FEB 1994

Los Alamos
Los Alamos National Laboratory
Los Alamos, New Mexico 87545

DATE: February 22, 1994
IN REPLY REFER TO: EM/ER:94-A066
MAIL STOP: M992
TELEPHONE: (505) 667-0819

Mr. Joseph C. Vozella, Chief
Environment, Safety, and Health Branch
US Department of Energy
Los Alamos Area Office, MS A316
Los Alamos, NM 87544

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FEB 28 2 19 PM '94

Dear Mr. Vozella:

SUBJECT: RESPONSE TO NOTICE OF DEFICIENCY CONCERNING INSTALLATION WORK PLAN, REVISION 3 ACTIVITY DATA SHEET NUMBER 2105 WORK BREAKDOWN STRUCTURE NUMBER 1.4.2.6.3.4.16

The Los Alamos National Laboratory's response to the Notice of Deficiency concerning Revision 3 of the Installation Work Plan for Environmental Restoration is enclosed. The Notice of Deficiency was received at the Los Alamos Area Office on January 31. A response within thirty days is required by the Environmental Protection Agency. A documented quality review of the enclosed response has been performed in accordance with procedure LANL-ER-AP-01.3. A certification form signed by the appropriate Laboratory official is enclosed, too.

The enclosed response shows the comments from the Notice of Deficiency and the proposed changes to be made to the Installation Work Plan, Revision 3. When the responses are approved by the Environmental Protection Agency, appropriately changed pages of the draft Revision 3 will be distributed to all who have received copies of the draft revision. The changes proposed for Appendix K are fairly extensive, and some minor wording changes would be desirable in parts of that appendix not affected by the Notice of Deficiency. A complete version of Appendix K is enclosed with strike-throughs to show the portions of the original text that are proposed for deletion and text in bold for proposed additions. We propose to issue a complete Appendix K when the other changes to Revision 3 are distributed. Typographical errors in Appendix F have been distributed in the draft revision. Generally they list a Class code as "007HSWA", or similar, instead of "HSWA". These typographical errors will be corrected. Then a complete Appendix F will be distributed, too.

Please let me know if there are questions about this response to the Notice of Deficiency, or contact Ted Norris of my staff at 665-5136. I would appreciate having the response and the signed certification form sent to the Environmental Protection Agency, Region 6, in time to meet the required deadline.

Sincerely,

12473

David J. McInroy
Acting Program Manager
Environmental Restoration Program

Enclosures: Responses to Notice of Deficiency
Certification form
Appendix K with proposed revision shown

Cy: T. Baca, EM, MS J591
J. Shipley, EM, MS J591
P. Aamodt, EM/ER, MS M992
T. Norris, EM/ER, MS M992
B. Barnett, IS-1, MS M769
V. Maes, EM/ER, MS M992, (A.I. #A006)
RPF, MS M707
CRM-4, MS A150, w/o enclosure

The attached document, "Response to Notice of Deficiency Concerning Installation Work Plan, Revision 3, Activity Data Sheet Number 2105,Work Breakdown structure Number 1.4.2.6.3.4.16" is true and accurate to the best of our knowledge. The document has been through an internal formal review, to ensure accuracy and in accordance with the Los Alamos National Laboratory (LANL) Environmental Restoration (ER) Program quality assurance procedure. The formal review comments and the resolutions to these comments are part of the public record and are available in the LANL ER Records Processing Facility. The document is being submitted to the Environmental Protection Agency as part of the requirements under the Hazardous and Solid Waste Amendments Operating Permit.

Paul L Aamodt

Paul Aamodt, EM/ER, MS M992
Programmatic Project Leader

2/22/94
Date

Pat Shanley

Pat Shanley, ESH-8, MS K490
ESH-8 Representative

2/23/94
Date

David McInroy

David McInroy, EM/ER MS M992
Acting Program Manager
Environmental Restoration

2/22/94
Date

LIST OF DEFICIENCIES, DISCUSSION, AND PROPOSED TEXT CHANGES FOR INSTALLATION WORK PLAN, REVISION 3

1. 3.3.1.3 Uncertainty in Exposure Concentration Estimates, p. K-16 -

EPA would prefer that LANL use the more traditional approach to risk assessment (RAGS). LANL may use Monte Carlo methods as a measure of uncertainty in the uncertainty section of any risk assessment document. If LANL chooses to use the Monte Carlo technique, LANL will need to justify the distribution used for each exposure parameter.

Discussion

In general LANL plans to use the 95th UCL for the arithmetic mean as the simplest point estimate of contaminant concentration for the purposes of estimating the chronic RME, together with the conventional values for other pathway parameters that are tabled in Attachment 1, pp. K-53 through K-55. LANL agrees completely that the Monte Carlo alternative requires a great deal of additional work to justify the distributions selected for parameters other than concentrations in site media. A similar effort is required for any quantitative uncertainty analysis. (RAGS guidance requires only a qualitative discussion.) Where such an uncertainty analysis indicates that the point estimates are in one of the extreme tails of the likely distribution of exposure concentrations, and where the decision has major impacts on budget, community relations, or other critical factors, LANL may wish to make this investment in an alternative RME estimate.

Section 3.3.1 ("Methods") has been rearranged and slightly rewritten to make LANL's position clearer.

Proposed Text Changes

3.3.1 Methods

3.3.1.1 ~~Direct~~ Estimation Use of Sampling Data

Insofar as possible, sampling data will be used to directly estimate exposure concentrations for ER Program risk assessments. Direct use of sampling data is particularly applicable when exposure involves direct contact with the environmental medium of concern (e.g., direct contact with contaminated soils) and in cases where monitoring has occurred directly at an exposure point (e.g., a drinking water well). For these example exposure scenarios, sampling data provide the best estimate of current exposure concentrations. ~~The data quality objectives (DQO) process applied to the design of sampling plans during the phases of the RFI will result in the collection of environmental data appropriate for use in the risk assessment.~~

~~**3.3.1.2 Modeling Approaches**~~

In some instances, it may not be appropriate to use ~~direct~~ sampling data ~~alone~~ **directly** to estimate exposure concentrations, and some form of environmental fate and transport calculations or computer modeling applications will be required. Instances in which fate and transport calculations or modeling may be required include the following:

- Fate and transport calculations for contaminants will be required when exposure points are remote from contaminant sources and when release mechanisms exist to transport contaminants to an exposure point (e.g., transport of contaminants in groundwater to a receptor).

- Modeling will be required when temporal distribution of contaminants is needed and monitoring data are lacking. Typically, environmental sampling data give an indication of current conditions. Although there are situations in which it is reasonable to assume that contaminant concentrations will remain constant over time, in many cases, the time span over which data are taken during one-time sampling are not adequate to predict future exposure concentrations.

- When direct measurements of contaminant concentrations in air are unavailable, fate and transport modeling will be needed to determine air concentrations based on contaminated soil concentrations.

The level of effort required to estimate exposure concentrations depends on the type and quantity of data available, the level of detail required in the assessment, and the expected level of risk to human health and the environment. In general, estimating exposure concentrations involves analyzing direct sampling data and applying simple environmental fate and transport calculations. **Section 3.3.2 describes simple methods for estimating exposure concentrations for the exposure routes of inhalation of particulates and gas, ingestion of contaminated plants, and external gamma radiation, on the basis of concentrations of contaminants in surface and subsurface soil.**

More sophisticated modeling approaches than those presented there may be required, based on PRS-specific needs. For those efforts requiring a complicated modeling approach, a wide variety of EPA-approved contaminant fate and transport models is available for use in exposure assessments. Contaminant fate and transport models are applied in ER Program risk assessments on a PRS-specific basis. The procedure ~~to conduct~~ **for** complicated contaminant fate and transport modeling includes

- identifying and bounding the modeling problem and approach,

- selecting qualified experts to identify the appropriate models to be applied (this step may also include developing models to account for unique conditions at the Laboratory), and

- communicating with regulatory agencies to obtain approval of the modeling approach before conducting the modeling exercise.

