp 198-200), the volumetric water content is determined by estimating a yearly average. However, recharge events are intermittent and transient phenomena, particularly in arid environments. What errors are introduced to estimation of leaching using this average approach? Also, as indicated in the RESRAD documentation (p. 290), dispersion is not considered in the RESRAD code. As a result, breakthrough concentrations will be overestimated by the model.

As stated in the TSD (p. 3-30), "The parameter values used in the calculations were selected as realistic but conservative estimates of the conditions at the generic test site for each of the three scenarios [suburban, rural residential, and commercial / industrial]." However, Table 3-11 of the TSD shows that the hydrological data are identical for all three scenarios. Effectively, the groundwater / leaching model is simulated using only one scenario. This scenario is illustrated in Figure 3-4 of the TSD. Given the scenario in this figure and the data in Table 3-11, this generic test site is very conservative with inconsistencies in the hydrological data, as follows.

The depth to the water table for the generic site is only 4 m (2 m of contaminated zone plus 2 m of vadose zone). For many sites in the western United States, the depth to the water table is greater by one to two orders of magnitude. The given parameters of hydraulic conductivity K = 5,550 m/yr, hydraulic gradient i = 0.02, and porosity $\Phi = 0.2$ yield a groundwater velocity of 555 m/yr (1821 ft/yr). Very few aquifers have velocities that are this high, and this velocity is overly conservative. Hydrologic parameter values reported in TSD Table 4-5 (p. 4-24) for Reference Sites -A, -B, and -C (subsets of Reference Sites XIII and XVI-XXI) lead to even more extreme and unrealistic groundwater velocities of 4.5 to 6.7 km/yr. In these cases, the large hydraulic gradients assumed for these cases appear to be inconsistent with the associated large hydraulic conductivities. Velocities calculated for the remaining reference sites range from 2 mm/yr (Reference Site IX, based on Rocky Flats) up to 240 m/yr (Reference Site II, based on Fernald).

The pumping for all scenarios is 250 m^3/yr or 0.13 gal/min. This pumping rate is low. The well intake depth is at 3 m below the water table, which is too shallow. Such well construction for many types of wells is probably not allowed in many States. Because of seasonal water-level fluctuations and for more realistic pumpage and groundwater flow conditions, a pump set so shallow would go dry many times during the year. Such a depth is overly conservative and unrealistic.

For the generic base case, the precipitation is 1 m/yr and infiltration is 50% or 0.5 m/yr (Table 3-12 of the TSD). These values are very high and overly conservative. In the RESRAD documentation (p. 198), this infiltration rate is based on an irrigation rate of 0.2 m/yr. What about sites where there is no irrigation? What is the justification for selecting a hydraulic conductivity above the water table that is 25 times less than the hydraulic conductivity below the water table?

The infiltration rate and thickness of the vadose zone are addressed in the sensitivity analysis. Other parameters varied in the sensitivity analysis include area of the contaminated zone, thickness of the contaminated zone, and distribution coefficients. No sensitivity analysis was performed on any aquifer parameters.

In summary, several aspects of the generic site model are overly conservative and unrealistic. The importance of these parameters was not assessed in a sensitivity analysis. Additional work is required before the importance of the drinking water pathway can be fully assessed.

5.6.3 Transport Processes and Parameter Values

A major assumption in the source-term definition is that the soil volume containing the highest concentration of the dominant nuclide under consideration also contains the highest concentration for each nuclide at the site. This assumption implies that all radionuclides migrate together at roughly the same velocity, which is not generally the case. There is no discussion of how this assumption may or may not bias the result of the calculations.

Radionuclides such as ^{137}Cs, ^{226}Ra, ^{232}Th, and ^{239}Pu are generally highly attenuated in soils due to their high affinity for various mineral surfaces. Radium-226 and ^{137}Cs are somewhat of a special case because they are sorbed primarily by ion exchange processes. If ion-exchanging minerals (e.g., clays) are deficient in the soil, ^{226}Ra and ^{137}Cs may not be attenuated as much as is assumed in the calculations. For this and other reasons, it is clear

that site-specific data (e.g., actual depth of radionuclide penetration) are essential for realistic risk calculations.

Existing data in the scientific literature suggest that colloidal species can enhance the subsurface transport of radionuclides, such as ^{232}Th and ^{239}Pu, that otherwise would be fairly immobile (e.g., see reviews of this subject by McCarthy and Zachara (1989) and Triay *et al.,* (1995)). Similarly, organic solvents may increase the mobility of radionuclides. Yet these transport mechanisms are completely ignored in the TSD modeling.

The RESRAD groundwater module assumes perfect mixing in an idealized homogenous aquifer with linear equilibrium adsorption and radioactive decay. In reality, the processes determining radionuclide speciation are much more complex than given by the linear equilibrium adsorption model, with adsorption and phase transformations dependent on pH, oxidation-reduction (redox) state and multicomponent interactions. A geochemical model is required to represent such interactions. At many sites, deviations from the ideal groundwater model in RESRAD may also occur due to the presence of non-aqueous phase liquids (which may result in facilitated transport of radionuclides), fractured media, and nonequilibrium sorption. The EPA should bound the potential impacts on radionuclide transport of differences in local geochemical conditions at reference sites (e.g., redox state of uranium).

The equation used by RAGS/HHEM in section 2.1.7.2 (bottom of p 2-33 of the TSD) relates the groundwater (soil-water) concentration (C_w) to the soil concentration (C_s). Ignoring the dilution factor (DF, which transforms groundwater concentration in the contaminated soil zone to that at the receiving well), the equation can be rearranged to express the soil concentration C_s as a function of the groundwater C_w concentration in the contaminated zone:

$$C_s = [K_d + \theta S/\rho)]C_w$$

The first part of the equation ($C_w K_d$) represents the radionuclide that is actually bound to the soil matrix. The second part $[(\theta S/\rho)C_w]$ represents the radionuclide that is in the interstitial water (θ=soil porosity; S=fraction of porespace saturated with water; and ρ=bulk soil density). This is an appropriate way to represent C_s. It implies, however, that such a relationship is in fact how C_s will be measured when implementing the regulation, i.e., take the total mass of radionuclide in the sample (soil+interstitial water) and divide by the dry weight of the soil. Because the definition of "soil concentration" is a matter of controversy, this needs to be explicitly addressed–and the issue does not appear to be addressed anywhere in the review document. Because this is an issue related to the implementation of the proposed regulations, it is again addressed in Chapter 8.

Underlying assumptions and model formulations for the individual transport processes are very important in deriving reasonable risk levels. For example, for the infiltration / leaching module, the simple linear basic equation used in RESRAD is logical and easy to understand:

$$I = (1 - CE)\,[(1 - CR)\,P + IR]$$

where I = infiltration rate, CE = evapotranspiration, CR = runoff, P = precipitation rate and IR = irrigation rate. However, the parameter values often have wide ranges and, in many cases, are not available for a specific site. When the parameter values are selected from the literature or when heavy reliance is placed on the default values in the code, compound errors can be very large, even a couple orders of magnitude. This point emphasizes the need for uncertainty analyses, in which the effect of ranges for each parameter can be appropriately taken into account .

5.6.4 Findings and Recommendations Concerning the Groundwater Pathway

5.6.a ***Recommendation.*** With respect to the groundwater pathway, several aspects of the generic site model are overly conservative and unrealistic. The importance of these parameters was not assessed in a sensitivity analysis. Additional work is required before the importance of the drinking water pathway can be fully assessed.

5.6.b ***Finding.*** The population risk model (CU-POP) incorporates the assumption that population exposures and risks can be directly related to the inventory of radionuclides at a site through the assumption that a constant fraction of the groundwater originating at the site is consumed for drinking water by some person, independent of the distribution of on-site and off-site residents. In reality, groundwater use is probably extremely dependent on such site-specific factors as the quality of the water before being affected by radionuclides, details of the regional hydrogeology, and discharge rates to surface water.

5.6.c ***Recommendation***. EPA should make a greater effort to identify and use site-specific K_d values for long-lived radionuclide species sensitive to oxidation/reduction (redox) conditions. Sensitivity analyses should be conducted on K_d values for specific reference sites where these nuclides are present (i.e., not just for the generic base case) in order to assess the sensitivity of the model results to this parameter. The objective should be to bound the potential impacts on radionuclide transport of differences in local geochemical conditions at reference sites (e.g., redox state of uranium).

5.7 Pathway Model for Surface Water

The surface water models for RESRAD and RAGS/HHEM are very simplified, considering a pond receiving runoff from the site diluted according to the fraction of the contributing watershed which is occupied by the site. The runoff concentration associated with the site is the leachate concentration of the interstitial soil water in the contaminated zone. This approach ignores contributions from particulate-phase radionuclides associated with soil that is eroded from the site as part of the rainfall-runoff process. For sites having small contaminated areas, dilution due to runoff from the uncontaminated parts of the watershed may be sufficient to negate the importance of this exposure pathway. However, the aftermath of the Chernobyl accident is relevant in that it shows that erosion can be a significant source of contamination for surface water bodies when large portions of the watershed areas are contaminated. Radionuclides can be transported by runoff water either in solution or attached to particles, according to the chemical and physical properties of the contaminant. In the case of the Chernobyl nuclides, both kinds of transport were identified. However, the main point here is that surface water contamination by water runoff must not be neglected because there are conditions when it is more important than contamination by ground-water transport (Konoplev *et al*., 1995; Bulgakov and Konoplev, 1992; Saxén and Rantvaara, 1987; Broberg and Andersson, 1991; Hammar *et al*., 1991). Consideration should be given to adding a soil erosion/runoff component (with associated radionuclide concentrations) to the surface water model for such sites.

5.7.a ***Recommendation***. The Agency should encourage and seek inclusion of soil erosion and particulate-phase radionuclides in the surface water modules of future models used for assessment of soil cleanup standards.

6. METHODOLOGIES FOR ESTIMATING RISK TO INDIVIDUALS AND POPULATIONS

6.1 Risk Coefficients

The outputs of the pathway models are radionuclide intake rates for exposed individuals by inhalation and ingestion (or, in the case of external gamma radiation, the whole body dose rate). In order to estimate the risks from these exposures, intake rates must be converted to equivalent radiation doses and then to cancer risk or, in some cases, directly to cancer risks. These conversions are accomplished through risk coefficients based on observations of radiogenic cancer in humans with supporting data from experiments with laboratory animals.

In its risk assessment supporting the cleanup standard, EPA used two different methods for expressing risk coefficients. EPA apparently used the methods of the ICRP (although not the most recent ICRP recommendations) to calculate an effective dose[7] through models of organ doses from ingested or inhaled radionuclides. This method was used to derive a dose rate standard (e.g., 15 mrem per year).

EPA also calculated risks directly from intake rates using slope factors which appear to have been derived usually through models that convert intake to organ dose and then organ dose to risk. For some radionuclides, the models appear to differ from the ICRP models, and for others, risk per unit intake is expressed directly, as for radon. The slope factor method was used to derive a standard expressed in risk terms (e.g., 1 in ten thousand lifetime cancer risk).

Nine references were identified in the TSD as sources for dose conversion coefficients and slope factors for both external and internal exposures to the RME individual and the population:

a) Dose conversion factors:

EPA (1993a): Federal Guidance Report No. 12
EPA (1988): Federal Guidance Report No. 11
DOE (1993a): RESRAD manual

b) Slope factors:

7 "Effective dose" is the currently preferred terminology replacing the term "committed effective dose equivalent (CEDE)."

NAS (1988): BEIR IV, with adjustments recommended by the National Academy of Sciences (radon only), as presented in EPA (1992b)

EPA (1992a): Health Effects Assessment Summary (HEAS) Tables

EPA (1993b): Health Effects Assessment Summary (HEAS) Tables

EPA (1994b): Estimating Radiogenic Cancer Risks

EPA (1994a): TSD Appendix Table B-1 (nominally based on EPA, 1993b; but actually from Puskin, pers. commun.; see discussion below)

c) Dose-to-risk conversion factor:

EPA (1989a): Risk Assessment Guidance for Superfund, Vol. 1: Human Health Evaluation Manual

Dose conversion factors for internal exposures for both population impacts and RME doses were, according to the TSD, taken from RESRAD. Dose conversion factors for external exposures were taken from Federal Guidance Report No. 12 (EPA, 1993a).

The TSD states that slope factors used to derive RME risks were taken from 1993 HEAST (EPA, 1993b). This statement in the TSD is incorrect and leads to confusion with regard to the risk analyses. In fact, the values used, as given in Appendix Table B-1 of the TSD, are from a personal communication with Jerry Puskin of EPA, not from the cited reference. To model population risks, both "total" as well as "fatal" cancer values were taken from the EPA report, Estimating Radiogenic Cancer Risks (EPA, 1994b); these values are also the most recent HEAST numbers. The confusion in the TSD with regard to the source of slope factors used in deriving RME risks makes it difficult to verify the accuracy of the calculated risks.

RESRAD dose conversion factors are presumably based on the ICRP methodology, which weights equivalent dose according to the risk of fatality. The slope factors used in the TSD reflect total risk of cancer, not risk of fatal cancer. Therefore, comparing effective dose based on the ICRP methodology to RME risk is misleading and not valid. See also Section 6.4, below.

Some confusion also exists with respect to the exposure time periods used in the document. For the RME, the exposure is assumed to occur over a thirty-year period, from birth to age 30. For the population impacts, the exposures are assumed to continue for a lifetime, 70 years. EPA has stated orally that it is reasonable to assume that the most exposed person will remain so at most 30 years, but that such a person could be born just as exposure begins, whereas for population risk, people will move in and out of high exposure areas, but overall they will be exposed 70 years. This rationale needs to be more clearly stated in the TSD.

The TSD also refers to a 50-year dose commitment. This phrasing may be a source of confusion for readers unfamiliar with health physics terminology and should be explained in the TSD. The 50-year dose commitment is the total effective dose from intake of a radionuclide, i.e., the integrated dose from the time of intake to 50 years post intake. This is the period over which an exposure received at age 20 (the beginning of an occupational exposure) is committed. For radionuclides that reside in the body for long periods of time, this dose may be spread over 50 years, but for shorter lived radionuclides, the 50-year dose commitment may be incurred mostly in a much shorter period.

In the remainder of this chapter, the Subcommittee discusses first the calculations that derive doses from intake estimates and, second, the conversion of dose to risk. Finally, the difficulties of using two different systems are discussed.