The data quality objectives (DQO) process applied to the design of sampling plans during the phases of the RFI will result in the collection of environmental data appropriate for use in the risk assessment, whether direct or indirect modeling approaches are required.

3.3.1.2 Estimation of Reasonable Maximum Exposure (RME)

EPA (1989, 0305) suggests using an "upper 95% confidence limit on the arithmetic average concentration of contaminants at a site (or portion thereof)" as the appropriate statistic to represent the RME. In general, LANL will use the upper 95% confidence limit for the average concentration within an exposure unit as an initial point estimate of RME. Selection of statistical estimators for this average and for the corresponding 95% upper confidence limit depends on the statistical distribution of the observations. In Bayesian analyses, 95% upper confidence limits would be replaced by upper 95% probability bounds.

A recent Science Advisory Board review (EPA 1993, 1013) of the *Human Health Evaluation Manual* recommended that EPA move toward a distributional approach for

calculating RMEs. This approach would require the development of distributions for the parameters that enter the calculation of individual exposures. These distributions, together with Monte Carlo simulation methods, would then be used to determine the distribution of exposure, and an appropriate quantile of the distribution (e.g., the 95th percentile) would be used as the RME. The ER Program at Los Alamos endorses this distributional approach to exposure and risk assessment, and distributions for environmental data and parameters used in calculating dose/intake should be developed. However, this represents a substantial investment of time and energy, and initially such distributions will be used primarily for quantitative uncertainty analysis.

3.3.1.3 Uncertainty in Exposure Concentration Estimates

Evaluating uncertainty is a very important component of the exposure assessment. EPA (1989, 0305) specifies that a qualitative discussion of uncertainty in the exposure analysis should include (1) a tabular presentation of the ranges of estimates of exposure concentrations and exposure parameters consistent with the data, the point values used in calculating the RME dose or intake, and a brief description of the rationale used in selecting the point value; and (2) a summary of major sources of uncertainty in the risk assessment that specifically addresses monitoring data, the models used to estimate exposure concentrations, and the ~~values~~ **point estimates** selected for intake ~~variables~~ **parameters**.

~~The same EPA guidance document suggests using an "upper 95% confidence limit on the arithmetic average [that is, the mean or expected value] concentration of contaminants at a site (or portion thereof) [that is, within an exposure unit]" as the appropriate statistic for representing an RME. The words in brackets above interpret the phrases used in the EPA regulatory guidance in terms that are used in the Los Alamos ER Program. The arithmetic mean of the observations from an exposure unit is not the only reasonable estimator for the mean concentration in that exposure unit. Depending on observed distributional shapes, there may be substantially better estimators. The selection of an appropriate estimator of the mean is tailored to each situation.~~

~~The EPA guidance assumes a classical approach to statistical data analysis. If a Bayesian approach is taken instead, constructs such as confidence bounds can be replaced by probability intervals, and significance levels can be replaced by probabilities that hypotheses are true. Although the basic idea of estimating an RME with an upper bound for a mean concentration may remain, the Bayesian construction of such an estimate is quite different from the classical construction espoused in the current guidance. A Bayesian analysis demands construction of an exposure concentration distribution from which pertinent statistics (such as means, probability bounds, quantiles) may be calculated directly.~~

~~A recent Science Advisory Board review (EPA 1993, 1013) of the *Human Health Evaluation Manual* recommended that EPA move toward a distributional approach for calculating RMEs. This approach requires that the EPA develop distributions for the variables needed to calculate individual exposures; these distributions, together with Monte Carlo simulation methods, are used to determine distributions of exposure, and an appropriate quantile of the distribution (e.g., the 90th or the 95th percentile) could be used as the RME (e.g., the 90th percentile of the exposure distribution corresponds to an exposure concentration for which there is a 90% probability that an individual is exposed at no greater level). The ER Program at Los Alamos endorses this distributional approach to exposure and risk assessment, and, where possible, distributions for environmental data and parameters used in calculating dose/intake should be developed and presented. However, at present, EPA's policy is to use distributional approaches only as a means of assessing uncertainty.~~

~~In the following subsections, simple methods are described by which to estimate exposure concentrations for the exposure routes of inhalation of particulates and gas, ingestion of contaminated plants, and external gamma radiation, on the basis of concentrations of contaminants in surface and~~

~~subsurface soil. More sophisticated modeling approaches than those presented may be required, based on PRS-specific needs.~~

In addition, particularly where a complex modeling approach to exposure estimation is required, quantitative assessments of the effect of uncertainty in all parameters, including both those estimated from site-specific data and those whose values are selected from other sources, is recommended. The Monte Carlo simulation method mentioned in the preceding section may indicate that the point estimates are in an extreme tail of the likely distribution of actual exposure concentrations. In this case a reevaluation of the selected point estimate may be required.

*****

2. 3.5.4.2 Corrective Measures Study Report, p. 3-23 -

Text in the last paragraph indicates that EPA will approve or request a revision of the CMS report within 120 days of receipt. There is no provision in the permit requiring review by EPA within any given time-frame. Certainly, EPA will attempt to have the CMS Report reviewed within 120 days; however, the sentence should be changed to indicate there is no deadline for EPA review.

Discussion

Text at the top of p. 3-24 has been changed to eliminate the wording pertaining to time frame.

Proposed Text Change

~~Within 120 days of~~ **After** receipt of the draft report, EPA will approve or request a revision of the CMS report. EPA's response will consider comments received from NMED and the public. DOE/UC will finalize the draft CMS report, incorporating comments received from EPA within 30 days of receipt.

*****

3. 4.1.1 Decisions During the RCRA Process, p. 4-3 -

LANL shall add to the first statement under decisions required during the RFI: "determining whether contaminants have been released to the environment ***and if so the extent of that release*** *(the site screening decision in Figure 4-1)".*

Discussion

Determining the extent of the release, and more generally characterizing "the nature, extent, direction, rate, movement and concentration of releases," will be done as required to "determin[e] whether corrective action is required for the site," i.e., under the second bullet under the RFI decisions on p. 4-3. See also response to #5 below.

We have amplified the paragraph immediately following the bullets to make this clear.

Proposed Text Change

4.1 RFI and Corrective Action Strategies

4.1.1 Decisions During the RCRA Process

The RCRA process outlined in Chapter 1 is designed to generate and implement appropriate decisions concerning corrective actions for PRSs identified by the RCRA facility assessment (RFA). Following RFA identification of potentially significant releases, the RCRA process generally proceeds through RFI, CMS, and corrective measures implementation (CMI), as illustrated in Figure 1-1. Progress is demarcated by several decision points, as illustrated in Figure 4-1. Many of these decisions require the collection and assimilation of environmental data. The Los Alamos technical approach to environmental restoration begins with statements of these decisions and the associated decision criteria that are sufficiently explicit to guide the design of field investigations and the evaluation of the resulting data.

Decisions required during the RFI include

- determining whether contaminants have been released to the environment (the site screening decision in Figure 4-1),
- determining whether corrective action is required for the site (a risk-based decision that may be made before or after CMS has been initiated), and
- determining whether a formal CMS is required to select and design an appropriate corrective action.