6.2 Exposure-to-Dose Calculations

It appears that not even the calculations of dose from risk in the RESRAD code are based upon the latest metabolic and dosimetric models that have been published by the ICRP and the National Council on Radiation Protection and Measurements (NCRP) in recent years. In the past, the overall policy of EPA (as well as those of DOE, NRC and DOD) has been to use the models recommended by the ICRP and NCRP. For example, Federal Guidance Report No. 11 (EPA, 1988) gives limiting values of radionuclide intake and air concentration and dose conversion factors for inhalation, submersion and ingestion derived directly from ICRP publications. The values are appropriate for a "reference man," specifically, a male worker. Because Federal Guidance Report No. 11 was published in September 1988, it does not reflect the work of the ICRP over the past seven years in developing models for all members of the public. EPA does not seem to have implemented the more recent ICRP 60 (ICRP, 1990) recommendations or NCRP (1993) recommendations, as evidenced by the use of the obsolete term, committed effective [annual] dose equivalent (CEDE) (see TSD page 3-2) instead of "effective dose," which is in current use world wide. It is noted, too, that EPA (as well as other government agencies) have not yet adopted SI units, although Federal Guidance Report No. 11 gives values in both SI and conventional units.

In its January 1995 presentation to the Subcommittee, ORIA stated that EPA is required to use Federal Guidance Report No. 11 (EPA, 1988) in developing cleanup standards. Federal Guidance Report No. 11 was prepared to implement "Radiation Protection Guidance to Federal Agencies for Occupational Exposure," signed into law by President Reagan on January 20, 1987. Federal Guidance Report No. 11 thus was intended to provide protection against the intake of radionuclides in the workplace and, as such, its application to members of the public may not be appropriate. At best, EPA's use of Federal Guidance Report No. 11 makes its projection of the "maximum exposure that [any individual] is reasonably expected to [receive] at a site" (RME) quite uncertain. Recent publications of the ICRP (ICRP, 1989, 1993a) show that the adult is more frequently the least exposed and the 3-month old infant is the most exposed individual in a population following ingestion of radionuclides. Examples are given in Table 6-1 on the following page, which compares ingestion dose coefficients to age 70 yrs for adults and 3-month old children (ICRP, 1993a). The values of dose coefficients for other members of the public are generally between those for the adult and 3-month old child.

Except for the transuranic elements, the dose coefficients for adults are similar to those given in Federal Guidance Report No. 11 (EPA, 1988). Metabolic models for the transuranics were updated after publication of Report No. 11. The ingestion dose coefficients for 3-month old infants range up to almost 50 times those for the adult. It is to be noted, however, that the effective dose coefficients tabulated in ICRP (1993a) and illustrated in the above table are for acute intakes and that coefficients for chronic intakes (30 years in the case of the EPA scenarios) may be less where growth is substantial during the exposure period. Nevertheless, the lifetime effective dose will be greater for a person whose exposure begins as a child than for a person whose exposure begins as an adult, even though the duration of exposure is the same.

Whether or not EPA's conclusions about soil volumes requiring cleanup or cancers averted by cleanup would be significantly affected by the differences in Federal Guidance Report No. 11 and the current ICRP recommendations is unknown. Ideally, EPA should update its analysis using the latter. At a minimum, it should state the existence of more recent and age-specific dose coefficients and explain what the effect might be on the results.

Table 6-1. Ingestion Dose Coefficients to Age 70 Years (Sv/Bq) (from ICRP, 1993a)

Radionuclide	Adult	3-Month Old Child
H^{3} water	1.8 E-11	6.3 E-11
H^{3} organic	4.2 E-11	1.2 E-10
C^{14}	5.8 E-10	1.6 E-09
Ru^{106}	7.0 E-09	8.4 E-08
Cs^{137}	1.4 E-08	2.1 E-08
Ce^{144}	5.2 E-09	6.6 E-08
Pu^{238}	2.3 E-07	4.0 E-06
Pu^{239}	2.5 E-07	4.2 E-06
Pu^{240}	2.5 E-07	4.2 E-06
Pu^{241}	4.8 E-09	5.7 E-08
Am^{241}	2.1 E-07	3.7 E-06
Np^{237}	1.1 E-07	2.0 E-06
Ra^{224}	6.3 E-08	2.8 E-06
Ra^{226}	2.8 E-07	4.7 E-06
Ra^{228}	6.6 E-07	3.1 E-05
Sr^{89}	2.6 E-09	3.6 E-08
Sr^{90}	2.8 E-08	2.3 E-07

6.3 Exposure-to-Risk Calculations

The calculations of risk directly from exposure, both in the RESRAD code and in the EPA model for estimating cancers averted from the radionuclide inventory at a site, are based on coefficients developed independently by EPA (1994b) from information provided by the National Academy of Sciences (NAS, 1990) and other sources. In developing those estimates, EPA requested the RAC to review the methods for estimating risk from estimated doses to the whole body or to various organ systems. The RAC did not review EPA's methods for estimating dose from exposure to specific radionuclides. While the methods for estimating risk from dose were found acceptable by the RAC (SAB, 1992), they differ in detail from those used by the ICRP. The methods for estimating dose from exposure apparently are also different from those used by the ICRP. Together, these differences lead to sometimes markedly different results for various nuclides.

Lacking a clear statement of the above history, it is easy to be skeptical of the EPA approach, which uses the slope factor tables in EPA (1994b) to translate radioactivity intakes by ingestion and inhalation directly to risks for each radionuclide and for each body tissue and organ. The EPA methods for translating exposure to dose and dose to risk must be understood from the original document (EPA, 1994b), not from the TSD.

The Subcommittee cautions EPA to be consistent when it specifies how compliance with its standard would be verified. Perhaps the final TSD will dictate how doses are to be calculated at the cleanup sites to assess compliance with the cleanup standard (15 mrem/yr is being suggested). In any case, the dosimetric methods used for the compliance calculations should be the same as those which led to the establishment of the risk associated with a dose rate of 15 mrem/yr. There is a potential for confusion if the risk limits are established with the EPA risk factors, while the doses to determine compliance are calculated with the methods of the ICRP and NCRP (Effective Dose using quality factors and organ weighting factors) (ICRP, 1989).

In the NRC Generic Environmental Impact Statement (GEIS) in Support of Rulemaking on Radiological Criteria for Decommissioning of NRC-Licensed Nuclear Facilities (NRC, 1994), it is noted that the EPA risk factor published in the EPA NESHAPS Background Information Document (EPA, 1989b) was used. Because the EPA has recently revised its risk factor (EPA, 1994b), the application of this risk factor to estimate the "maximum exposure (effective dose) that any individual is reasonably expected to receive at a site" for the TSD would not seem to be a departure from EPA-approved methodology.

6.4 Use of Two Methods for Relating Exposures to Risks

As explained above, EPA uses two methods for translating estimates of exposure to estimates of cancer risk, one that passes through an explicit estimate of dose and one that jumps directly from exposure to risk. If EPA chooses to continue calculating risks in both ways, it should still recognize the difficulties inherent in such an approach.

Because the two methods have different endpoints (fatal cancer vs. total cancer incidence) and are derived using different methodologies, the results should not be directly compared. Nevertheless, the EPA has effectively compared them, calculating values of "risk per mrem" by dividing the ICRP values for dose per unit intake into the EPA values for risk per unit intake. If the two methods were essentially identical, then the ratios for all radionuclides should be the same and equal to the risk per mrem value derived from the Japanese data. In fact, the ratios vary substantially: in Tables 3-1 to 3-3 of the TSD, using the RESRAD assessment code, they range from 3.66×10^{-8} per mrem to 2.96×10^{-6} per mrem, a factor over 80. With the PRESTO code (not used for the actual assessment), the range is a factor of 300. Because the dose per unit intake varies by organ system and therefore with the route of exposure, the ratios also differ for different exposure scenarios.

The risk per mrem values for the more important radionuclides tend to cluster near the mean and the importance of the greater variations is less than it might first seem. Nevertheless, the large variation is clear evidence for the difference in the two approaches. The inclusion of risk per mrem in the tables is at best confusing and may adversely affect the credibility of the analysis.

6.5 Comparison between TSD and NRC Estimates of Mortality from Exposure to Residual Radioactive Soils at NRC Licensed Nuclear Facilities

The Nuclear Regulatory Commission (NRC) and the Environmental Protection Agency (EPA) have both conducted analyses of the impacts of various levels of cleanup of radionuclide contamination at nuclear facilities in support of cleanup standards. The objective of the NRC analyses was to assess the environmental impact of the proposed amendment to the regulations in 10 CFR Part 20 to include radiological criteria for decommissioning of land and structures at nuclear facilities licensed by the NRC [Generic Environmental Impact Statement (GEIS); NRC, 1994]. The objective of the EPA analysis was considerably broader, in that it was to assess the environmental impact of the proposed regulations in 40 CFR Part 196 that would specify radiological criteria for soil, ground water, surface water, and structures at Federal facilities to be released for public use.

The EPA's Technical Support Document (TSD) (EPA, 1994a) and the NRC's GEIS used similar types of information, such as estimated source terms, to derive estimates of impacts, including potential fatal cancers, for various cleanup and occupancy scenarios. Although the methods used by the two Agencies differ significantly, a comparison of the two documents was judged worthwhile in that it could provide a degree of confidence in the TSD estimates to the extent that there were cases for which the results of the two analyses were similar.

The TSD includes 22 reference facilities in the analysis of impacts; the GEIS uses ten reference facilities. Four of the reference facilities are common to both documents: commercial nuclear power reactors, research reactors, rare earth extraction facilities, and uranium fuel fabrication facilities. Appendix B (TSD) provides details of the comparison of the TSD and GEIS source term assumptions, including radionuclide concentrations in soil and areal extent and volume of soil contamination, the use of the source terms in calculating impacts, and the results of the analyses in terms of fatal cancers projected at various residual dose values. The comparison shows that EPA estimates of soil volume at these reference sites were significantly larger than NRC estimates, by factors up to 100. However, despite the difference in methods and in source term characterization, the two agencies derived similar estimates of the number of fatalities associated with a population residing on the contaminated soil, well within a factor of ten of each other, for three of the four reference facilities. The exception was the case of the uranium fuel fabrication facility, for which mortality estimates in the TSD were higher than those calculated by the NRC by factors ranging from 2.5 to 100. The reasons for this difference are not clear and should be investigated by the EPA.

6.6 Findings and Recommendations Concerning Risk Estimation Methodologies

The set of risk coefficients used in the TSD for risk-based standards differs from that used for dose-based standards, and neither set incorporates the latest changes recommended by the ICRP or NCRP. This choice can lead to considerable confusion on the part of the reader. While the Subcommittee understands the difficulties EPA faced in deciding how to reconcile its internally generated risk coefficients with those in RESRAD derived from Federal Guidance Reports Nos. 11 and 12, leaving the problem unresolved can lead the unwary user of the TSD into erroneous conclusions. Although the Subcommittee is not specifically recommending that the Agency undertake the magnitude of effort needed to produce a completely consistent document, it does make the following recommendations:

6.6.a ***Recommendation.*** The TSD should include a discussion of the overall intent or objective of the proposed annual dose limit as it applies to cleanup (e.g., 15 mrem/yr). The selection of appropriate assumptions and parameter values for modeling RME exposures will depend upon whether the standard is developed as an annual dose limit for "any" member of the public or as an annual measure of compliance with the long-term (lifetime) individual risk limit. The assumptions used to define the RME depend on the length of the exposure period that is relevant to assess compliance with the protective limit. If the annual limit is intended to serve as a surrogate for the lifetime risk objective, then the approach is generally acceptable and the importance of the age-specific factors presented below is diminished. If, however, the goal of the standard is to assure that every individual is explicitly protected to the annual limit, then the Subcommittee recommends that EPA consider revising the TSD analysis to incorporate the age-dependent dose factors developed by the ICRP and NCRP. This cannot be achieved using the exposure-to-dose conversion factors tabulated in Federal Guidance Report No. 11 which are based on metabolic and dosimetric models developed for workers. However, proposed Federal Protection Guidance for Exposure of the General Public permits the use of ICRP dosimetric models and conventions.

6.6.b ***Recommendation.*** The Subcommittee recommends that EPA give consideration to adoption of the recent recommendations of the ICRP and NCRP that provide updated metabolic and dosimetric models and approaches for calculating age-dependent doses for the inhalation and ingestion of radionuclides for all members of the public. Adoption of the ICRP and NCRP approaches offers a procedural advantage to the EPA in that they have been extensively peer reviewed and are widely accepted. Use of these approaches would likely increase the technical acceptance of the standards by the scientific community. If alternative approaches are used in the TSD, then EPA should explicitly explain its methodology and justify its reasons for departures from ICRP and NCRP recommendations.

6.6.c ***Recommendation.*** The TSD should include a section specifically describing the dose conversion factors and the slope factors used for each of the types of assessments. While this information can for the most part be gleaned from the various chapters, it is currently difficult to sort out. In addition, deviations from published values should be adequately explained and the methodology for making adjustments described.

6.6.d ***Recommendation.*** The Subcommittee recognizes that, by Presidential directive in 1987, EPA should use the exposure-to-dose conversion factors tabulated in Federal Guidance Reports Nos. 11 and 12 and their subsequent revisions. This guidance is based on a "linear extrapolation to zero" exposure-to-dose relationship from observed, but much higher, dose-effect studies. As noted in the TSD, the scientific community has been unable to come to a consensus on issues such as the possibility of threshold doses below which no effects occur, the validity of extrapolating curves from known high exposure effects to zero, and the possibility of hormesis (the concept that small doses of radiation may be beneficial to humans). The Subcommittee recommends that the uncertainties associated with extending risk analyses to very low radiation exposures in the absence of scientific consensus be reflected in the presentation of the final results.

6.6.e ***Recommendation.*** The TSD is not entirely clear on the distinction between the assumed exposure periods for the various scenarios and the period over which the dose commitment should be calculated. The understanding of the Subcommittee is that the TSD assumes exposure periods of 30 years for the on-site residential scenario and 25 years for the commercial/industrial scenario, with dose commitments of 70 years for a person who begins residential exposure at birth vs. 50 years for a person who begins occupational exposure at age 20. The document should include in one place a clear and explicit explanation of these distinctions and the rationale for choosing the values for exposure time and commitment period.

6.6.f ***Recommendation.*** At a minimum, EPA should explain the variability in the risk per mrem results shown in TSD Tables 3-1 to 3-3 when discussing its overall approach to dose and risk conversions; the Subcommittee recommends that EPA remove the risk per mrem column from the tables so that the casual user will not be confused. Ideally, EPA should fix on a method for converting intakes to risks and use it regardless of whether the standard will be based on risk or dose.

6.6.g ***Recommendation*****.** Several risk metrics are used in the TSD to describe health risks associated with radionuclides in soils. These include cancer fatalities, cancer incidences, individual and population doses. It is recommended that these metrics be fully described and used in an internally consistent manner throughout the TSD.