Usually environmental data will need to be acquired during the RFI in order to support these decisions. **The site screening decision requires information about the concentrations of contaminants in site media, particularly the upper range of such concentrations. Risk-based decisions may require more information about the extent of contamination, as well as about the direction and rate of its migration in the environment and the hydrogeological parameters that control that migration. Other site-specific or bench data may be needed to choose among corrective actions unless the choice of a remedy is obvious.**

*****

4. 4.1.1 Decisions During the RCRA Process, p. 4-5, fifth paragraph -

Approval of VCAs as final remedies should also follow the guidelines of public participation in a Class III permit modification.

Discussion

Wording has been added, as shown below.

Proposed Text Change

EPA approval of VCAs as final remedies will be requested **as a Class III permit modification, with public participation,** and a final report will be submitted to EPA and NMED, provided that cleanup standards have been attained and DOE has approved the final report. (The table of contents for the final

report is shown in Table 4-2). In other cases, VCAs may serve merely as interim measures, followed by a return to the normal RFI/CMS process.

*****

5. 4.1.2 The Streamlined Approach to Environmental Restoration, p. 4-7 -

Under the proposed Subpart S (EPA 1990), the general performance objective of an RCRA facility investigation is to characterize the nature, extent, direction, rate, movement, and concentration of releases from a site. In the IWP, LANL states the goal of an RFI is to determine whether the problem warrants corrective action, and then to develop an understanding of probable site alternatives. LANL should revise their approach to be consistent with Subpart S.

Discussion

The LANL and SAFER approach is to focus site characterization on corrective action decisions. DOE believes that in order to make wise use of our limited resources, we must stay focused on these decisions throughout planning for data collection and subsequent evaluation of data. The DQO process is one tool for maintaining this focus, and the use of risk-based decision criteria is one way to ensure that our priorities remain aligned with EPA's and the public's. We believe that our approach is entirely consistent with proposed Subpart S.

Proposed Text Change

None.

*****

6. Generic Quality Assurance Project Plan, Annex II, p. II-2 -

Text indicates that although the generic QAPjP was previously included in the IWP that since 1992 this document has been issued separately. When will EPA receive a revised generic QAPjP? LANL has had several problems related to inconsistency in use of analytical methods, and these problems might be avoided if the QAPjP was updated to meet all programmatic needs and was followed by the program.

Discussion

Desirability of updating the Quality Assurance Project Plan (QAPjP) is recognized. A start on a revised QAPjP was made a year ago. Resources are being identified to carry the revision process to completion. We anticipate having the first draft of the revision ready for internal review by the end of the fiscal year.

Proposed Text Change

None.

*****

7. 5.4 Objective 4 - Continue to Afford Opportunities for Public Comment on Environmental Restoration Program Activities, p. V-14 -

Paragraph three discusses Class III Modifications as being the least extensive change of the three classes while it should be the most extensive change. Please revise the text.

Discussion

Third paragraph of Section 5.4 of Annex V has been corrected as shown below.

Proposed Text Change

If the proposed changes fall in the category of a Class III modification (the class of modification that involves the ~~least~~ **most** extensive change of three classes) and if the public requests a public hearing, the EPA will schedule a formal hearing. If a public hearing is not requested, EPA will publish the decision without a hearing. In the case of Class III modification, the EPA will provide a 45-day comment period after the public hearing before the modifications become part of the permit. Although transcripts of public hearings are optional, if EPA decides to prepare transcripts, they will be available to the public in the Laboratory's community reading room and in the document repositories in the main public libraries in Los Alamos, Española, and Santa Fe.

*****

8. Appendix F, p. F-1 -

LANL indicates that the PRS Class Code, "RCRA PM" is for a unit regulated by RCRA which appears in the 1990 HSWA Module, but is one of 38 SWMUs deleted from the module by the permit modification proposed in March 1993. No units were removed from the HSWA permit in the permit modification proposed in March 1993. LANL needs to correct this statement and the table.

Discussion

The PRS Class Codes for Appendix F were devised in part on the basis of information for a permit modification that was sent from the Department of Energy's Los Alamos Area Office on March 29, 1993, to the New Mexico Environment Department, but which apparently was not received by the Environmental Protection Agency's Region 6. Therefore, the Appendix F Class Code designations and table entries are revised to eliminate reference to the RCRA PM class code.

Proposed Text Change

PRS Class Codes

HSWA	Listed in HSWA Module (one of the 605 SWMUs in the 1990 RCRA permit).
HSWA PM	Unit added in the permit modification proposed in February 1993.
RCRA	Unit regulated by RCRA; does not appear in 1990 HSWA Module.

~~RCRA PM~~ ~~Unit regulated by RCRA; appears in 1990 HSWA Module but is one of 38 SWMUs deleted from the module by the permit modification proposed in March 1993.~~

AOC/PRS Unit may not be a SWMU as defined by RCRA, but ER Program is investigating.

Current Yr	Class
15-003	~~RCRA PM~~ **HSWA**
16-010(b)	~~RCRA PM~~ **HSWA**
16-010(c)	~~RCRA PM~~ **HSWA**
16-010(d)	~~RCRA PM~~ **HSWA**
16-010(e)	~~RCRA PM~~ **HSWA**
16-010(f)	~~RCRA PM~~ **HSWA**
16-010(j)	~~RCRA PM~~ **HSWA**
16-012(d)	~~RCRA PM~~ **HSWA**
16-012(i)	~~RCRA PM~~ **HSWA**
16-012(j)	~~RCRA PM~~ **HSWA**
16-012(l)	~~RCRA PM~~ **HSWA**
16-012(m)	~~RCRA PM~~ **HSWA**
16-012(n)	~~RCRA PM~~ **HSWA**
16-012(p)	~~RCRA PM~~ **HSWA**
16-012(t)	~~RCRA PM~~ **HSWA**
16-012(u)	~~RCRA PM~~ **HSWA**
16-012(x)	~~RCRA PM~~ **HSWA**
16-018	~~RCRA PM~~ **HSWA**
21-003	~~RCRA PM~~ **HSWA**
21-028(a)	~~RCRA PM~~ **AOC/PRS**
21-028(b)	~~RCRA PM~~ **AOC/PRS**
21-028(c)	~~RCRA PM~~ **AOC/PRS**
21-028(d)	~~RCRA PM~~ **AOC/PRS**
21-028(e)	~~RCRA PM~~ **AOC/PRS**
53-001(a)	~~RCRA PM~~ **HSWA**
53-001(b)	~~RCRA PM~~ **HSWA**
53-002(a)	~~RCRA PM~~ **HSWA**
53-002(b)	~~RCRA PM~~ **HSWA**
54-009	~~RCRA PM~~ **AOC/PRS**

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9. Appendix H, 4.2 Binomial Sampling Strategy, p. H-8 and H-9 -

How does LANL ensure that consistency is maintained in the picking the probability of detecting contamination in at least one sample? In the examples used, LANL decided to use a probability of 0.85, for detecting contamination. What is the basis for using that probability rather than 0.99 or any other?

Discussion

This problem arises whenever a statistical approach to sample design is adopted; one must choose, in advance, what level of uncertainty, that is, what probability of error, can be tolerated. The usual *ad hoc*, nonstatistical approaches to sample design simply ignore this problem. LANL has no magic solution to this perennial problem, nor do we believe that there should be a "one-size-fits-all" solution. Some of the considerations that should be taken into account, on a site-by-site basis, are mentioned at the top of p. H-8. They include the expected heterogeneity of the site (based on the mechanisms by which the release might have occurred, the subsequent history of

construction and other disturbances at the site, and the behavior of the potential contaminants in the environment) and the toxicity of the potential contaminants.

Proposed Text Change

None.