6.6.h ***Recommendation.*** A comparison was made of EPA and NRC estimates of contaminated soil volumes and cancer risks associated with that soil, for commercial nuclear power reactors, research reactors, rare earth extraction facilities, and uranium fuel fabrication facilities. The EPA estimates of soil volume were significantly larger than NRC estimates, by factors up to 100. Nonetheless, for three of these four types of facilities, the estimates of the number of fatalities associated with contaminated soil were generally comparable, within a factor of ten of each other, thereby confirming that the TSD results based on the available site data are not unreasonable. The exception was the case of the uranium fuel fabrication facility, for which mortality estimates in the TSD were higher than those calculated by the NRC by factors ranging from 2.5 to 100. The reasons for this difference are not clear and should be investigated by the EPA.

7. SENSITIVITY AND UNCERTAINTY ANALYSES

7.1 Overview of EPA Approach

Given that policy decisions are to be made using the numerical results of the TSD models as one criterion, it is critical that the quantitative uncertainties about those results be disclosed and emphasized in the presentation. In support of this objective, EPA's Technical Support Document includes

a) a quantitative sensitivity analysis for selected assumptions and parameters using the generic site characteristics as a touchstone,
b) a quantitative uncertainty analysis of the risk factors for the generic site,
c) a quantitative sensitivity analysis of the results for the reference sites for four major assumptions, and
d) a qualitative discussion of other uncertainties in the reference site analyses.

The TSD does not attempt a quantitative uncertainty analysis in the sense of portraying the confidence with which risks, soil volumes, and cancers averted can be stated to be less or greater than various possible values. Consequently, while EPA states that volumes of soil requiring remediation at various target risk levels and population impacts in terms of cancers averted are probably both overestimated due to conservative assumptions in the analysis, the degree of potential overestimation cannot be assessed very well with the information provided. This characteristic of the assessment is important because the selection of a target cleanup risk level will presumably be based in part on a balancing of the benefits (cancers averted) and costs (soil volume requiring remediation). When both are estimated conservatively, but with a degree of conservatism that is not quantified, the actual balance achieved with a given target cleanup level may differ substantially from that estimated. Recognizing this limitation, the sensitivity and uncertainty analyses conducted by EPA for this study do provide a number of useful insights. The remainder of this section thus focuses on the strengths and weaknesses of the analyses that were conducted.

Chapter 3 of the TSD includes a number of sensitivity studies of the analysis used to determine soil concentration standards at the generic sites, including intercomparison of predictions for three models (RESRAD, RAGS/HHEM, and PRESTO), a simple parametric sensitivity analysis (varying parameters one at a time) for the risk factors generated with RESRAD, a hypothetical uncertainty analysis for the RME risks at the generic sites (the uncertainty analysis is perforce hypothetical because the sites themselves are hypothetical), and evaluation of risk factor predictions (which should ostensibly be the same) made by five different, independent modelers. The results of the latter comparison indicated four modelers with similar results (within a factor of four) and one outlier due to some fundamental errors in the application. The quantitative results are accompanied by a thoughtful qualitative discussion of the fundamental model assumptions that result in the different predictions (although it does not address processes omitted from all of the models, such as those identified for groundwater in section 5.6 of the Subcommittee's report) including tradeoffs among different pathways (e.g., more mobile radionuclides result in less direct exposure, but more groundwater exposure). This discussion helps provide insight into the important assumptions underlying the models and the relative importance of the different routes of exposure.

In TSD Chapter 6, sensitivity and uncertainty analyses are conducted on the expected costs (soil volumes remediated) and benefits (fatal cancer cases avoided) of the proposed regulation, based on analysis of the reference sites. The results are relatively sensitive to the alternative risk level chosen for the standard (from 10^{-6} to 10^{-2} RME risk), insensitive to the time horizon of the analysis (from 100 to 10,000 years), moderately sensitive to the land use assumed for future use of the reference sites (rural residential land use results in factors of 2 to 5 higher fatal cancer cases avoided than does commercial land use), and insensitive to whether indoor radon is considered in the analysis. The analysis is informative and helps to bound the range of impacts that can be expected, albeit given the highly idealized conceptual representations for the physical and regulatory world. Again, the quantitative uncertainty bands are accompanied by thoughtful qualitative discussion of the key assumptions and limitations of the analysis.

7.2 Sensitivity Analysis for the Generic Site

The sensitivity analysis of the generic site included one-by-one consideration of the following assumptions and parameter values:

a) choice of overall risk assessment model,
b) dimensions of the contaminated soil region (area, long dimension, depth),
c) infiltration rate for precipitation,
d) distribution coefficients for radionuclides (soil vs. water), and
e) thickness of the unsaturated zone

The sensitivity analysis with respect to the model used was quite informative in that it demonstrated that significant differences did exist among the models, but that the differences were generally much less for the radionuclides that appear to dominate the overall need for cleanup. In some respects, this conclusion is weakened in that all the models chosen were very similar with respect to the treatment of the key radionuclides and pathways, but it is consistent with the idea that we know somewhat more about the important pathways and that knowledge is captured acceptably well in all the models.

The sensitivity analyses with respect to the parameter values used was well documented and effectively demonstrates the sensitivity of the RESRAD model to these parameters for the touchstone conditions assumed for the generic site. However, the generic site was designed to have a high potential for the groundwater pathway to be important. The sensitivity of the results to changes in single parameter values therefore might have been different if a site less subject to groundwater transport had been selected. Because many of the reference sites do not appear to be dominated by groundwater concerns, the conclusions about sensitivity from the analysis of the generic site may not be robust for the analysis as a whole. The parameters chosen for the sensitivity analysis are called "key parameters." The EPA should provide in the TSD a short explanation about the reasons that these particular parameters were selected for the sensitivity analysis.

The sensitivity analysis varied a single parameter over a predetermined range while keeping the others at the default values (TSD Tables 3-12 and 3-13). The Agency was correct in varying each parameter across the whole range of values comprised by the 16 reference sites (except for the contaminated zone area, where the value ranged from 30 times lower than the minimum value of the reference sites up to a value that approximates the median value for the area of the reference sites). However, the results of a sensitivity analysis will depend to a great extent upon the nominal or default values adopted for the model parameters that are not varied and upon the structural form of the equations used to represent actual processes. RESRAD provides only approximations of the real processes. Therefore, the applicability of the sensitivity analysis is limited. Furthermore, a similar analysis performed with the parameters in the PRESTO or RAGS/HHEM code may show different results.

Finally, the parameter values analyzed in the generic sensitivity analysis are limited to those of interest to a groundwater modeler, even though they do affect the results for other pathways. For example, for ^{137}Cs, parameters affecting the transfer from soil to terrestrial and aquatic foodstuffs should be more important than the hydrologic parameters that were varied in the analysis. However, parameters that affect soil ingestion and resuspension rates (such as vegetative cover or particle size distribution) were not investigated. The sensitivity to assumptions regarding dust loading in the atmosphere is not treated in the same fashion as the groundwater parameters. Factors that would affect the transport of radon are not systematically investigated. While the EPA correctly states that many of these sensitivities are linear in the choice of parameter values, an equivalent level of presentation for them would enhance the utility of the report. It may be prudent for the EPA to conduct a sensitivity analysis for a generic site with characteristics similar to those of the reference sites with the greatest risk per unit soil concentration.

The sensitivity analysis for contaminated zone thickness showed a significant anomaly which should be further investigated. The ratio of calculated soil concentration to base case soil concentration at a 1 x 10^{-4} risk level, as a function of contaminated zone thickness (Table 3-16, p. 3-58 of the TSD), shows a value two orders of magnitude higher for ^{230}Th (5183 for a depth of 2 cm) than for any other nuclide. This indicates that the risk from ^{230}Th in the first 2 cm of soil is a factor of 50 less than the risk from the same total activity distributed through 2 meters of soil. This anomaly may be due to the ingrowth of ^{226}Ra and ^{222}Rn over time because 70% of the risk from ^{230}Th is due to radon inhalation or direct radiation from the short-lived radon daughters. The tables show that the time period of maximum risk increases with increase in depth of contamination. However, the reasons for such a drastic difference in risk for the same total radioactivity concentration are not obvious and should be explained.

The text describing the effect of contaminated zone thickness indicates that, at the 10^{-4} risk level, the "RSC {Radionuclide Soil Concentration} will increase by a factor of approximately 100 when the thickness decreases by a factor of 100 from the base case (i.e., from 2 to 0.02 m)." This statement is misleading because, in fact, if ^{230}Th is taken out of the average, the RSC increase is a factor of approximately 16. For a ten-fold decrease in thickness, the RSC increase is 3.5, not 6.11, if ^{230}Th is deleted. The inclusion of ^{230}Th distorts the effect of increasing contaminated zone thickness on risk from other nuclides.

7.3 Uncertainty Analysis for the Generic Site

The uncertainty analysis of the risk factors for the generic site was conducted by the Monte Carlo method, the type of method that might be appropriate for the whole analysis if resource limitations were not important. However, it was conducted with the RAGS/HHEM model, which makes its findings difficult to interpret in relation to the sensitivity analyses conducted with the RESRAD model. The choice is understandable in that RAGS/HHEM is implemented by a spreadsheet computer program, which is especially convenient for off-the-shelf Monte Carlo simulation programs, and it does provide some valuable insights. The Subcommittee notes that a Monte-Carlo version of RESRAD has been developed by Argonne National Laboratory and has been applied in international model testing programs.

The parameters evaluated statistically in the TSD were mostly those that relate media concentrations to human exposures: water and foodstuff ingestion rates, daily time of exposure to the site conditions, gamma shielding factor, and so on. The parameters evaluated in the generic sensitivity analysis were almost always kept constant at their nominal (generic site) values for the Monte Carlo analysis. Certain "policy" assumptions, such as the duration of residency in the same location, were also maintained at their nominal values. The cancer slope factors for both isotopes of radon were varied, but those for other radionuclides were not. Because many of the parameters were kept constant at the value for the generic site, the conclusions of the analysis suffer from the same doubts as for the generic sensitivity analysis. EPA states that the Monte Carlo type of analysis is more appropriate for a specific real site than for a regulatory analysis and presents the results only as a "proof of concept." It does not rely on the results for any conclusions regarding the robustness of the analysis of the reference sites. The Subcommittee believes that the choice of parameters for which distributions were used was limited and therefore limits the interpretation of the results. This problem is common in Monte Carlo assessments and can give the erroneous impression that the confidence limits are well understood, when in fact model uncertainties are not incorporated to any significant degree.

7.4 Sensitivity Analysis for Reference Sites

The sensitivity analysis for the reference sites analyzes the sensitivity of the results for soil volume and cancers averted to the following choices:

a) the target cleanup level (10^{-6} to 10^{-2} RME individual risk),
b) the time horizon (100 to 10,000 years),
c) future use scenario (rural residential and commercial/industrial, with further consideration of agricultural use and population density); and
d) with and without indoor radon.

All of these choices are beyond scientific resolution and are policy choices to be made by the drafters of the cleanup rule. They therefore are extremely important for understanding the implications of the policy choices and whether or not the choice affects the results greatly or only marginally. In general, the choices appear to be considerably more important to the estimates of cancers averted than they are to the estimates of soil volumes requiring remediation. However, some of the lack of sensitivity of soil volumes is due to the manner in which the analysis was structured. The lack of sensitivity of soil volumes to the time horizon is related to the lack of importance of the groundwater pathway, the only situation in which maximum individual risks would occur later than 100 years in the future. If a substantially different characterization of source terms and site conditions were to arise in later analyses, this conclusion would need to be revisited.

Because the above parameters are not ones that could be subjected to a probabilistic uncertainty analysis, they do not tell us anything about the robustness of the analysis with respect to the assumptions and parameter values chosen for phenomena that demonstrate real variation or are subject to uncertainty about their true but unknown values. That issue is addressed with the qualitative (or perhaps more accurately semi-quantitative) uncertainty analyses presented alongside the sensitivity analyses for each representative site. Those uncertainty analyses include separate overviews of the uncertainties in soil volumes and cancers averted and site-by-site discussion of uncertainties for each reference site.

7.5 Qualitative Uncertainty Analyses

The soil volume uncertainty analyses include a qualitative discussion of uncertainties in site contamination patterns, an "independent" derivation of the risk factors for each site, and a qualitative discussion of the parameter uncertainties in the risk factors. Both qualitative discussions are thoughtful and touch on most of the important uncertainties. They cannot, of course, provide a very good sense of the overall magnitude of the uncertainties in the estimated soil volume needing remediation at the various target cleanup levels, but the overall conclusion that the soil volumes could be significantly overestimated but probably not substantially underestimated seems justified. The discussions lack much detail about how the various types of uncertainty could interact. For example, the uncertainties in the source term could markedly affect the importance of various uncertainties in the risk factors. The "independent" derivation of the risk factors is an important verification that RESRAD was operating approximately as designed. Because the secondary calculations used essentially the same assumptions and parameter values as the RESRAD model, the conformance of results is not fully independent and does not indicate much about the uncertainty of the final estimates.

The cancers averted uncertainty analysis includes a verification of the computer calculations and a qualitative discussion of the uncertainties in the key calculational parameters and assumptions for the population risk analysis. As with the soil volume analysis, the verification step is valuable but is not truly an uncertainty analysis. Again, as with the soil volume analysis, the qualitative discussions of uncertainty are thoughtful and relatively complete. For the cancers averted, the TSD lacks a summary statement about the overall direction and significance of any bias in the assumptions and parameter selections. Such a statement would be valuable.

The Subcommittee notes that any biases in the estimates of individual RME risks associated with specific soil concentrations of radionuclides will tend to be reflected as biases in the estimates of <u>both</u> soil volumes to be remediated <u>as well as</u> cancers averted by soil remediation. For example, if individual risks tend to be overestimated, then the concentrations needed to reduce risks to the target level will be underestimated, and both soil volumes and cancers averted will tend to be overestimated. A statement to that effect is recommended.

7.6 Generic versus Site-Specific Uncertainty Analyses

In many regulatory analyses, generic calculations of risk are best made in the form of screening estimates in order to identify situations in which site-specific risk assessments are warranted. The output of such screening calculations is a bounding estimate of risk that should have a high probability of containing the true, but unknown, risk to a maximally exposed individual. Site-specific assessments then are warranted when generic screening calculations indicate the need for cleanup but the cost of remediation is potentially high. When the cost of cleanup is low, the need for a detailed site-specific analysis should be marginal, at best.

The TSD is different in the sense that both its estimates of cancers averted and its estimates of soil volumes requiring remediation are influenced by the uncertainties in the risks of specified concentrations of radionuclides in soil. The higher the estimate of risk per pCi/g, the more soil must be remediated to achieve a desired RME risk level <u>and</u> the greater number of cancers will be averted by remediating to that level. If the risk managers will use the balance between cost and cancers averted as one measure of the suitability of the various possible RME risk targets, they may be misled by an analysis that consistently overestimates RME risk. Therefore, the meaning of a screening calculation is less clear in this situation than it may be in others.

The Subcommittee understands from oral comments by EPA that site-specific assessments would be performed in order to determine how much cleanup, if any, is required to meet the risk-based goals implicit in the proposed cleanup standard. The Subcommittee supports that policy. Such site-specific assessments should use data from the site to quantify model parameters and include all realistic pathways of exposure. Site-specific assessments that are carried beyond the screening phase should be as realistic as possible and include quantitative estimates of uncertainty for exposure, risk, and the cost of cleanup. The most important contributors to uncertainty should be identified and additional effort spent to reduce the uncertainty in these important components of the calculation of the human health risk. If EPA's generic analyses prove as expected to be conservative in most cases, less soil will need remediation than anticipated in the reference site analyses of the TSD. In some cases, however, EPA's assumptions may prove less than conservative, for example in environments where uptake of radionuclides by crops and livestock is greater than that modeled.

7.7 Findings and Recommendations Concerning Sensitivity and Uncertainty Analyses

7.7.a ***Finding.*** Given that policy decisions are to be made using the numerical results of the TSD models as one criterion, it is critical that the quantitative uncertainties about those results be disclosed and emphasized in the presentation. Overall, the

Subcommittee commends the EPA on conducting sensitivity and uncertainty analyses for the risk and soil volume calculations and on its thoughtful discussions of the important assumptions and parameter value choices. However, because of limitations of the analyses, the reader is left without a sound appreciation for the magnitude of the overall uncertainties in soil volume requiring remediation and cancers averted via remediation. The generic sensitivity and uncertainty analyses are limited by the choice of parameter values that were varied, the need to select nominal values for all other values when varying one parameter, and the lack of analysis regarding model uncertainties. Although quantitative sensitivity analyses of the reference sites are conducted with respect to policy choices such as the target risk level, time horizons, and land-use scenarios, only a qualitative discussion of scientific uncertainties is offered. Thus, the true but unknown values for cancers averted and soil volumes to be remediated at each RME risk level may be quite different from those presented in the report.

The importance of uncertainty analyses and communication of the results of those analyses to environmental risk managers was underscored by the EPA Administrator in a recent memorandum transmitting the EPA Risk Characterization Program to EPA staff (EPA, 1995). Consistent with the spirit of that guidance, the Subcommittee recommends that ORIA improve its risk assessment and characterization in the TSD in the following areas:

7.7.b ***Recommendation.*** Discussion in the TSD should clarify the purpose of the risk calculations: are these to be screening or bounding estimates, high end estimates (e.g., above the 90th or 95th percentile), or central tendency estimates of the true value (risk)? The objective then should guide the selection of appropriate parameter values used in pathway modeling.

7.7.c ***Recommendation.*** EPA should, at a minimum, discuss qualitatively any biases in its estimates of cancers averted as well as biases in the soil volumes to be remediated. The positive correlation of these biases with any in the individual risk estimates should also be mentioned.

7.7.d ***Recommendation***. To the extent possible, EPA should provide best estimates of cancers averted (total and fatal) and soil volumes to be remediated for the various proposed standards, as well as uncertainty ranges about each of those estimates, in addition to the nominal values currently provided. For example, such statements could be of the form: “The number of fatal cancers averted is likely to fall between *x* and *y*. EPA's conservative nominal estimate is *n* fatal cancers averted, while its central estimate is *z* fatal cancers averted."

7.7.e ***Recommendation.*** In the future, when EPA evaluates the need for cleanup at a specific site in response to the final cleanup standard, it should not only allow but also explicitly encourage quantitative uncertainty analyses in the site assessment. The level of detail in such uncertainty analyses should be commensurate with the stakes (potential for risk reduction and cleanup costs) revealed in a screening analysis. Such an approach is consistent with current industry standards such as the Risk-Based Corrective Action (RBCA) guidance in ASTM ES380-94.

7.7.f ***Recommendation.*** The TSD lists the calculated clean-up volumes required to achieve specific risk levels from 10^{-6} to 10^{-2} in Table 6-7 of the report (TSD, p. 6-74). It would be useful to have the collective dose or risk averted at each clean-up level also provided in the main part of the document. This information is available in the TSD Appendices but should be given greater emphasis.

8. ISSUES AT THE SCIENCE/POLICY INTERFACE

The Subcommittee's charge was to review scientific and technical aspects of the methodology used by the Agency to model radiation risks to individuals at radioactively contaminated sites. The results of the risk assessments will be used by the Agency in making policy decisions on cleanup levels for soil. The review by the Subcommittee primarily focused on the evaluation of source terms, environmental transport, and estimation of risks. However, in the course of this work, some issues were identified that were outside the scope of the Subcommittee's charge. The following comments on these issues, which involve the interface of science and policy, are provided for consideration by the Agency.

8.1 Radioactive Site Decontamination and Remediation Issues Excluded from the EPA Analysis

One major concern is the separation presented in the TSD of (a) soil cleanup from (b) cleanup of structures, (c) cleanup of aquifers, (d) disposal of radioactive waste, and (e) recycle and reuse of materials and equipment after cleanup. This distinction is necessary for focusing on the various topics, and simultaneous treatment of all five topics may be beyond staffing limits of the Agency. Nevertheless, interactions among these five activities are expected to have a substantial effect on the estimated costs and benefits of the proposed standard, and hence should be discussed in the TSD, at least in a qualitative sense. The amounts of soil to be remediated, for example, will depend on the extent to which structures and waste areas are also remediated for public access or retained as controlled areas. At sites where cleanup for complete public access seems unreasonable, it may be relatively simple either to move soil to the restricted area or to extend the fence to include the more highly contaminated soil. Similar suggestions apply to locations where groundwater contamination will prohibit water use for the indefinite future.

Another consequence of segregating the contaminated features of a site into discrete categories to be treated under separate regulations is the added cost and implications which a facility would have in complying with the regulations and these measures are likely to be vastly inefficient and unduly expensive if these regulatory actions are separated from one another. For example, if one area which is to remain restricted (e.g., a surface waste repository) is next to a site to be cleaned up for unrestricted public use, it may be most cost-effective to include the more radioactive portion of soil with the restricted area. Other potential areas of regulatory ambiguity include contaminated pipes beneath structures and contaminated groundwater passing underneath soil to be removed for decontamination.

8.2 Other Soil Cleanup Issues Excluded from the EPA Analysis

The Subcommittee notes that a full risk characterization for the proposed radionuclide soil cleanup standards will include not only human cancer risks, but also other potential health impacts, ecological risks, risks to natural resources, and risks to the economic and social welfare of affected individuals and communities.[8] This will require evaluations that go well beyond the individual and population cancer risk calculations which form the basis for the report, including estimates of the distribution of risk to different subpopulations and communities (with possible implications for environmental equity), and careful review of stakeholder concerns and priorities. The Subcommittee recognizes that such studies are being conducted in parallel with the Technical Support document reviewed by the Subcommittee, and that the Agency does have plans to synthesize these studies. The Subcommittee strongly encourages these synthesis efforts, so that no-one outside the Agency develops the mistaken impression that the cancer risk/soil volume tradeoff calculations presented in this report are the sole driving factor for development and evaluation of the proposed standards.

The TSD document examines only the extent of soil cleanup and the volumes of soil removed. To evaluate fully these aspects, the EPA must consider the costs of soil removal, the non-radioactive adverse aspects, and the disposal requirement for removed soil. The costs include soil removal, containment, transportation, storage, and disposal, as well as monitoring to control removal and assure compliance with transportation and disposal regulations, and landscaping the cleared areas. Adverse impacts (beyond radiation exposure of workers and the public due to cleanup, which has been considered in the TSD) are labor and transportation accidents and incidents with associated injury and death, damage to the ecosystem due to soil removal, and the withdrawal of sites from other uses for soil storage and disposal.

Additional consideration must be given to comparisons with the no-action alternative and with in-situ soil remediation. The former consists generally of maintaining controlled areas with the associated costs of maintenance and security and of radiation doses to on-site workers and to the public due to effluents. In-situ remediation can be achieved by various forms of solidification or by covering the soil surface to limit infiltration, radioactive particle suspension, and direct radiation exposure.

Although direct radiation exposures to cleanup workers are estimated, it is recognized that other worker risks exist. Accidents, exposures to nonradiological hazards, mixed waste issues

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[8] *Vis-a-vis* ecological considerations, ORIA should take into consideration the fact that other ecosystem components and non-human animal species could have utility for exposure assessment and development of the cleanup standard. Other species could serve as useful biomonitors and, in some cases, might serve as the focus for development of the site standard.

and criticality incidents are of concern and probably represent collectively a greater risk to the worker than direct radiation exposures.

8.3 Choice of Risk Metric

The importance of individual risk versus population risk is a philosophical and policy question, not a technical matter, and the Subcommittee is not asking EPA to reconsider its approach. The following discussion is intended to clarify the distinction between the two metrics. Our comments are in support of the policy issued recently by the EPA Administrator on risk characterization (EPA, 1995), which called for both estimates of individual risk and population risk. Although the decision as to which of these should form the basis of a regulation is clearly a risk management decision, the new policy states that both types of risk estimates must be presented.

The TSD reports population risks (which EPA will use to estimate the benefits of the standard in terms of cancers avoided) as well as risks to a hypothetical person with reasonable maximum exposure (RME, which EPA will use to define the standard). The lower EPA sets the risk-based standard for an individual, the more cancers can be avoided, but the benefits of that more stringent standard depend on how many people will incur lower doses and risks as a result of the standard. In the TSD, cases are analyzed for individual risk limits ranging from one in a million (10^{-6}) to one in a hundred (10^{-2}). EPA has generally assumed that risks greater than one in a hundred will be abated regardless of the soil cleanup standard EPA eventually sets. Not evident in the TSD is whether or not EPA will set a maximum risk level regardless of the number of cancers avoided at that level. (The option to regulate based on an annual dose to the RME person is in principle nearly the same as setting an individual risk limitation.)

The Subcommittee notes that EPA could have chosen to base its regulation on some other risk metric, for example on the residual population risk after cleanup for any one facility. Under such an option, cleanup would be considered adequate if the facility-wide population risk did not exceed a certain number (e.g., 10 cancers in 1,000 years), regardless of how high the individual RME risk was calculated to be.

Population density has traditionally been a consideration in siting facilities for such potentially hazardous activities as radioactive waste disposal. When population density is low, the population risk for a given pattern of exposure to the hazards of the facility is minimized. At contaminated sites, however, the population is already at risk and the purpose of the cleanup is to reduce that risk. Here the question becomes whether it is sufficient to establish an acceptable level of RME risk and set the standard accordingly, or whether EPA needs to keep population risk below a fixed level as well. If the standard is based on individual risk, then arguments about future growth of population on or near the facility are less important. However, a linear no-threshold dose-response relationship implies that population risks might be limiting in terms of public health detriment even when individual radiation exposures are small fractions of the variability in natural background exposures.

A more stringent standard involves moving greater volumes of material, which in turn has environmental impacts and increases the risks of radiation exposures and conventional accidents to remediation workers. EPA has assessed the radiation risks to such workers and has concluded that the cumulative cancer impacts attributable to any reasonable cleanup standard are very small compared to the potential cumulative impacts averted in post-cleanup, on-site populations. That analysis is based on aggregate person-rem calculations, which relate to population risk, not to individual risks. The maximum level of individual worker risk is not stated, but could be higher than the individual risk level used for the standard.[9] While the Subcommittee does not assert that such decisions are inappropriate, some mention of this difference in risk management philosophy may be worth discussion. EPA is apparently also conducting an analysis of the risks from conventional accidents, but the Subcommittee has not reviewed that analysis.

8.4 Time Horizons

As is well-recognized by the EPA, extrapolations in risk assessment are necessary procedures that are fraught with inherent variability and uncertainty (EPA, 1995). In the case of temporal extrapolations with long-lived radioactive contaminants, however, the extrapolations carry the added uncertainties associated with an inability to factor in social, economic, and scientific changes that are bound to occur in the long run. While these added uncertainties cannot be assessed effectively either qualitatively or quantitatively, the risk manager should be made aware of their existence and their implications when reaching a decision.

EPA conducted analyses of the number of cancers averted for three time periods of exposure: 100, 1,000, and 10,000 years. The number of cancers calculated to be averted will be one of the factors for consideration in policy decisions on a standard, and this number increases as the time period of the exposure increases. The following comments are offered with respect to the time period over which to consider exposure.

When the standard is implemented, large expenditures of present-day dollars will be required to reduce predicted cancers over some future time period. As noted in the EPA's

[9] The maximum risk to individual workers is not presented in the TSD because it is not considered to be pertinent to the assessment. Workers will be covered under a radiation protection program which limits their exposures to the occupational radiation protection standard, which is considerably higher than the 15 mrem/yr level considered in the Rule for post-cleanup on-site populations of residents or of commercial/industrial (but non-radiation) workers (A. Wolbarst, memo to J. Watson dated May 18, 1995).

document, there is uncertainty in predicting the number of cancers averted, and, at low doses, it is also noted that the possibility of zero risk can not be ruled out. This uncertainty may be reduced in the future. Extrapolating today's medical, industrial, and scientific technology into the future adds further uncertainty. For example, cancer was not a major health concern in the 1800's, principally because other causes of death were so prevalent. Will it still be a major concern 100, 1,000, or 10,000 years in the future? It is possible that medical researchers will discover how to prevent or cure cancer within the next 100 years. Also, it is possible that better, more efficient cleanup methods than are currently available will be discovered in the future. On the other hand, it is conceivable that society may slip backwards, in terms of its ability to deal with cancer as a health problem. For all these reasons, as the exposure time period increases, the uncertainty in the number of cancers averted increases.

By estimating risks (cancers avertable) over time horizons of 100, 1,000, and 10,000 years, EPA has also provided the cleanup standard decision-makers with a range of options for evaluating the benefits of the standard. However, the Subcommittee is concerned that risks incurred more than 100 years in the future are highly speculative given the uncertainty about the rate and direction of change in medical science and other technological and social arenas, which could either reduce or accentuate the relative importance of cancer as a health risk in the future. The Subcommittee therefore recommends that, in transmitting its findings to the EPA group responsible for proposing the cleanup standard, the technical support group should emphasize the additional uncertainty of its estimates for 1,000 and 10,000 years in comparison with those for 100 years.

8.5 Choice of Target Risk Levels

The analysis in the TSD includes calculations of the risks and soil cleanup volumes associated with 10^{-2} to 10^{-6} risk levels. The analyses of risk associated with ^{226}Ra suggest that cleanup levels below 10^{-3} are probably not feasible because of the natural variation in the abundance of this isotope (see p. 6-20 of the TSD). For comparison, the cleanup standard for ^{226}Ra at uranium mill tailings sites is 5 pCi/g above background levels in the top 15 cm of soil and 15 pCi/g above background in deeper layers (40 CFR 192), corresponding to lifetime risks of approximately 10^{-2} (based upon extrapolation of results in TSD Table 3-1, p. 3-4). This illustrates a lack of consistency between the lower risk levels being considered in the TSD and those corresponding to existing regulations dealing with radionuclide cleanup, such as the Uranium Mill Tailings Radiation Control Act (UMTRCA) regulation (40 CFR 192).