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10. Appendix H, 5.2.2 Composite Sampling, p. H-16 -

The discussion of composite sampling does not take into consideration the extra time required if reanalysis of samples must occur because the value of the composite sample is greater than the screening action level. This potential time delay becomes a significant concern when data analysis takes several months, and this is an additional reason EPA does not support the use of composite sampling in the RFI process.

Discussion

LANL agrees that time delay can be a factor when composite sampling is used for "hot spot" detection or SAL comparisons. Most of this section discusses the use of composite sampling to estimate, with improved confidence, the arithmetic average concentration of contaminants at a site, as required for risk assessment, not for "hot spot" detection or SAL comparisons; reanalysis is not an issue in this application. We agree that some of the discussion at the end of the section obscured this point.

We have rewritten the latter part of this section and incorporated additional text into the first paragraph to clarify these distinctions and have included mention of the time delay factor (last sentence of rewrite).

Proposed Text Change

5.2.2 Composite Sampling

As a general rule, the larger the population variance, the more samples are needed to estimate a mean with given precision. This subsection and the following two subsections present sampling strategies that, correctly applied, can increase ~~precision~~ **the precision of the estimate of the mean** without increasing the number of laboratory analyses required.

One way to reduce sampling variability is to form composite samples. A composite sample is generally made up in the field. Grab samples [called "increments" in the statistical literature on composite sampling (Elder et al. 1980, 0795)] are collected as usual following the appropriate procedures, except that less material may be required in each grab sample than if each were to be analyzed separately. The material from several grab samples is combined and homogenized to form a composite sample from which subsamples (seldom more than two) are removed and packaged as usual for laboratory analysis. Clearly, this brief description passes over some significant practical details. In particular, it is desirable that each grab sample contribute an equal amount to the composite and to the subsamples, which suggests that any sieving to remove rocks and other debris should be done before the increments are composited.

Homogenization should be as thorough as possible. If volatile contaminants are of concern, compositing should not be used.

In order to estimate the population mean from composite samples, the following assumptions are made:

- The estimate is computed as the average of n = r x s subsamples, that is, Equation 5.1 is used with the measurements on s subsamples from each of r composites. (Equation 5.3, below, can be modified if not all composites have the same number of subsamples.)
- The volume of each composite is equal to that of s subsamples. (If s = 1, that is, if the entire composite is analyzed to produce one measurement per composite, the last term in Equation 5.3 should be ignored.)
- Each composite is formed from m≥1 grab samples. Equation 5.3 assumes that each grab sample is homogeneous; however, modifications can be made if this is not the case.
- σ^2 is the between-grab sample variability, usually more significant than any inhomogeneity within grab samples.
- τ^2 is the variance of the contribution of the individual grab samples to the measured subsamples. Ideally, each subsample has a contribution from each increment that went into its parent composite that is 1/n of the volume of the total subsample so that τ^2 is zero and, again, the last term of Equation 5.3 vanishes.
- θ^2 is the analytical error variance.

With these assumptions, the variance of the estimate of the mean is given by

$$\frac{\sigma^2}{nm} + \frac{\theta^2}{rs} + \frac{m}{rs}\frac{S-s}{S-1}\sigma^2\tau^2 \quad . \qquad (5.3)$$

Thus, when the variability among grab samples is the major component of variance (that is, when σ^2 is much larger than either θ^2 or τ^2), its effect on the estimate of the mean of a population can be reduced by compositing.

When collecting data for a risk-based decision, sampling designs that include physical averaging do not adversely affect the estimation of averages at the exposure unit scale, provided that the compositing design does not composite grab samples from different exposure units. ~~Composite sampling also has advantages for screening assessment (Boswell and Patil 1990, 0978). For example, if 18 increments are taken from a site and are used to create 3 composite samples and 2 subsamples are taken from each composite, 6 laboratory analyses are required. If all 6 composite subsampels indicate concentrations below SAL-related levels, the site can be proposed for NFA. For composite sampling, SALs must be modified to account for the averaging effect of compositing. If 6 increments go into one composite but only one is above SAL, their mean may be below the SAL; however, assuming increments of equal~~

~~weight, their mean cannot be below 1/6 of the SAL, so 1/6 of the SAL can be used for screening assessments based on composites of 6 increments.~~ **However, increased precision in the estimate of the exposure unit mean using composite sampling may be attained at the expense of spatial resolution within an exposure unit. If spatial resolution on a smaller scale is important, then the selection of increments to composite should be compatible with the scale of spatial resolution required. For example, composites may be formed within quadrants or smaller fractions of exposure units and should not combine strata likely to differ significantly (e.g., material from channels with mesa-top soils in cases where contaminants may have been released to outfalls).**

In some cases it is possible to recover spatial resolution using a two-phase analysis plan (Boswell and Patil 1990, 0978). In this way compositing may be used for screening assessment or "hot spot" detection. For example, 18 increments may be used to create 6 composite samples. If all 6 composite subsamples indicate concentrations below SAL-related levels, the site can be proposed for NFA. If this is not the case, then the increments that made up a high composite must be reanalyzed to determine whether the actual concentration in any one increment exceeded the SAL. (Note the expression "SAL-related levels." For composite samples SALs must be modified to account for the averaging effect of compositing. If 6 increments go into one composite but only one is above the SAL, their mean may be below the SAL. However, assuming increments of equal weights, their mean cannot be below 1/6 of the SAL, so 1/6 of the SAL can be used for screening assessments based on composites of 6 increments. Similar calculations are required when using composites for "hot spot" detection.) This procedure is not practical if the constituents of interest have short holding times, if adequate sample storage space is not available, or if time is a critical factor.

~~When the constituents of interest do not have short holding times and when adequate storage space is available, a fraction of each grab sample can be retained for individual analysis in cases where the composite sample produces a large measurement. Designs that exploit this possibility to improve spatial resolution or to characterize the upper tail of the contaminant distribution have been proposed in recent literature (Bloom 1992, 0979; Gore and Patil 1993, 0980). Continuing the previous example, if 1 composite indicates contamination above SAL-related levels, 6 more chemical analyses could be performed to determine the maximum concentration, which still results in a saving compared with performing 18 discrete analyses.~~

~~The appropriate number of increments used to form a composite can be optimized according to prior expectations of contaminant concentrations with respect to SALs. Many other design considerations can be incorporated in the selection of grab samples to be composited, e.g., spatial resolution can be maintained by compositing neighboring samples or samples from comparable depths or soil types instead of compositing at random across the site.~~

*****

11. Appendix I, 4.1 Proposed No Further Action and Deferred Action after Archival and Field Reconnaissance Investigations, p. I-18 -

The operation of a PRC after November, 1988 is not a reason for NFA of a site unless the New Mexico Environment Department agrees the site should not be in the HSWA portion of the RCRA permit.

Discussion

Text has been revised to allow for contingency, as shown below.

Proposed Text Change

> Step 2: In the second step, the OUPL determines whether the site should be addressed by another program and excluded from the ER Program. Any of the following conditions could exclude a PRS from the ER Program and **may** lead to a recommendation for NFA and delisting from the HSWA Module.

*****

12. Appendix O -

LANL needs to provide an explanation of the schedule as to what is the difference between the plan and estimate portions of the schedule.

Discussion

The "plan" portion of the schedule is the baseline. It represents a snapshot in time that indicates the anticipated schedule for accomplishing a particular milestone as a result of planning efforts. The "estimate" portion of the schedule represents the movement of milestones in time as actual activities are accomplished either ahead of or behind schedule. An explanatory note is added at the head of the table so stating

Proposed Text Change

The "plan" portion of the schedule is the baseline. It represents a snapshot in time that indicates the anticipated schedule for accomplishing a particular milestone as a result of planning efforts. The "estimate" portion of the schedule represents the movement of milestones in time as actual activities are accomplished either ahead of or behind schedule.