In some cases, large variations in the local background concentrations of naturally-occurring radionuclides such as potassium, radium, thorium and uranium make it difficult to identify the presence of residual radioactive material from natural sources. Even on individual residential properties, radium concentrations can be several times higher than the concentration value equivalent to 15 mrem/yr or 10^{-4} risk-based limits. Areas which have been used for disposal of furnace or fireplace ash can contain several pCi/g of radium while other locations on the property have background concentrations of 1 pCi/g or less. In another example, survey teams in Wayne, New Jersey (another thorium processing facility near the Maywood site), found road-bed asphalt/ballast material and rock gravel used in landscaping to have above-normal gamma dose rates. The rock material came from a local quarry and had uranium and thorium concentrations of 3 and 1.7 pCi/g, respectively, and exposure rates of 60 to 80 µR/hr (DOE, 1995)[10]. Identifying concentrations of 0.08 or 0.4 pCi/g above such concentrations would be problematic. Some of the landscaping rocks had natural concentrations that were higher by an order of magnitude. If not appropriately qualified, a 10^{-4} risk-based standard could require cleanup of typical residential material (as noted above) or enhanced monitoring costs to differentiate residual and naturally-occurring materials. Other sources of natural interference include certain slag refuse from metal production, phosphogypsum (fertilizer), certain ceramic building materials, and cinder blocks. It is questionable whether concentrations that are a fraction of background have sufficient risk associated with them to address. Furthermore, it is not clear, given that the variations in background could exceed these levels, how one can be certain that the "clean fill" which could have elevated natural background is not increasing risk over and above those being mitigated by removal of the contaminated material.

A related concern has to do with the technical feasibility of implementation of the proposed standard (p. 7-14 of the TSD). With regard to measurements, EPA concluded that a combination of laboratory analyses and field monitoring could detect all radionuclides at concentrations corresponding to 15 mrem/yr (i.e., 3×10^{-4} risk limits). For some radionuclides, however, only the more costly laboratory analyses have the requisite low detection limit. Costs of monitoring have been considered a valid consideration in establishing concentration limits for other regulatory activities, such as regulations promulgating the Safe Drinking Water Act.

8.6 Potential Application to NORM

In comments to the Subcommittee, members of the public have expressed concern that criteria in the site cleanup regulations being developed might be applied to sources of Naturally Occurring Radioactive Material (NORM). The Subcommittee is not offering an opinion on the advisability of using the proposed cleanup standard as an ARAR (Applicable or Relevant and Appropriate Regulation) or as the precedent for any NORM regulations that might be proposed in the future. However, these possibilities should be noted in the TSD, and the costs and benefits of

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[10] Regarding the DOE reported exposure rates, the Subcommittee observes that these appear questionable for the soil concentrations given. It should not be construed by the reader that the Subcommittee endorses these numbers.

any such actions should be discussed in the Regulatory Impact Analysis. The Subcommittee notes that the TSD does not provide the technical basis for cleanup criteria for NORM because the analyses presented in the TSD do not address cancers averted and volumes of soil affected for sources of NORM. Also, the feasibility issue noted in the preceding section would be pertinent because ^{226}Ra is one of the principal NORM radionuclides.

The cost/benefit analysis of the proposed regulation would be enhanced if results with and without NORM sites were presented. Because the source term for NORM is even less well understood than for the reference sites already analyzed, this may be an unrealistic suggestion. EPA could, however, discuss qualitatively how the results might change if the standard were to be applied to NORM sites.

8.7 Consistency Between Model Assumptions and Field Sampling Methods

The leachate equation used in the TSD to relate the groundwater concentration (C_w) of a radionuclide to the soil concentration (C_s) assumes that the soil concentration is determined by taking the total mass of radionuclide in the original sample (soil + interstitial water) and dividing by the dry weight of the soil (see section 5.6.3 of this report). Because soil concentrations can be measured in a variety of ways, it is important that the model assumptions in the TSD be consistent with the methods specified for measuring soil radionuclide concentrations in the regulations. This is an important issue affecting all soil cleanup standards, and it is important that a consistent approach be taken both within the EPA and across Federal Agencies. The TSD should include some discussion of standard and approved methods for measuring radionuclide concentrations in soils and how these relate to the assumptions of the modeling exercises in the TSD.

8.8 Findings and Recommendations

8.8.a ***Recommendation***. Because the TSD presented the issue of soil cleanup separately from related cleanup problems, the Subcommittee had difficulty in understanding how the Agency intended to take into account the interrelationships among its proposed regulatory actions to deal with soil cleanup, aquifer cleanup, cleanup of structures, cleanup of aquifers, disposal of radioactive waste generated by the cleanup, and recycle and reuse of materials and equipment after cleanup. Interactions among these five activities are expected to have a substantial effect on the estimated costs and benefits of the proposed standard, and hence should be discussed in the TSD, at least in a qualitative sense. The amounts of soil to be remediated, for example, will depend on the extent to which structures and waste areas are remediated for public access or retained as controlled areas. While recognizing that the Agency is currently constrained by statutes or regulations to address these aspects as isolated problems, in truth, these problems are integrated, not isolated. The Subcommittee feels strongly that the current fragmentation of regulatory impact analyses leads to unrealistic estimates of risks and benefits. More scientifically robust estimates of overall exposure and risk would be derived from an integrated analysis.

8.8.b ***Finding.*** An estimate of the volume of soil to be remediated to achieve a specified level of acceptable risk and dose is only one aspect contributing to the technical and scientific judgment for site cleanup. Other pertinent aspects include soil removal costs, adverse impacts of removal (including non-radioactive impacts), availability of repositories for the contaminated soil, and the feasibility of remedial alternatives such as soil stabilization, soil covers, and no action (continued control and surveillance). The Subcommittee strongly endorses EPA's intentions to include these aspects in the Regulatory Impact Analysis.

8.8.c ***Finding***. EPA's decision to use lifetime risk corresponding to reasonable maximum exposure as a risk metric for the proposed standard is appropriate. Although another metric (such as the population risk attributable to a facility) could have been used, the formulation of the standard is consistent with other EPA waste management strategies. In its comparison of the risks of remedial activities to those avoided through remedial action, EPA uses population risk (cancer fatalities caused vs. cancer fatalities averted). While this choice is also reasonable, it bypasses the issue of potentially higher individual risks to some remediation workers.

8.8.d ***Recommendation.*** EPA should be more explicit about the reasons it chose to use an individual risk or dose metric for the standard and a population risk metric for describing the costs and benefits of the standard. A brief mention of the potential for higher-than-average individual risks to remediation workers is also recommended.

8.8.e ***Recommendation***. By estimating risks (cancers avertable) over time horizons of 100, 1,000, and 10,000 years, EPA has also provided the cleanup standard decision-makers with a range of options for evaluating the benefits of the standard. However, the Subcommittee is concerned that risks incurred more than 100 years in the future are highly speculative given the uncertainties about the rate and direction of change in medical science and other technological and social arenas, which could either reduce or accentuate the relative importance of cancer as a health risk in the future. The Subcommittee therefore recommends that EPA should emphasize the additional uncertainty of its estimates for 1,000 and 10,000 years in comparison with those for 100 years.

8.8.f ***Finding.*** The analyses of risk associated with ^{226}Ra suggest that cleanup levels below 10^{-3} are probably not feasible because of the natural variation in the abundance of this isotope. For comparison, the cleanup standard for ^{226}Ra at uranium mill tailings sites is 5 pCi/g above background in the top 15 cm of soil and 15 pCi/g above background in deeper layers (UMTRCA regulation, 40 CFR 192), corresponding to lifetime risks of approximately 10^{-2}. This illustrates a lack of consistency between the lower risk levels being considered in the TSD and those corresponding to existing regulations dealing with radionuclide cleanup, such as the UMTRCA regulation.

8.8.g ***Recommendation***. The Subcommittee is not offering an opinion on the advisability of using the proposed cleanup standard as an ARAR (Applicable or Relevant and Appropriate Regulation) or as the precedent for any NORM regulations that might be proposed in the future. However, these possibilities should be noted in the TSD, and the costs and benefits of any such actions should be discussed in the Regulatory Impact Analysis. The Subcommittee notes that the TSD does not provide the technical basis for cleanup criteria for Naturally Occurring Radioactive Material (NORM) because the analyses presented in the TSD do not address cancers averted and volumes of soil affected for sources of NORM. Also, the feasibility issue noted in 8.8.e would be pertinent since ^{226}Ra is one of the principal NORM nuclides.

8.8.h ***Recommendation.*** ORIA should take into consideration the fact that other ecosystem components and non-human animal species could have utility for exposure assessment and development of the cleanup standard. For example, other species could serve as useful biomonitors.

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APPENDIX A -COMPARISON OF SOURCE TERM INFORMATION DEVELOPED BY THE NRC AND BY THE EPA FOR REFERENCE NUCLEAR FACILITIES

1. INTRODUCTION

The Nuclear Regulatory Commission (NRC) and the Environmental Protection Agency (EPA) have both conducted analyses of the impacts of various levels of cleanup of radionuclide contamination at nuclear facilities in support of cleanup standards. The objective of the NRC analyses was to assess the environmental impact of the proposed amendment to the regulations in 10 CFR Part 20 to include radiological criteria for decommissioning of land and structures at nuclear facilities licensed by the NRC [Generic Environmental Impact Statement (GEIS); NRC, 1994]. The objective of the EPA analysis was considerably broader, in that it was to assess the environmental impact of the proposed regulations in 40 CFR Part 196 that would specify radiological criteria for soil, ground water, surface water, and structures at Federal facilities to be released for public use.

The EPA's Technical Support Document (TSD) (EPA, 1994a) and the NRC's GEIS used similar types of information, such as estimated source terms, to derive estimates of impacts, including potential fatal cancers, for various cleanup and occupancy scenarios. Although the methods used by the two Agencies differ significantly, a comparison of the two documents was judged worthwhile in that it could provide a degree of confidence in the TSD estimates to the extent that there were cases for which the results of the two analyses were similar.

The TSD includes 22 reference facilities in the analysis of impacts; the GEIS uses ten reference facilities. Four of the reference facilities are common to both documents. The TSD and GEIS source term assumptions, including radionuclide concentrations in soil and areal extent and volume of soil contamination, are compared in Section B.2 of this Appendix. In Section B.3, the use of the source terms in calculating impacts and the results of the analyses in terms of fatal cancers projected at various residual dose values are examined. Conclusions concerning the comparison of source term parameter values and results of risk calculations are presented in Section B.4.

2. SOURCE TERM INFORMATION AT REFERENCE SITES

Both the GEIS and the TSD address the difficulty of obtaining sufficient contamination data to model the reference sites. The methods and sources used to estimate the soil radionuclide activity concentrations and the areal extent and volume of contaminated soil for the four reference facilities common to both documents are summarized below and in Table B-1.

2.1 Commercial Nuclear Power Reactors (TSD Reference Site XVI)

Soil contamination levels: The level of soil contamination used in the TSD to represent commercial nuclear power reactors is based on NUREG/CR-4289 (Abel et al., 1986), the same reference as is used in the GEIS. In both analyses, Co-60 is considered to be the nuclide of greatest concern. In the analysis, the TSD assumes a peak concentration of Co-60 of 300 pCi/g (TSD, Figure 4-21, p. 4-93). The GEIS gives low, medium and high values for soil contamination for each of the nuclides of concern.[11] The "high" value for Co-60 for power reactors given in the GEIS is 60 pCi/g (GEIS, Appendix C, Table 5.5.1, p. C 5.14)

Areal Extent and Volume: The areal extent and volume of soil contamination used in the TSD is based on information contained in requests to the NRC for on-site disposal of soil at nine reactors, in compliance with 10 CFR 20.302 (TSD, p. 4-92). The on-site disposal volumes for these nine reactors average approximately 580 m^3. The TSD adjusts this value to 1060 m^3 to account for soils which may be contaminated at levels above background but below the NRC release limits for Co-60 and Cs-137. These volumes are translated in Table 4-6 of the TSD (p. 4-27) to a surface area contamination around a reactor facility of 7,000 m^2 or approximately 1.7 acres. The level of contamination is assumed to be uniform and limited to the first 15 cm of soil, the maximum depth of the soil samples (TSD, Table 4-6, p. 4-27). The analysis in the TSD for all reference sites assumes that radionuclide contamination is limited to a specific depth and is uniform to that depth (TSD, p. 4-2).

The GEIS uses an areal extent of contamination of 2,000 ft^2 (186 m^2) (GEIS, Table 4-1, p. 4-14) based on Abel et al. (1986) which indicates that "radionuclide contamination of soils around nuclear power plants was limited to small patches of very low concentrations of radionuclides," and that "the volumes in most cases would be very small, e.g., tens of cubic meters," (Abel et al., (1986) p iv, 25, 26). The depth profile assumed for contamination in soil in the GEIS is based on diffusion theory and the soil models of NUREG/CR-5512 (NRC, 1992). The model, as used in the GEIS, assumes that the contaminant was deposited at the soil surface at time zero and has penetrated the soil for a known period of time. The dispersion of the initial radionuclide inventory is calculated taking into account the K_d for the particular soil and radionuclide and the soil porosity. The assumed contamination depths, comparable for those assumed in the TSD, are not specifically stated in the GEIS but the approximate contamination depths can be inferred from Figures 5.5.1 - 5.5.6 in Appendix C of the GEIS (p. 5.10-5.13). In contrast to the TSD

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[11] Note that the peak levels of contamination, as reported in the TSD, are not directly comparable to the "high" level given in the GEIS. The "high," "medium," and "low" levels given in the GEIS are intended to give some flexibility to the analysis and are average levels over the site. The "peak" levels from the TSD may exist in only a small portion of the site. The NRC values are both alternate estimates of average concentrations, based on the real data available at the time, and some measure of sensitivity. With these few test points, it could not qualify as an analysis.

assumptions, the contamination levels assumed in the GEIS analysis are not constant with depth. Therefore, a direct comparison between the two methods is not feasible.