1.0 INTRODUCTION

The purpose of this appendix is to present the fundamental risk assessment methodology to be used to evaluate potential risks to human health associated with exposure to contaminants at, or released from, potential release sites (PRSs) at Los Alamos National Laboratory (the Laboratory). This methodology has been developed to establish the scope of the baseline risk assessment as an integral part of the Resource Conservation and Recovery Act (RCRA) facility investigation (RFI) process and to establish a consistent risk assessment approach to be applied at all operable units (OUs).

Risk-based decisions are used to support an RFI decision that no further action (NFA) is needed or to provide a point of comparison relative to which the benefits of a proposed corrective action can be evaluated. ~~Risk assessments address both long-term risks that remain after a proposed remedial alternative has been implemented and short-term risks to workers and the public during and after the proposed alternatives have been implemented.~~ **Risk assessments address both long-term and near-term risks. Long-term risk assessments evaluate the fate and transport of contaminants far into the future. Near-term risk assessments focus on the current human health impacts of contaminants on both the public and workers.**

The major elements of the risk assessment process addressed in this appendix are

- developing a set of chemical and radiological data for risk assessments. Contaminants of concern are identified by means of the screening process described in Appendix J. Once contaminants of concern have been identified, a set of chemical data for use in the risk assessment is developed.

- exposure assessment. The exposure assessment process identifies appropriate land use scenarios, receptors, contaminant transport pathways, exposure pathways, and intake parameters so that concentrations of contaminants in environmental media can be converted to chemical intake and radiological dose levels for human receptors.

- toxicity assessment. The toxicity assessment identifies significant toxic effects and endpoints so that appropriate toxicity values may be selected to quantify potential toxic effects.

- risk characterization. The final step of the process integrates the results of the exposure assessment and toxicity assessment so that the investigator can estimate excess cancer risks and noncarcinogenic toxicological impacts.

Each of the components of the risk assessment, as it will be applied at the Laboratory, is described in detail in the following sections.

2.0 DEVELOPING A SET OF CHEMICAL AND RADIOLOGICAL DATA AND INFORMATION FOR USE IN THE RISK

ASSESSMENT PROCESS

Evaluation of environmental data at the Laboratory begins with the screening assessment process, which involves comparing measured concentrations of potential contaminants of concern with screening action levels (SALs) ~~and, when necessary, background concentration distributions.~~ **and/or with background concentrations when background concentrations are below SALs.** SALs are media-specific concentration levels for potential contaminants that are derived using conservative risk-based criteria. SALs for nonradiological contaminants are based on the methodology presented in proposed Subpart S of 40 CFR 264 (EPA 1990, 0432) to calculate action levels. Radiological SALs are based on a 10-mrem/yr dose using a residential exposure scenario. Environmental sampling and analysis during the screening assessment process are designed to determine whether potential contaminants of concern are present at concentrations above SALs or background concentration distributions. For the purposes of performing risk assessment at the Laboratory, any contaminant present above a SAL or background ~~distribution is considered a contaminant of concern, which, in contrast to NFA, warrants further evaluation.~~ **distribution, when background is higher than SALs, is considered a contaminant of concern. A site with contaminants of concern requires further investigation.** A detailed description of the screening assessment process and the derivation of SALs is presented in Appendix J.

Environmental data from screening assessments or more detailed environmental investigations conducted during the RFI process at the Laboratory are used as input to risk assessments.

A set of site-specific chemical and radiological data that has undergone extensive quality assurance evaluations for use in risk assessment is critical to the success of evaluating the potential impacts on human health. Approaches to data evaluation are documented in Appendix H of this Installation Work Plan (IWP). The Environmental Restoration (ER) Program will follow Environmental Protection Agency (EPA) guidance (1989, 0305) to focus the data set that will be carried through each step of the risk assessment.

~~Distributions of background constituants are evaluated in a different manner from that used for a screening assessment, depneding on whether the consitutuant is peresumed to have a nonthreshold (carcinogenic) or thershold (noncarcinogenic) response. Therefore, chemical and radioctive carcinogens are evaluated both for incremental risks (risk above that associated with background concentrations) and total risk (which inlcudes background contributions), and noncarcinogens are evaluated only for total risk.~~

3.0 EXPOSURE ASSESSMENT

3.1 Introduction

Exposure assessment requires the framework of a conceptual site model, which identifies contaminant sources, potential current and future receptors, and exposure pathways that link the sources and receptors. Conceptual site models are used to examine the completeness of potential human exposure pathways. Any exposure pathway considered in quantitative human health risk assessment

must have each element present, i.e., a contaminant source and mechanism for release, a potential for human contact with the exposure medium, and a route of human exposure from that medium.

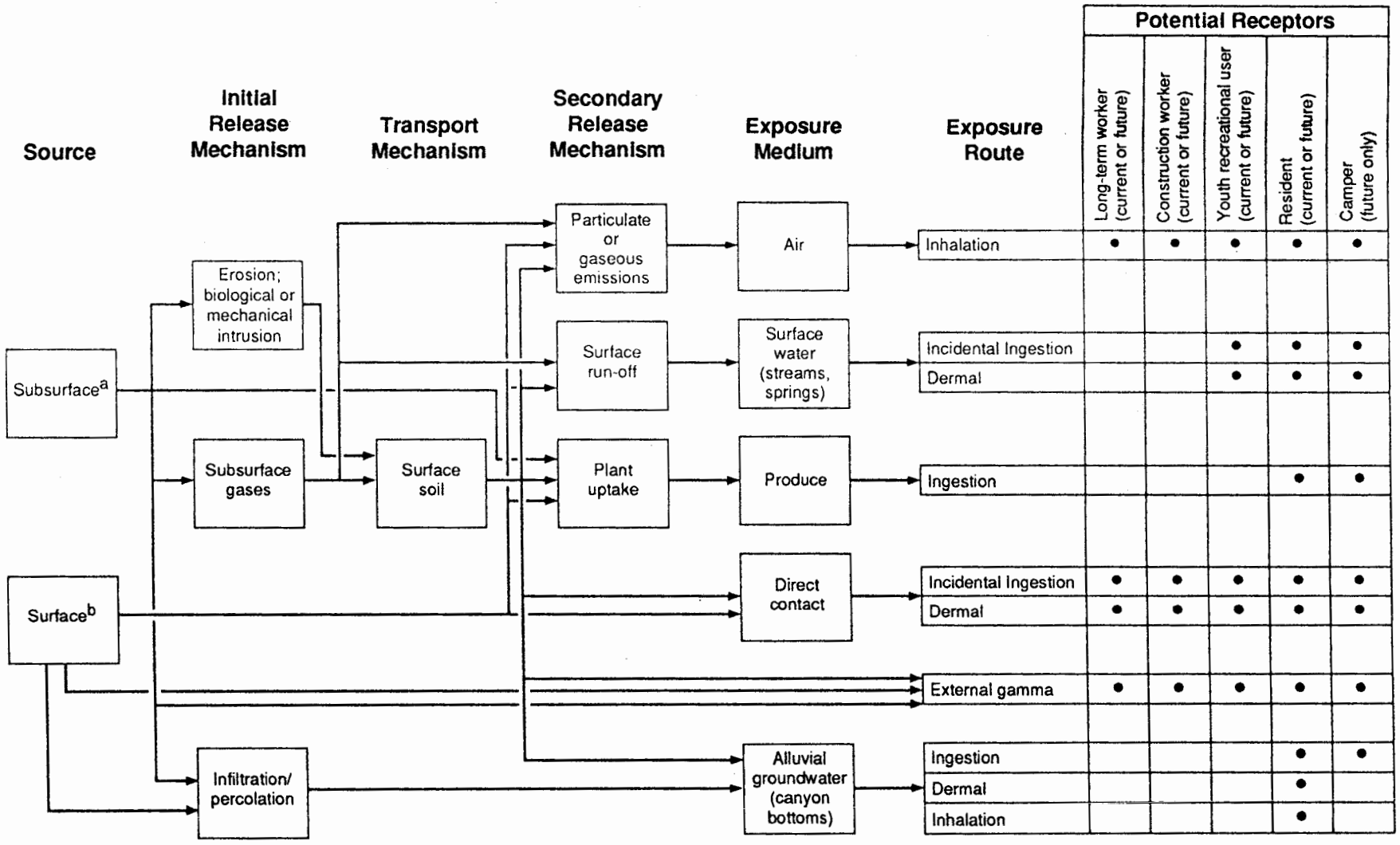
At the Laboratory, contaminants were originally deposited in either surface or subsurface soils, as shown in the conceptual site model (Figure K-1) and Table K-1. Through release mechanism pathways, contaminants were or could have been distributed in potentially contaminated media of concern, which include soil, air, edible plants, and water (Section 3.2.2). Receptors are exposed through contact with a contaminated medium by means of exposure routes (i.e., ingestion, inhalation, dermal contact). Land use and receptor-specific characteristics must be considered when identifying significant exposure routes for quantitative dose estimates. Section 3.2.3 identifies the most likely receptors for evaluation in ER Program risk assessments. This conceptual framework for the receptors is used to identify appropriate receptor-specific exposure routes (Section 3.2.4) for evaluation using quantitative risk assessment; these routes are indicated by bullets in Figure K-1.

Baseline risk assessments are conducted during the RFI to evaluate the potential threat to human health that a site may pose in the absence of any remedial action. Baseline risk assessments are risk assessments used to evaluate corrective actions that are conducted during the corrective measures study (CMS) or voluntary corrective action. In assessing exposure of current receptors, the investigator considers currently existing restrictions to areas of contamination (e.g., fencing, cover materials). Under the EPA's risk assessment guidance for evaluating potential future receptors, the investigator must assume that current access restrictions, such as fencing, are no longer in place and that disturbance of subsurface contamination may occur (EPA 1989, 0305). The ER Program generally applies this EPA guidance in developing risk assessment methodology.

PRSs at the Laboratory consist of the solid waste management units (areas in which solid wastes have been routinely and systematically released, including chemically contaminated wastes) identified in Module VIII of the Laboratory's permit to operate under RCRA [the Hazardous and Solid Waste Amendments (HSWA) Module] and other areas of concern (sites that potentially contain hazardous substances, such as radionuclides, but no hazardous constituents defined by RCRA) designated by the Laboratory's ER Program. For risk assessment, PRSs have been grouped in two categories: those located inside the Laboratory's current boundaries (Laboratory PRSs) and those located outside of the Laboratory's present boundaries (extra-Laboratory PRSs). The scope of this appendix is limited to exposures occurring within a given PRS or OU for both Laboratory and extra-Laboratory PRSs. If Phase I and/or Phase II data from RFIs indicate contaminant migration past OU or Laboratory boundaries, additional methods for evaluating effects upon distant receptors will be developed.

Each of the elements of the conceptual site model for the Laboratory is discussed in detail in this section, and detailed information is provided on estimating chemical intake and radiological dose for receptors likely to incur the "reasonable maximum exposure (RME)" (EPA 1989, 0305) to site contaminants under both current and future exposure conditions. In general, these receptors are those persons who potentially spend the most time near an area of contamination; in some instances, the receptor is selected because of substantial

physical contact with a contaminated medium (e.g., construction workers). Schematic representations of the conceptual site model for Laboratory and extra-Laboratory PRSs are presented in Figure K-1.



a. Examples of subsurface sources include landfills (buried wastes and structures), MDAs, underground storarge tanks, dry wells, and septic systems.

b. Examples of surface sources include firing site and drop tower experiments, surface disposal and burn areas, outfalls, cooling tower releases, buildings, spills, and overflows.

KEY: ● identifies exposure routes for evaluation using quantitative risk assessment

Figure K-3.1. Conceptual site model for LANL PRSs.

TABLE K-1

SUMMARY OF CONCEPTUAL MODEL ELEMENTS

PATHWAYS/MECHANISM	CONCEPT/HYPOTHESES
HISTORICAL SOURCES	• Operations/processes that contributed to the creation of the PRS (i.e., storage area, etc.).
INITIAL RELEASE MECHANISM	• Any spilling, leaking, pumping, pouring, emitting, emptying, discharging, injecting, leaching, dumping, or disposing into the environment.
TRANSPORT MECHANISMS	
Atmospheric Dispersion	• Entrainment is limited to chemicals in surface soils.
Particulate Dispersion	• Entrainment and deposition are controlled by soil properties, surface roughness, vegetative cover and terrain, as well as atmospheric conditions.
Volatilization	• Volatilization of volatile organic compounds occurs in surface soils, subsurface soils, and surface water.
Surface Water Run-off ~~Surface Water~~	• Surface water run-off is directed by natural topographic features or manmade diversions and flows toward the canyons. A topographic low can cause the water to pond on the mesa top; however, in most cases, the water flows into the canyon. • Chemical transport by surface run-off can occur in solution, sorbed to suspended sediments, or as the result of mass movement of heavier bed sediments. • Surface run-off may carry chemicals beyond the OU boundary. • Contaminated surface run-off may infiltrate the canyon-bottom alluvium.
~~Sediments~~ and Sediment Transport	• Surface soil erosion and sediment transport are functions of run-off intensity and soil properties. • Chemicals dispersed on the soil surface can be collected by surface water run-off and concentrated in sedimentation areas in drainages. • Erosion of drainage channels can extend the area of contaminant dispersal in the drainage.
Alluvial aquifers	• Surface run-off discharged to the canyons may infiltrate into sediments of channel alluvium.
Infiltration	• Infiltration into surface soils depends on the rate of precipitation or snowmelt, prior soil water status, depth of soil, and hydraulic properties of the soil. • Infiltration into the tuff depends on the unsaturated flow properties of the tuff. • Joints and fractures in the tuff may provide additional pathways for infiltration to enter the subsurface regime.
SECONDARY RELEASE MECHANISM	
Leaching	• Storm water/snowmelt can dissolve chemicals from soil or other solid media, making them available for ~~contact~~ leaching. • Water solubility of chemicals and their relative affinity for soil or other solid media affect the ability of leaching to cause a release. • Leaching and subsequent resorption can extend the area of contamination.
Soil erosion	• The erosion of surface soils is dependent on soil properties, vegetative cover, slope and aspect, exposure to the force of the wind, and precipitation intensity and frequency. • Depositional areas, as well as erosional areas exist, and erosive loss of soil may not occur in all locations. • Storm water run-off can mobilize soils/sediments, making them available for contact. • Storm intensity/frequency, physical properties of soils, topography, and ground cover determine the effectiveness of erosion as a release mechanism. • Erosion may also enlarge the contaminated area.