2.2 Test/Research Reactors (TSD Reference Site XVIII)

Soil contamination levels: The level of radionuclide contamination used in the TSD for the reference case is based on data from the Cintichem reactor, a 5 MWt reactor (TSD, p 4-94). Cs-137 and Sr-90 were assumed to be the nuclides of concern (TSD, p. 4-95). The GEIS uses two separate references cases: (a) the 5 MWt Plum Brook reactor is used as the reference test reactor, and (b) the 1 MWt Oregon State University reactor is used as the reference research reactor, a category that includes a number of research reactors in the kWt range. There are about four test reactors and 59 research reactors in the United States. The TSD analysis assumes a peak concentration of Cs-137 of 10,000 pCi/g (TSD, p 4-94 and Figure 4-22 p. 4-96). The GEIS assumes a "high" general concentration of Cs-137 in soil equal to 20 pCi/g (GEIS, Appendix C, Table 5.5.1, p. C.5.15).

Areal extent and volume: The areal extent and volume of contamination assumed in the TSD analysis is also based on the Cintichem reactor. The TSD acknowledges that the pattern of soil contamination at Cintichem may not be typical of that at a research reactor (TSD, p. 4-94). The area of contamination at the Cintichem reactor is estimated to be 3,300 m^2 (approximately 0.8 acres) around the facility and the soil volume is approximately 500 m^3 (TSD, p. 4-94). The power level of Cintichem is 5 MWt which is larger than nearly all research reactors which are generally TRIGAs (Training, Research, and Instruction -- General Atomics, which is a small reactor manufactured by General Atomics.) or AGNs (Atomic General Nuclear -- another line of small reactors often found in universities.) of power level equal to or less than one MWt (NRC, 1982; Table 3.1-1). NRR (NRR stands for U.S. Nuclear Regulatory Commission, Office of Nuclear Reactor Regulation) input indicates that the Cintichem reactor is the only currently licensed research reactor with significant soil contamination. Experience would indicate that only one other decommissioned research reactor site has had significant soil contamination and that contamination was primarily from byproduct material work that was performed under a separate license at the site.

The GEIS uses a contaminated area of 5,000 ft^2 (465 m^2) for the reference test reactor based on conditions at the Plum Brook reactor (GEIS, Table 4-1, p. 4-14). For the reference research reactor, the GEIS assumes a contaminated area of 500 ft^2 (47 m^2).

2.3 Rare Earth Extraction Facilities (TSD Reference Site XXI)

Soil contamination levels: The level of contamination used in the TSD is based on data from the Molycorp facility at Washington, PA, a rare earth extraction facility listed in the NRC's Site Decommissioning Management Program (SDMP) (NRC, 1993). The radionuclide of concern is Th-232 (and its decay products). The peak soil concentration in the TSD analysis is 300 pCi/g (TSD, Figure 4-24, p. 4-101). The "high" concentration used in the GEIS analysis is 200 pCi/g (GEIS, Appendix C, Table C.5.5.1, p. C.5.15).

Areal extent and volume: The areal extent and volume of contamination used in the TSD analysis is based on the Molycorp site in Washington, PA. The volume of contaminated soil is estimated based on an areal extent of contamination of 13,800 m^2 and a vertical depth of contamination over this area of 2.5 m (TSD, Table 4-6, p. 4-27). The depth of soil contamination over the site is estimated on the basis of borehole information gathered at the site; however, it was noted that the concentration data from this information should be regarded as provisional (TSD, p. 4-102). NRC (1993) notes that operations at Molycorp resulted in thorium slag which was used as fill over portions of the site (NRC, 1993, p. A-65). The GEIS analysis uses an areal extent of contamination of 100,000 ft^2 (9,300 m^2) also based on operations at the Molycorp site (GEIS, Table 4-1, p. 4-14).

2.4 Uranium Fuel Fabrication Facilities (TSD Reference Site XX)

Soil contamination levels: The level of contamination assumed in the TSD is based on data from the Babcock and Wilcox facility at Apollo, PA, a uranium fuel fabrication facility listed in the NRC's SDMP (NRC, 1993). The GEIS uses both the Wilmington NC facility and the Apollo facility. In the analysis, the TSD assumes a peak concentration of approximately 2,000 pCi/g for U-234; 300 pCi/g, U-238; and 70 pCi/g, U-235 (TSD, Figure 4-23, p. 4-98). The GEIS, Appendix C, Table 5.5.1 (p. C.5.15) and Figure 5.5.5 (p. C.5.12) refers only to concentrations of U-235 and shows a "high" value of 1,000 pCi/g.

Areal extent and volume: The areal extent and volume of contaminated soil used in the TSD are based on data from Apollo and on extrapolation of the available data. The amount of contamination at >30 pCi/g was taken from recent Apollo information and is given in Table 4-6 of the TSD (p. 4-27) as approximately 20,000 m^2. Two values are reported for contamination depth in TSD Appendix Table B-1 (p. B-9). A depth of 36 cm was used to calculate population impacts; the more conservative value of 100 cm was used to calculate RME risks (TSD, p. 4-99). This information was extrapolated to estimate the total volume of soil contaminated to levels greater than 1 pCi, approximately 490,000 m^3. It is not clear what the corresponding contaminated area is assumed to be. The GEIS assumes 50,000 ft^2 (4,600 m^2) as the areal extent of contamination based on Wilmington and Apollo information (GEIS, Table 4-1, p. 4-14). As with the other reference site, the depth profile is based on diffusion theory and soil parameters. For this site the depth of contamination is over 35 cm (GEIS, Appendix C, Figure 5.5.5, p. C.5.12).

2.5 Summary

The "peak" and "high" estimated soil concentration levels, the areal extent of contamination, and the estimated depth of contamination for the four common reference sites are summarized in Table B.1. The following observations are made:

a) The estimated levels of contamination at the reference sites common to the EPA and NRC analysis are generally of the same order of magnitude. However, the peak levels of contamination, as given in the TSD, are not directly comparable to the "high" level given in the GEIS. The NRC estimated soil surface contamination on the basis of reports on the level of contamination in nuclear facilities (GEIS, p. 4-11). The “high,” “medium,” and “low” levels used in the GEIS represent average soil concentrations over the reference site and take into account the uncertainties and potential ranges in soil contamination. They are intended to “bound the problem” and allow the range of potential impacts to be estimated (GEIS, p. 5-13). In contrast, the “peak” levels from the TSD may exist in only a small portion of the site. The "peak" levels from the TSD may exist in only a small portion of the site.

b) The bases for estimating the areal extent of contamination are similar for the two sets of analyses. However, the values differ by orders of magnitude in some cases.

c) The methods for estimating depths of contamination are significantly different for the two sets of analyses. The TSD assumes uniform contamination to a specific depth. For the TSD risk analysis, discrete contaminated segments are assumed to be removed to a uniform depth to achieve alternative dose criteria. The GEIS assumes that contamination occurs on the surface and has dispersed vertically through the soil over time according to characteristics of the soil and radionuclide. Dose criteria are achieved by removing layers of contaminated soil. The concentration of radionuclides is not uniform over depth; however, the average concentration of contaminant remaining in the soil as layers are removed was used in the dose calculations.

3. ESTIMATES OF MORTALITY FROM EXPOSURE TO RESIDUAL RADIOACTIVITY IN SOIL AT FOUR REFERENCE SITES

The approach to calculating total cancer mortality for populations residing on sites remediated to a specific residual dose criterion in the GEIS is significantly different from the method used in the TSD. However, a comparison of the results of the two independent analyses, as shown in Tables B-2 to B-5 of this Appendix, may be useful as a check on the reasonableness of the TSD results.

For this analysis, only the residential scenario is considered. The EPA method assumes that the contaminated structures on site are not occupied and that the dose is derived only from contaminated soil. The residential scenario for the NRC method also assumes no occupancy of contaminated buildings.

The NRC analysis does not consider the fatalities averted by actions taken to reduce the dose at the unremediated site to 100 mrem/year because the NRC assumes that 100 mrem/year is the maximum reasonable value for the residual dose criterion. Therefore, the comparison in this appendix only considers the number of cancers averted for residual dose criteria ranging from 100 mrem/year to 0.1 mrem/year.

The EPA method used in the TSD is based on the site being remediated to a residual soil contamination level such that the reasonable maximum exposure (RME) to any individual does not exceed the criterion level. The population doses and risks are then calculated based on the residual soil contamination, population density, and appropriate dose conversion factors. Fatalities averted are calculated based on the estimated number of fatal cancers expected for the unremediated site. The number of population fatalities averted is calculated for RME annual dose levels ranging from 0.1 mrem/year to 100 mrem/year. This information is contained in Appendix M to the TSD.

The NRC method, as described in the GEIS, simply calculates the total number of cancer deaths which would be averted in a population receiving radiation doses at the residual dose criteria levels. The NRC assumes that the criterion would not be set higher than the current 100 mrem/year limit for the general public. Therefore, the fatalities averted in remediating the site to a condition where the annual dose would be 100 mrem, are not estimated. The number of cancer fatalities is directly proportional to the number of individuals exposed. The population exposed for the residential scenario is assumed to be the contaminated site area for the reference facility multiplied by a population density factor of 0.0004 persons/m^2 (400 persons/km^2) (GEIS, Appendix B, p. B-7). The exposure duration is 70 years and the risk conversion factor, 3.92×10^{-4} per rem (GEIS, Appendix B, p. B-7). The following is a sample calculation for the power reactor at the 100 mrem residual dose criterion using the data from Table B-1:

$$\text{Fatalities} = \text{Area} \times 0.0004/\text{m}^2 \times 0.1 \text{ rem/yr} \times 70 \text{ yr} \times 3.92 \times 10^{-4}/\text{rem}$$
$$= 186 \text{ m}^2 \times 0.0004 \text{ m}^{-2} \times 0.1 \text{ rem/yr} \times 70 \text{ yr} \times 3.92 \times 10^{-4}/\text{rem} = 2.04 \times 10^{-4}$$

The EPA method uses a minimum integration period of 100 years for population effects as compared to the NRC method which assumes a 70-year lifetime. The residual dose criteria levels are also somewhat different for the EPA method as compared to the NRC method. In addition, the TSD reports cumulative fatalities averted for cleanup to a dose criterion, whereas the GEIS reports residual expected population cancer fatalities at each residual dose criteria levels.

Comparisons between the number of cancer fatalities expected for onsite residents for the TSD and the GEIS are shown in Tables B-2 through B-5. In order to compare the two analyses, the risk estimates from the TSD on fatal cancers averted were converted to total fatal cancers at the residual dose criteria levels by subtracting the cancers averted at each RME dose level from the number of cancers averted at a dose level of 0.1 mrem/year. In the EPA analysis, the majority of the cancers averted result from remedial actions taken to reduce the RME to 100 mrem/year except for the uranium fuel fabrication reference facility. The differences which result from further reductions are small compared to the total fatalities averted and, for the lower dose criteria, are outside the range of significant figures for total cancers averted. In the EPA analysis, the significant figures reported are not sufficient to calculate the residual cancer burden for the lower dose criteria.

It should be noted that the EPA analysis shows that the great majority of effect, in terms of potential cancers averted, for reactor reference sites occurs due to remediation of the site to an RME level of 100 mrem/year. The effect of reducing the RME below 100 mrem per year is much smaller in comparison. In contrast, for the uranium fabrication facility reference site, there is a reduction in projected total fatalities by a factor of 25 in remediating the site from a residual RME dose level of 100 mrem/year to 0.1 mrem/year (Table B-5).

Data from TSD Appendix Tables M-89 and M-90 (TSD, p. M-89 and M-90) were used in the comparison. These tables list calculated potential cancer deaths averted, including radon in the analysis, for the "Reasonable Occupancy Scenario." Data for the NRC analyses were obtained from GEIS Tables 5-1, 5-2, 5-3, 5-4 and 5-10 (GEIS, p. 5-20 to 5-25).

4. CONCLUSIONS

The following observations are made from the comparison of the EPA and NRC risk analyses:

a) The <u>volumes and aerial extent of contaminated soil</u> assumed by EPA in the TSD are generally greater than those assumed by the NRC in the GEIS.

b) The estimated <u>peak soil concentrations</u> reported in the TSD are higher than the "high" average concentrations used in the GEIS, as would be expected.

c) The estimated <u>number of fatalities averted</u> for specific dose criteria are comparable. The NRC (GEIS) method simply multiplies the residual dose criterion level by the population density and a dose conversion factor. In contrast, the residual dose criteria for the EPA analysis in the TSD were based on the dose derived for the RME. Therefore, it is not a simple conversion from the dose criterion level to potential cancer deaths. The RME doses were calculated using RESRAD, while the population risks were calculated using CU-POP. In addition, the integration time for population exposure in the TSD is a minimum of 100 years while the GEIS uses the average lifetime, 70 years, for the calculation of potential fatalities.

While the TSD assumes a much greater contaminated soil area for the commercial nuclear power reactor reference site, the calculated number onsite resident fatalities is lower than that reported in the GEIS for a similar reference site. The reason for this difference is not immediately apparent but may be due to the fact that the critical nuclide for this reference site is Co-60. The NRC method does not allow for decay whereas the EPA method does. (NRC-1994A, Vol. 1, page 5-4, column 1, last full sentence, which states... "Radiological decay over this period is not included in the analysis of these impacts." Because Cs-137 was assumed to be much more mobile than Co-60 in both soil and concrete, removal of the first layers removed virtually all of the Co-60 leaving Cs-137 to deliver the remaining dose. The resulting error for the assumption over a 70-year residency is thus less than a factor of 4. The half-life of Co-60 is short enough (5.2 years) that not taking it into account would significantly affect the results of the analysis.

The estimates for the number of onsite resident fatalities for the test/research reactor reference sites are similar for the EPA and NRC analyses. The estimated onsite resident fatalities for the rare earth extraction facility reference site are also similar for the two methods but slightly lower for the TSD.

The estimates for the uranium fuel fabrication reference sites ranged from a factor of 2.5 to a factor of 100 higher for the TSD analysis compared to the values reported in the GEIS. The reasons for this difference are not clear.

d) The methods used by NRC and EPA differ significantly; however, the results in terms of potential cancer fatalities in a population residing on the remediated site are similar, except for the case of the uranium fuel fabrication reference site.

e) In summary, the results of the analyses of fatal cancers averted for a specific residual dose criterion are reasonably consistent between the GEIS and the TSD. However, because the methods differ significantly in their assumptions, it is not clear whether the consistency is evidence for the robustness of the analysis, or simply fortuitous. In any case, the residual cancer estimates below 100 mrem/yr are all below 0.01 cancers, which suggests that the differences would not be very significant for regulatory policy.