TABLE K-1 (continued)

SECONDARY

PATHWAYS/MECHANISM	CONCEPT/HYPOTHESES
~~POTENTIAL~~ RELEASE MECHANISM (continued)	
Mass Wasting	• The loss of rock from the canyon walls is a discontinuous, observable process. • The rate of the process is extremely slow.
Resuspension (wind suspension)	• Wind suspension of contaminated soil/sediment as dust makes chemicals available for contact via inhalation/ingestion. • Physical properties of soil (e.g., silt content, moisture content), wind speed, and size of exposed ground surface determine effectiveness of wind suspension as a release mechanism. • Wind suspension can enlarge the area of contamination and create additional exposure pathways, such as deposition on plants followed by plant consumption by humans/animals.
Excavation	• Manual or mechanical movement of contaminated soil during construction, remediation, or other activities makes contaminated soil available for dermal contact, ingestion, and inhalation as dust. • The method of excavation (i.e., type of equipment), physical properties of soil, weather conditions, and magnitude of excavation activity (i.e., depth and total area of excavation) influence the effectiveness of excavation as a release mechanism. • Excavation can increase or decrease the size of the contaminated area, depending on how the excavated material is handled.
EXPOSURE ROUTE	
Inhalation	• Vapors, aerosols, and particles (including dust) can be inhaled and absorbed by the lungs and mucous membranes. • Physical and chemical properties of airborne chemicals influence the degree of retention in the body after being inhaled.
Ingestion	• Ingestion of soil, water, food, and dust can lead to chemical intake via absorption in the gastrointestinal tract.
Direct Contact	• Some hazardous chemical constituents absorb through the skin when in contact with contaminated surfaces of soil, tuff, or rubble. • Physical and chemical properties of chemicals influence the degree of dermal absorption. • Factors such as skin moisture and temperature affect the degree of dermal absorption.
External Penetrating Radiation	• External, or whole body radiation, can occur through exposure to gamma-ray-emitting radionuclides that may be present in soil either directly through the soil or re-entrained dusts. • Exposure to penetrating radiation can also occur through inhalation or ingestion when radionuclide-contaminated soil or tuff surfaces erode and/or dusts become re-entrained.

3.2 Exposure Scenarios

3.2.1 Sources of Contamination

Potential sources of environmental contamination at the Laboratory are leaking tanks and other subsurface structures, spills, surface disposal, etc. Contaminants released during past operations could be transported from the medium in which a spill occurred to other media (e.g., surface soil contaminated by an operational release could provide a source of contamination for sediments in drainages). In some cases, the source itself may be the exposure point. Information is being gathered during the RFIs to identify possible sources of contamination and release mechanisms for each PRS at the Laboratory.

The original release sources at the Laboratory include but are not limited to

- subsurface sources: buried wastes [landfills, materials disposal areas (MDAs)], septic systems (tanks and drainlines), sanitary waste treatment facilities, sumps, underground storage tanks, and storm sewers.

- surface sources: firing-site experiments (including firing pads and chambers), blowdown from cooling towers, surface disposal units (e.g., burn areas), drop tower experiments, spills, storage areas, buildings, outfalls, debris, and fallout from stack emissions.

Many of the constituents found in the environment are naturally occurring (e.g., metals and polycyclic aromatic hydrocarbons); however, past Laboratory operations may have contributed to the release of several classes of compounds at levels that exceed background levels. The following paragraphs describe sources that may have contributed to environmental contamination.

Many types of volatile organic compounds (VOCs) have been used in Laboratory experiments and maintenance operations. Nonhalogenated VOCs were used as solvents and scintillation cocktails (e.g., acetone, toluene, and benzene) and may have been present in paints, paint thinners, and fuels. Halogenated VOCs were also used as solvents and degreasing agents (e.g., 1,1,1-trichloroethane and trichloroethene) and may have been present in research chemicals. These types of compounds may be found in buried waste, sumps, drainlines, underground storage tanks, septic systems, sanitary waste treatment facilities, spills, storage areas, buildings, and outfalls.

Semivolatile organic compounds (SVOCs) include a variety of chemical groups [e.g., polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), pesticides, herbicides, and others that may have been used in research applications]. Because PCBs have been used widely as coolants and lubricants in transformers, capacitors, and other electrical equipment, it is expected that transformers, MDAs, storage areas, and landfills contain PCBs. PAHs are typically generated as products of incomplete combustion or detonation of fuels and/or fuel oil. Thus, PAHs are most likely to be found in buried wastes, firing sites, burn areas, drop tower experiments, waste oils, spills, storage areas, buildings, outfalls, and debris. Sources of possible pesticide and herbicide contamination include maintenance and chemical storage areas ~~(such as magazines)~~ and any surface soil at locations where pesticides and/or herbicides were used regularly to

control pests and vegetation. Other SVOCs may be found at source areas such as buried waste sites, sumps, drainlines, spills, storage areas, buildings, outfalls, and debris.

Metals and related compounds were used for a variety of purposes in past Laboratory operations, and their presence is likely to be confirmed at many source areas. Metals are typically associated with plating activities (chromium and nickel), blowdown from cooling towers (chromates and molybdenates), photoprocessing operations (silver and cyanide), machining operations, electroprocessing operations (nickel, lead, silver), and activities associated with machining, testing, and disposing of explosives (beryllium and lead).

Some PRSs may contain high explosives (HE) used in formulations (e.g., TNT, RDX and HMX), their environmental degradation products (e.g., DNT and TNB), impurities of commercial HE, inert components (e.g., cyanuric acid), residual HE, and binders (e.g., nitrocellulose). Several types of explosives and their degradation products may be present in the soils and/or sediments at sites where explosives were or are currently processed, assembled, machined, stored, tested, or disposed. PRSs that contain sites of past HE activities are likely to include buried waste, sumps, drainlines, firing sites, surface disposal units, drop tower experiments, storage areas, burn areas, buildings, outfalls, and debris.

Radionuclides have been used in a variety of Laboratory studies and operations, including medical research (iodine, tritium); animal studies (lanthanum, ^{90}Sr, cobalt, cesium); reaction (iodine, tritium); detonation of test assemblies (^{140}La, depleted uranium); radiographic facilities for examination of HE charges (radium, ^{90}Sr, cobalt); weapons research and testing (plutonium, uranium, americium, tritium); and other research (fission and activation products). Sources that contain radionuclides are widespread across the Laboratory; these sources include septic tanks, dry wells, drainlines and drain fields, outfalls, storage areas, surface disposal areas, MDAs, landfills, stack emissions, and firing sites.

3.2.2 Exposure Pathways

Exposure pathways describe the courses that contamination takes from its source to a receptor. Exposure pathway analysis at the Laboratory links sources, locations, and types of environmental releases with locations and activity patterns of receptors to determine the significant pathways of exposure (Figure K-1). Exposure generally involves three elements: (1) a source and mechanism for contaminant release, (2) a pathway consisting of a retention or transport medium and a location of potential human contact with the contaminated medium (often referred to as the exposure point), and (3) a receptor who may receive a dose of a contaminant via ingestion, inhalation, or dermal contact at the exposure point. A medium contaminated as a result of a past release can be a contaminant source for other media, e.g., soil contamination can be a source for groundwater or surface water contamination. In some cases, the source itself is the exposure point, without any release to other media.

3.2.2.1 Contaminant Release Scenarios

Possible release mechanisms and potentially impacted environmental media are described in Table K-1. Under current conditions, the release mechanisms considered most likely to lead to human exposures to contaminants from PRSs are (1) dispersion of airborne particulates or gases from contaminated soil into the

air, (2) surface water run-off, which would redistribute soil contaminants into drainage pathways, (3) release of external gamma radiation, (4) generation of radon gas from a few PRSs that contain high concentrations of radon-generating contaminants (i.e., radium and thorium), and (5) volatilization for a few PRSs that contain high concentrations of tritium or volatile organic compounds.