Table A.1: Comparison of Source Term Parameters for the GEIS and the TSD

Reference Site (Nuclide of concern)	Est. Soil Conc.[1] (pCi/g)		Areal Extent (m^2) of contamination		Depth (cm) of contamination[2]	
	TSD	GEIS	TSD	GEIS	TSD	GEIS
Power Reactors (Co-60)	300	60	7,000	186	15	<20
Test/Research Reactors (Cs-137)	10,000	20	3,300	465/46[4]	15	<40
Rare Earth Facilities (Th-232)	300	200	13,800	9,300	250	< 6
Uranium Fuel Fabrication Facilities (U-tot)	2,300	1,000[3]	20,000	4,600	36/100[5]	<45

1. Estimated soil concentrations are "peak" concentrations for the TSD and "high" average concentrations for the GEIS.
2. The depths of contamination for the GEIS are estimated from Figures 5.5.1 - 5.5.5 (GEIS, Appendix C, p. C.5.10 to C.5.12).
3. The GEIS refers specifically to the concentration of U-235 in Table 5.5.1 and Figure 5.5.4. It is not clear whether that value is intended to reflect concentrations of U-235 or total uranium.
4. Assumed areal extent is 470 m^2 for the reference test reactor (Plum Brook) and 47 m^2 for the reference research reactor (Oregon State University).
5. Two values are reported for contamination depth in TSD Appendix Table B-1 (p. B-9). A depth of 36 cm was used to calculate population impacts; the more conservative value of 100 cm was used to calculate RME risks (TSD, p. 4-99).

Table A.2: Comparison of GEIS and TSD Estimates of Radiogenic Cancer Fatalities for the Commercial Nuclear Power Reactor Reference Site (TSD Reference Site XVI)

Residual Dose Criterion (mrem)	On-Site Residential Fatalities[1]		Fatalities Averted[4]
	GEIS[2]	TSD[3]	
> 100		2.18 E-3	
100	2.0 E-4	4 E-5	2.14 E-3
75 60	 1.2 E-4	3 E-5 	2.15 E-3
30 25	6.1 E-5 	 1 E-5	 2.17 E-3
15	3.1 E-5	1 E-5	2.17 E-3
10	2.0 E-5	<1 E-5	2.18 E-3
3	6.1 E-6	<1 E-5	2.18 E-3
1	2.0 E-6	<1 E-5	2.18 E-3
0.1	2.0 E-7	<1 E-5	2.18 E-3

1. GEIS estimates of onsite resident fatalities are from Table 5-1 of the GEIS (p. 5-20). Estimates for the TSD are from Appendix M, Tables M-89 and M-90 (p. M-89 and M-39). Appendix M values are potential cancer deaths averted. The onsite resident fatalities for the TSD were calculated by subtracting the total cancer deaths from the total for the unremediated site. For example, at the 15 mrem/yr residual dose criterion, the total cancers averted is 2.17 E-3. Subtracting that value from the total cancers expected for the unremediated site, 2.18 E-3, gives a value of 1 E-5.
2. The total area of the contamination assumed for the GEIS is 186 m^2
3. For the TSD, the total area is 7,000 m^2.
4. From TSD, Appendix M, Tables M-89 and M-90 (p. M-89 and M-90).

Table A.3: Comparison of GEIS and TSD Estimates of Radiogenic Cancer Fatalities for the Test/Research Reactor Reference Site (TSD Reference Site XVIII)

Residual Dose Criterion (mrem)	On-Site Residential Fatalities[1]		Fatalities Averted[4]
	GEIS[2]	TSD[3]	
> 100		6 E-4	
100	5.1 E-4	6 E-4	2.59 E-2
75 60	 3.1 E-4	4 E-4	2.61 E-2
30 25	1.5 E-5	 <1 E-4	 2.65 E-2
15	7.6 E-5	<1 E-4	2.65 E-2
10	5.1 E-5	<1 E-4	2.65 E-2
3	1.5 E-5	<1 E-4	2.65 E-2
1	5.1 E-6	<1 E-4	2.65 E-2
0.1	5.1 E-7	<1 E-4	2.65 E-2

1. GEIS estimates of onsite resident fatalities are from Table 5-1 of the GEIS (p. 5-20). Estimates for the TSD are from Appendix M, Tables M-89 and M-90 (p. M-89 and M-39). Appendix M values are potential cancer deaths averted. The onsite resident fatalities for the TSD were calculated by subtracting the total cancer deaths from the total for the unremediated site. For example, at the 75 mrem/yr residual dose criterion, the total cancers averted is 2.61 E-2. Subtracting that value from the total cancers expected for the unremediated site, 2.65 E-2, gives a value of 4 E-4.
2. The total area of the contamination assumed for the GEIS is 460 m^2 for research reactors and 46 m^2 for test reactors.
3. For the TSD, the total area is 3300 m^2.
4. From TSD, Appendix M, Tables M-89 and M-90 (p. M-89 and M-90).

Table A.4: Comparison of GEIS and TSD Estimates of Radiogenic Cancer Fatalities for the Rare Earth Extraction Facility Reference Site (TSD Reference Site XXI)

Residual Dose Criterion (mrem)	On-Site Residential Fatalities[1]		Fatalities Averted[4]
	GEIS[2]	TSD[3]	
> 100		7.47 E-2	
100	1.0 E-2	5.1 E-3	6.96 E-2
75 60	 6.1 E-3	3.1 E-3 	7.16 E-2
30 25	3.1 E-3 	 1 E-3	 7.37 E-2
15	1.5 E-3	5 E-4	7.42 E-2
10	1.0 E-3	3 E-4	7.44 E-2
3	3.1 E-4	1 E-4	7.46 E-2
1	1.0 E-4	<1 E-4	7.47 E-2
0.1	1.0 E-5	<1 E-4	7.47 E-2

1. GEIS estimates of onsite resident fatalities are from Table 5-1 of the GEIS (p. 5-20). Estimates for the TSD are from Appendix M, Tables M-89 and M-90 (p. M-89 and M-39). Appendix M values are potential cancer deaths averted. The onsite resident fatalities for the TSD were calculated by subtracting the total cancer deaths from the total for the unremediated site. For example, at the 15 mrem/year residual dose criterion, the total cancers averted is 2.42 E-2. Subtracting that value from the total cancers expected for the unremediated site, 7.47 E-2, gives a value of 5 E-4.
2. The total area of the contamination assumed for the GEIS ranges from 9300 m^2.
3. For the TSD, the total area is 13,800 m^2.
4. From TSD, Appendix M, Tables M-89 and M-90 (p. M-89 and M-90).

Table A.5: Comparison of GEIS and TSD Estimates of Radiogenic Cancer Fatalities for the Uranium Fuel Fabrication Reference Site (TSD Reference Site XX)

Residual Dose Criterion (mrem)	On-Site Residential Fatalities[1]		Fatalities Averted[4]
	GEIS[2]	TSD[3]	
> 100		1.28 E-2	
100	5.1 E-3	1.24 E-2	4.07 E-4
75 60	 3.1 E-3	1.24 E-2 	4.39 E-4
30 25	1.5 E-3 	 1.23 E-2	 5.50 E-4
15	7.6 E-4	1.20 E-2	8.46 E-4
10	5.1 E-4	1.08 E-2	2.04 E-3
3	1.5 E-4	7.1 E-3	5.71 E-3
1	5.1 E-5	4.6 E-3	8.21 E-3
0.1	5.1 E-6	<1 E-4	1.28 E-2

1. GEIS estimates of onsite resident fatalities are from Table 5-1 of the GEIS (p. 5-20). Estimates for the TSD are from Appendix M, Tables M-89 and M-90 (p. M-89 and M-39). Appendix M values are potential cancer deaths averted. The onsite resident fatalities for the TSD were calculated by subtracting the total cancer deaths from the total for the unremediated site. For example, at the 15 mrem/year residual dose criterion, the total cancers averted is 8.46 E-4. Subtracting that value from the total cancers expected for the unremediated site, 1.28 E-2, gives a value of 1.20 E-2.
2. The total area of the contamination assumed for the GEIS is 4600 m^2
3. For the TSD, the total area is 20,000 m^2.
4. From TSD, Appendix M, Tables M-89 and M-90 (p. M-89 and M-90).

APPENDIX B - DETAILED COMMENTS ON SOURCE TERM INFORMATION USED TO DEFINE REFERENCE SITE CHARACTERISTICS OF DOE FACILITIES

1. Reference Site I

For this reference site, EPA used aerial survey data and assumed Cs-137 concentrations to estimate source terms and contamination volume/concentration relationships. While the analysis itself is reasonable, the extent to which this analysis is representative of the actual Hanford site is questionable. EPA recognized that the Hanford site included many more radionuclides than just Cs-137. However, those listed are not the complete list, although EPA's intentions to consider the most recent Remedial Investigation/Feasibility Study (RI/FS) documents (TSD, p. 4-39) should address this concern. Some of these other radionuclides may significantly impact waste volumes. depending on cleanup levels selected. The assumption that Cs-137 is the only important radionuclide needs further consideration. For example, EPA indicated that concentrations of Pu are well below 1 pCi/g. This may be true if concentrations are averaged over the entire site; however, many areas have concentrations well above this level. The Remedial Investigation Feasibility Study (RI/FS) for the Environmental Restoration Disposal Facility (ERDF) reported maximum concentrations for Pu-239/240 of 2,800 pCi/g (DOE, 1993b). Similarly, the statement that the U-238 concentrations are within the range of natural background is only true for offsite regions and those unused areas of the site (which do, however, make up most of the site). DOE (1993b) reported a maximum uranium-238 concentration of 20,000 pCi/g.

These data should help develop a reference site that is more representative than a reference site based on aerial survey data. While aerial data may be useful, the concentrations resulting from such data depend greatly on the assumed depth and extent of the contamination. Hence, the concentration isopleths and associated volumes at concentration intervals are totally dependent on the assumptions. All of the volume data are an artifact of the 5 cm depth assumption. While this assumption is reasonable for surface deposited material (Cs-137 typically moves very slowly through the surface and even in wet eastern sites, more than 90% remains in the top 15 cm), many of the most contaminated areas of Hanford are not the result of surface deposition. In addition, other radionuclides such as Sr-90 and Tc-99, migrate differently than Cs-137 and, therefore, the 5 cm assumption may not be appropriate. Much of the soil contamination at Hanford is at depths of 1.5 to 5 meters or more. Without incorporating consideration for these volumes of material, the data for Reference Site I can only be considered representative of a site contaminated with wind-blown Cs-137. This reference site requires further analysis using the more recently released Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) RI/FS and Focused Feasibility Study (FFS) reports for the 100, 200 and 300 areas of Hanford before it can be considered representative of the Hanford site.

2. Reference Site II

In this reference site, the modeling considered only the upper 18 inches of soil from several subunits of Operable Unit 5 at FEMP (Fernald Environmental Management Project). EPA recognizes that the contamination at the FEMP site exists at depths greater than 18" but the subsurface contamination was not considered because it did not fit well in the simplified modeling approach used to develop the reference sites. EPA also notes that the subsurface data are difficult to interpret.

The acceptability of this approach and use of the data are dependent on the desired use of the data. It is possible to complete multiple runs with the models used by EPA and analyze the more complex contamination (e.g., subsurface lenses). Furthermore, analysis of the complex data with multiple radionuclides at low concentrations would provide EPA with insight into the difficulty associated with designing remedial actions at low dose-based cleanup concentrations. Therefore, if the goal of the reference site is to assess the viability of establishing cleanup criteria at concentrations that are based on doses of 15 mrem/year and below, analysis of the more realistic data set would seem worthwhile and appropriate. This would be more beneficial than assuming the contamination depth was limited to 5 cm (TSD, page 4-45).

EPA should also review the waste volume/cost to concentration data provided in the DOE March 23, 1994, comments to NRC related to their cleanup standards. While most of the sites analyzed in these DOE comments were relatively small, there is much to learn from the ratios of volume to concentration. Some of these sites conducted operations that were similar to those conducted at Fernald.

3. Reference Site III

EPA chose to rely primarily on aerial survey data for this site. The volume to concentration ratios are largely an artifact of the assumptions. Many of the comments relating to Reference Site I also relate to this reference site. While the general shape of the concentration to volume curve (Figure 4-12) is reasonable, this analysis appears to provide little new insight into the relationships. It is not likely to produce good cost to cleanup concentration relationships because it is limited to Cs-137 alone and again assumes that contamination is restricted to the top 5 cm of soil.

Rather than conducting multiple reference sites that are largely controlled by the assumptions of the depth of the contamination, it might be more useful to concentrate on one site and conduct a more complex analysis. Such an analysis would also be beneficial in determining how representative the simplified sites are to the actual sites.

4. Reference Site IV

Estimates of contaminated volumes from the Weldon Spring site were made by DOE (from NEPA and CERCLA documents supporting the Record of Decision) and provided to NRC and EPA in a March 23, 1994, Comment letter (DOE, 1994a). The estimates of contaminated volume (rounded to 2 significant digits) for the Weldon site were:

Sludge	220,000 yd^3	170,000 m^3
Sediment	120,000 yd^3	92,000 m^3
Soil	340,000 yd^3	260,000 m^3
Structural Mat.	170,000 yd^3	130,000 m^3
Process Chemicals	4,000 yd^3	3,000 m^3
Vegetation	31,000 yd^3	23,000 m^3
Total	885,000 yd^3	678,000 m^3

As can be seen in this example, soil represents about 40% of the waste volume from the remedial action. (Vegetation represents about 10% of the soil volumes and, while it is typically part of the soil restoration, it does not add significantly to the volume.) The site is non-homogeneous. Excluding the highly contaminated raffinate pits, the soil includes a few highly concentrated areas extending tens of centimeters in depth, with the bulk of the soil contamination being slightly elevated surface contamination. Building and equipment debris constitute the largest fraction of the waste (excluding the waste pit area, regardless of the soil limit selected, the soil will comprise only a small fraction of the total). Therefore, the analysis conducted by EPA represents only a portion of the residues that result from the action.

The EPA analysis suggests that the relationship (Figure 4-12) between the volume of contaminated soil at a uranium standard in pCi/g is:

60 pCi/g	36,000 m^3 (approximate from Fig. 4-12)
30 pCi/g	48,000 m^3
15 pCi/g	60,000 m^3

For comparison, DOE's analysis provided in the March 23, 1994, comment package suggested remediated soil concentrations to volume relations as follows:

120 pCi/g	8,400 m^3
60 pCi/g	20,000 m^3
30 pCi/g	28,000 m^3
15 pCi/g	38,000 m^3

These volumes excluded the waste pits and quarry. In this case, the volumes differ by about 70%, i.e., EPA estimates are between 1.6 to 1.8 times higher than the DOE estimates. Although this difference is significant, a factor of 2 agreement is not unreasonable for volume estimates from different modeling analyses. In any case, the log-linear relationship between soil volume and cleanup limits seems to be a reasonable approximation for the soil at this site. The difference in the actual volume estimates is likely due to EPA's assumptions of uniform depth of 11 cm for the 15 pCi/g volume of the waste. Contamination at the site was non-homogeneous. Many areas consisted of very shallow surface contamination, with some localized highly contaminated ares that extended a few tens of centimeters in depth.

In general, if the goal of this reference site is to collect data that will be used to assess the cost of soil cleanup at a site absent of waste disposal activities and structures, this analysis of soil limits may be adequate. However, if it is to represent a site like the Weldon Spring Site, it is not clear how an adequate analysis of costs, risks and benefits can be completed without considering at least some of the other wastes. For example, the statement on page 4-61 of the TSD that little, if any, of the quarry waste falls within the scope of this report is not clear. If the quarry site were not already being decontaminated under a site-specific CERCLA Record of Decision (ROD), it would seem appropriate to consider the standard being developed in 40 CFR Part 196 in the action.

The comparisons above were based on data collected for DOE's March 23, 1994, comments to the Nuclear Regulatory Commission on their draft cleanup standards for 10 CFR Part 20. In the March 1995 Baseline Environmental Management Report ("Estimating

the Cold War Mortgage"), DOE provided the following estimates of waste from the various parts of the Weldon Spring site:

Quarry Site (9 acres)	- 127,000 yd^3 of soil and rubble. - 3 million gallons of contaminated water.
Four Waste lagoons	- 408,000 yd^3 of sludge and soil. - 57 million gallons of contaminated water
Chemical Plant area (44 buildings and structures)	- 348,000 yd^3 of soil and building material
Vicinity Properties (offsite locations near the former processing site)	- 25,300 yd^3 of soil

These data are not broken out sufficiently to compare to previous estimates. Total soil, sludge, rubble and building wastes comprise about 908,000 yd^3, which is only about 3% greater than the 885,000 yd^3 used in the above analysis.

5. Reference Site V

As noted in comments on the other reference sites, aerial survey data do not necessarily give a true representation of volume to concentration ratios. At this reference site, it is assumed that the distribution of contaminants in the soil is the same as the arithmetic mean of the fraction of soil found at other sites. Little real site-specific data have been used in the development of the source term for this site. As a result, it is not clear if there is any real benefit in including this reference site in the EPA analysis.

In any case, use of the average values from all five sites would result in 90% of the volume from this site being less that 100 pCi/g. However, if one only considered the eastern sites (West Valley and Oak Ridge Reservation), about 84% of the volume is less that 100 pCi/g and 50% rather than 58% is less than 10 pCi/g. While these are not significant differences given the large uncertainty of this approach, these data could result in differences of a few million cubic meters at the 5 pCi/g range.

6. Reference Site IX

In general, the estimates of this reference site provide a reasonable estimate of the soil contamination source term at the Rocky Flats site. It of course does not include building rubble or decontamination residues. However, as noted, these wastes are not within the scope of the study.

7. Reference Site XXII

Reference site XXII was developed to represent the Maywood Chemical site in particular and, in turn, other Formerly Utilized Sites Remedial Action Program sites (FUSRAP sites). Two general comments relate to these assumptions. First, the Maywood site is a former thorium processing site. While several FUSRAP sites are former thorium, rare earths or uranium process locations, most are not. Many sites are former uranium metal working sites (e.g., uranium metal machining, rolling or casting), while others are storage or conversion sites. Contamination at many FUSRAP sites consists mostly of building contamination. Therefore, while the Maywood site is an example of one of the larger soil contamination sites, it is not representative of many FUSRAP sites. It may, however, be representative of many similar licensed fuel cycle or rare earths processing facilities. The results should be compared to such sites.

The second comment relates to the development or data to support a standard for Maywood-like sites. In, general, the background documents do not appear to provide sufficient information in this regard. For example, Table 6-6 of the TSD indicates that over a 1000-year integration time, a cleanup to the 10^{-4} risk-based level will avert 53 cancers. However, the analysis does not provide the number of cancers averted at other levels (e.g., 10^{-3} or 10^{-2}). Such comparisons are needed to determine the incremental collective dose or risk averted by the 1 in 10,000 (10^{-4}) risk-based standard. Page 6-94 of the TSD indicates that the 1 in 10,000 standard would be met by a cleanup standard of 0.09 pCi/g for radium-226 and 0.37 pCi/g for thorium-232. These are extremely low concentrations and are not practically differentiable from background. (A copy of this analysis was provided to the SAB. See DOE, 1994b).

However, it is also not clear that the incremental protection afforded by a 0.09 pCi/g radium-226 standard is sufficient to justify the costs and risk resulting from the standard. In analyzing and selecting a standard, DOE compared several potential cleanup standards for the site. These are discussed in some detailed in the DOE comments to the Nuclear Regulatory Commission (DOE, 1994a). Background was not specifically considered (0.09 pCi/g cannot be distinguished from background); hence, the results do not overlap with EPA's analysis. However, useful comparisons can be made.

Under the no-action alternative, DOE estimated potential doses from continued use of contaminated properties to range from 12 to 2,800 mrem in a year. Over a 200-year period (DOE selected 200 years as a practical limit over which collective doses could be reasonably estimated; see Subcommittee comments on time period in section 8.4 of this report, the collective dose for continued use of the property was estimated to be about 12,000 person-rem. Assuming a risk factor of 5 in 10,000 cancers per person-rem, indicates a potential 6 fatal cancers. (Note: EPA estimated 53 cancers averted in 1,000 years which at background is sufficiently close to 6 in 200 years times 5 or 30 cancers in 1,000 years.) A cleanup standard of 30 pCi/g with backfill of clean soil resulted in a residual collective dose of 880 person-rem over 200 years or about 11,000 person-rems averted (5.5 cancers) over 200 years. The residual individual dose was estimated to have been reduced to 3.6 mrem in a year for residential properties and 8.2 mrem per year for commercial properties. A 15 pCi/g standard would reduce individual doses to one half that of the 30 pCi/g standard and would have a residual collective dose of 440 person-rem in 200 years. The incremental reduction in collective dose and risk between a 30 pCi/g and 15 pCi/g standard was estimated to be 440 person-rems or 0.2 fatal cancers averted. At 5 pCi/g, the incremental reduction was 280 person-rem in 200 years or about 0.1 cancers averted. The residual risk from the site following a 5 pCi/g cleanup was estimated to be 160 person-rems in 200 years or 0.08 cancers. Therefore, the incremental reduction of the 1 in 10,000 risk-based standard of 0.09 pCi/g would be less than 0.08 cancers per 200 years.

At a 5 pCi/g standard, the collective dose to remedial workers was estimated to be 30 person-rems or about 0.015 cancers. Transportation risks were estimated to range from about 0.002 to 0.2 fatalities depending on mode. Construction accidents were estimated to range from 0.001 to 0.01 fatalities. Therefore, at 5 pCi/g, worker risk begins to be comparable to risks averted by the action. While the DOE has not estimated risks at a cleanup to background, it is possible, given the data, that worker risks will exceed the incremental benefit of the cleanup. These could be estimated using EPA's volumes which are comparable to DOE estimates. DOE has estimated that at the cleanup standard of 5 pCi/g for the surface and 15 pCi/g for the subsurface that there is about 300,000 cubic meters (this is an in-situ volume estimate; excavated volumes would be about 30% greater). A 5 pCi/g overall standard was estimated to increase the volumes between 20 and 100% (the difficulty in measuring 5 pCi/g and less resulted in these large uncertainties). The 300,000 cubic meter volume is equivalent to the EPA 10^{-2} risk-based level in Table 6-7 of the TSD. Cleanup to the 10^{-4} risk level would increase the EPA estimated volumes to 1,300,000 cubic meters and, hence, would increase the transportation and accident fatalities proportionally while decreasing the public risk by less than 0.08 hypothetical cancers.

Appendices L and M of the TSD allow assessment of the incremental numbers of cancers throughout the range of 10^{-2} to 10^{-6} lifetime cancer risk, and for dose rates between 0.1 to 100 mrem/yr.

APPENDIX C - GLOSSARY OF TERMS AND ACRONYMS

AGN	Atomic General Nuclear (A line of small, research reactors found at many universities)
Am	Americium, as an element or as an isotope of thorium or uranium alpha-decay chains. . The isotope of most importance is Am-241, produced by the decay of Pu-241 present in fallout.
AMAD	Activity Median Aerodynamic Diameter (e.g., aerosol)
ARAR	Applicable or Relevant and Appropriate Requirement
BEIR	Biological Effects of Ionizing Radiation
C	Carbon, as an element or isotope. Isotope C-14 is radioactive with a half-life of 5,730 years
Ce	Cerium, as an element or isotope (e.g., Ce-144)
CE	evapotranspiration rate
CEDE	Committed Effective Dose Equivalent
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980 (Public Law PL 96-510, also known as “Superfund”)
CFR	Code of Federal Regulations
Ci	Curies , nuclear transformations (disintegrations). The special unit of activity. One curie equals 3.7×10^{10} disintegrations per second)
cm	centimeter
Co	Cobalt, as an element or isotope (e.g., Co-57, Co-60)
CR	runoff rate
C_s	Soil Concentration (Concentration in soil)
Cs	Cesium, as an element or isotope (e.g., Cs-137)
CU-POP	Cleanup Population Model
C_w	Porewater concentration
D	Decay product(s) or progeny of radioactive species, which may itself be radioactive
DCF	Dose Conversion Factor(s). The absorbed dose in rad per working-level month.
DF	Dilution Factor
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
E	A factor of 10 to an exponent (powers of 10)
EEC	Environmental Engineering Committee (U.S. EPA/SAB)
EG&G	Edgarton Germhausen & Grier, Inc.
EPA	U.S. Environmental Protection Agency (Also known as U.S. EPA, or "the Agency")
ERDF	Environmental Restoration Disposal Facility

FEMP	Fernald Environmental Management Project
FFS	Focused Feasibility Study
ft	feet
FR	Federal Register
FUSRAP	Formerly Utilized Sites Remedial Action Program
g	gram
gal	gallon
GEIS	Generic Environmental Impact Statement
H	Hydrogen, as an element or isotope (e.g., H-3)
HEAST	Health Effects Assessment Summary Table
hr	hour
i	Hydraulic gradient
I	Infiltration rate
I	Iodine (e.g., I-129)
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IDB	Integrated Data Base
INEL	Idaho National Engineering Laboratory
IR	Irrigation Rate
K	Hydraulic Conductivity (permeability). The volume of water that will move per unit time in the aquifer under a unit gradient through a unit cross sectional area perpendicular to the direction of flow.
K_d	Distribution coefficient
Kg	Kilogram
L	Liter
LANL	Los Alamos National Laboratory
m	Milli-, [10^{-3}] in combination with specific units
m	meters
M	Mega (e.g.., Mwt). Also MW(t), Mega Watts (thermal) or Million Watts (thermal
m^2	square meters)
m^3	cubic meters
min	minute
mrem	millirem (milli roentren equivalent man)
NACEPT	National Advisory Committee on Environmental Policy and Technology
NAS	National Academy of Sciences (National Research Council)
NCRP	National Council on Radiation Protection and Measurements
NEPA	National Environmental Policy Act (Public Law P.L. 91-190)
NESHAP	National Emission Standards for Hazardous Air Pollutants
NESHAPS	National Emission Standards for Hazardous Air Pollutants

NRR	U.S. Nuclear Regulatory Commission, Office of Nuclear Reactor Regulation
NORM	Naturally Occurring Radioactive Material
Np	Neptunium, as an element or isotope (e.g., Np-237)
NRC	U.S. Nuclear Regulatory Commission
ORIA	Office of Radiation and Indoor Air (U.S. EPA)
ORR	Oak Ridge Reservation
P	Precipitation rate
pCi	pico-Curies
p	pico-, [10^{-12}] in combination with specific units
Pb	Lead, as an element or an isotope of thorium or uranium alpha-decay chains (e.g., Pb-210)
pH	Negative log of hydrogen ion concentration
Po	Polonium, as an element or as an isotope of thorium or uranium alpha decay chains (e.g., Po-210)
PRESTO	A family of codes developed by the EPA to evaluate doses resulting from the disposal of low-level radioactive waste. These codes include PRESTO-EPA-CPG (assesses annual effective dose equivalents to a critical population group), and PRESTO-EPA-POP (assesses cumulative population health effects to the general population residing in the downstream regional basin on a low-level waste site)
Pu	Plutonium, as an element or as an isotope (e.g., Pu-239, Pu-240)
Ra	Radium, as an element or as an isotope of thorium or uranium alpha decay chains (e.g., Ra-223, Ra-224, Ra-226)
RAC	Radiation Advisory Committee (U.S. EPA/SAB)
RAD	Radiation Absorbed Dose
RAGS/ HHEM	Risk Assessment Guidance for Superfund/Human Health Evaluation Manual
RCF	Risk Conversion Factor
RCRA	Resource Conservation and Recovery Act, as amended by the Solid Waste Disposal Act
RCSS	Radionuclide Cleanup Standards Subcommittee (U.S. EPA/SAB/RAC)
redox	reduction-oxidation condition
rem	roentgen equivalent man. A special unit of absorbed dose equivalent numerically equal to the absorbed dose in rads multiplied by the quality factor, the distributional factor and any other necessary modifying factors.
RESRAD	Residual Radiation pathway model computer code developed by DOE
RI/FS	Remedial Investigation/Feasibility Study
RME	Reasonable Maximum Exposure (Also Reasonably Maximum Exposed) - Individual
Rn	Radon, as an element, or as an isotope (e.g., Rn-222)

RSC	Radionuclide Soil Concentration
Ru	Ruthenium, as an element or as an isotope (e.g., Ru-106)
S	fraction of porespace that is water-filled
SAB	Science Advisory Board (U.S. EPA)
SDMP	Site Decommissioning Management Program
SF	Slope Factors
SI	International System of units (invert S & I)
SSA	Soil Specific Activity
STUK	A model developed by the Finnish Centre for Radiation and Nuclear Safety
Sr	Strontium (e.g., Sr-90)
$t_{1/2}$	Half-life (of a radionuclide)
Tc	Technetium, as an element or isotope (Tc-99)
Th	Thorium, as an element or as an isotope (e.g., Th-228, Th-230, Th-232, Th-234)
TRIGA	Training, Research, and Instruction -- General Atomics (A small reactor manufactured by General Atomics.)
TSD	Technical Support Document
U	Uranium, as an element or as an isotope (e.g., U-234, U-235, U-238)
U_f	Uptake factor for food ingestion
UMTRCA	Uranium Mill Tailings Radiation Control Act
VAMP	An international model validation study, organized by IAEA using data sets provided by STUK
WLM	Working Level Month
Wt	Watt
yd	yard
yr	year

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AN SAB REPORT: REVIEW OF RADIONUCLIDE CLEAN-UP LEVELS FOR SOIL

REVIEW OF TECHNICAL ASPECTS OF ORIA'S TECHNICAL SUPPORT DOCUMENT FOR THE DEVELOPMENT OF CLEANUP LEVELS FOR SOIL BY THE RADIATION ADVISORY COMMITTEE