The potential for contaminants to migrate from surface and subsurface soil areas to the main aquifer is considered low because of the nature of the tuff underlying the Laboratory, the high rate of evapotranspiration relative to precipitation in northern New Mexico, and the great depth to the main aquifer (600 ft or more). However, the Laboratory is continuing to investigate the possibility of more rapid contaminant transport along fractures in the tuff and the extent of perched groundwater. Alluvial aquifers in some canyon bottoms contain significant quantities of contaminants, but these aquifers do not appear to provide enough water to support domestic or recreational use. When groundwater may be a reasonable source of human exposure, ER Program risk assessments evaluate potential exposure to perched and alluvial groundwater on a PRS-specific basis.

3.2.2.2 Contaminant Fate and Transport

A variety of site-specific and chemical-specific factors influence the fate and transport of environmental contaminants at the Laboratory. Once a contaminant has been released to the receiving medium, it may be

- transported (e.g., carried downstream in water or transported through air),
- physically transformed (e.g., by volatilization),
- chemically transformed (e.g., by hydrolysis or oxidation),
- biologically transformed (e.g., by biodegradation), and/or
- accumulated in one or more medium, including the receiving medium.

To determine the fate of contaminants at a particular PRS, information on the contaminants' physical and chemical properties is required. Representative physical and chemical properties of contaminants of concern at the Laboratory include K_{OC} (the organic-carbon-partitioning coefficient), K_{OW} (the octanol/water-partitioning coefficient), Henry's Law Constant, solubility, vapor pressure, bioconcentration factor, and medium-specific half-life. Site-specific characteristics that may influence fate and transport must also be considered. Examples of site-specific characteristics include soil properties, such as moisture content, organic content, grain size, and cation exchange capacity.

In performing risk assessments, the ER Program uses applicable chemical and site-specific information to evaluate transport within and between media and retention or accumulation within a single medium. Data from environmental sampling will be used to identify media that are contaminated now. Fate and transport modeling may be required to identify media that may be contaminated now (for media not sampled) or in the future.

3.2.3 Characterization of Potentially Exposed Populations

3.2.3.1 Demographic Data

Los Alamos County includes the two residential areas that border the Laboratory—the towns of Los Alamos and White Rock. The Los Alamos townsite, at the northern border of the Laboratory, had an estimated population of 12,000 in 1990; the town of White Rock immediately to the east of the main Laboratory site had an estimated population of 7,200 in 1990. The percent of the Los Alamos County population under age 18 is approximately 26%; about 9% of the population is 65 years of age or older (Environmental Protection Group 1990, 0497).

Large tracts of land to the south, southeast, and west of the Laboratory are managed by the National Park Service (Bandelier National Monument) and the Forest Service (Santa Fe National Forest). Bandelier National Monument is used exclusively for recreation (e.g., hiking and camping), whereas, on Santa Fe National Forest lands, grazing, logging, and other uses, in addition to recreational use, are allowed. The Rio Grande forms the southeastern boundary of the Laboratory; the land on the opposite bank belongs to the Santa Fe National Forest. The San Ildefonso Pueblo borders the Laboratory to the northeast.

3.2.3.2 Land Use

Ideally, estimation of risks to human health and the environment at the Laboratory will be based on reasonable and site-specific land use and exposure assumptions that reflect stakeholder input. However, until this input has been provided and incorporated, current and future land use at all OUs is assumed be residential.

An approach that will be proposed to stakeholders involves a number of scenarios for two categories of PRS: those located inside and those located outside Laboratory boundaries; therefore, these two categories are discussed separately. The land use scenarios discussed in this appendix are generalized scenarios that are likely to apply to many of the OUs under evaluation. However, OU-specific differences in current and potential future land uses should be considered in individual risk assessments. This OU-specific information should be used to identify appropriate receptors for evaluation.

3.2.3.2.1 Current Land Use Scenarios

3.2.3.2.1.1 Extra-Laboratory OUs

Three OUs—1071, 1078, and 1079—address areas located outside current boundaries of the Laboratory. Two of the extra-Laboratory PRSs are located in the town of Los Alamos; the third contains many sites located on canyon sides or in canyon bottom areas to the north, south, and east of the town. Some PRSs comprising these OUs ~~continue~~ are still managed and used by the Laboratory; others are considered to have been remediated by past decontamination and decommissioning (D&D) projects and have been released for unrestricted use. Changes in regulatory standards and guidance since past remediation activities have led to listing some of the released areas as solid waste management units in the HSWA Module. Current land use categories for extra-Laboratory PRSs can be classified as (1) commercial/industrial, (2) recreational, and (3) residential.

3.2.3.2.1.2 Laboratory OUs

The most prevalent current land use for OUs and PRSs located within the Laboratory's current boundaries is Laboratory operations, which encompasses potential exposure of employees. Risk assessments for these OUs will evaluate exposure of current employees to residual contaminants from past operations using data collected under the ER Program. Risks from current operations (i.e., occupational exposures) are monitored by the operator groups and are not included in ER Program risk assessments.

Large portions of Laboratory property are currently undeveloped. Many of these properties are located in canyons; a few are located on mesa tops. Much of the undeveloped land controlled by the Laboratory is used as a buffer zone to provide security for Laboratory facilities and as a reserve for future construction (Chapter 2). The public is allowed limited access to some Laboratory areas (e.g., parts of Ancho, Mortandad, and Pueblo canyons, and an archaeological site near the White Rock Y).

Some of the currently undeveloped areas of the Laboratory were used for operations or waste disposal in the past, and contaminants may have migrated to other areas (e.g., surface run-off may have carried contaminants into canyons). Current use of undeveloped areas is limited to occasional recreational use (e.g., hiking or biking) by Laboratory employees, the public (for the few areas listed above), and trespassers.

3.2.3.2.2 Future Land Use Scenarios

3.2.3.2.2.1 Extra-Laboratory OUs

Future land use for most extra-Laboratory PRSs (including canyon bottom areas) is presumed to be residential because this use is generally the most restrictive. Physical restrictions for some extra-Laboratory areas preclude residential use (e.g., steep canyon sides, narrow bottoms); an appropriate future land use assumption for those areas is likely to be recreational.

3.2.3.2.2.2 Laboratory OUs

Future land use assumptions for OUs located on Laboratory property include (1) continued Laboratory operations and (2) recreational use (hiking, biking, and camping). Future residential land use for most Laboratory OUs is implausible because they are likely to be managed by DOE over the long term. Proposed RCRA Subpart S supports the use of plausible assumptions on future patterns of use, assuming that institutional controls such as deed restrictions are used as necessary to guide those future uses (EPA 1990, 0432). Most Laboratory land would involve a transfer of ownership to the neighboring National Park Service or National Forest Service, whose functions include managing recreational lands. Generally, future recreational land use is the most appropriate assumption for those canyon bottom areas unlikely to be used for Laboratory operations. For mesa tops, continued Laboratory operations is generally the appropriate future land use assumption.

Notwithstanding the general assumption that future residential use of Laboratory OUs is implausible, it is acknowledged that limited areas released for recreational use might actually be used for residential purposes such as ranger housing. The

Laboratory may need to re-evaluate these limited areas at the time a change in land use is proposed, using assumptions for a residential exposure scenario.

3.2.3.3 Potential Receptors

The appropriate current and future receptors likely to come into contact with contaminated media are being assessed (Figure K-1). The characteristics of these generalized receptors are summarized below. Receptor-specific assumptions may need to be modified, or additional alternate receptors may be needed based on OU-specific data; however, it is expected that the generalized receptor scenarios discussed below will be applicable for most ER Program risk assessments.

3.2.3.3.1 Receptors under Current Land Use Scenarios

Currently, the receptors that may be evaluated in risk assessments are long-term workers, construction workers, youth (i.e., age 10-18) recreational users, and residents. Only current residents of extra-Laboratory PRSs will be evaluated. The worker scenarios are appropriate for any properties currently used for commercial/industrial activities (e.g., commercial areas in extra-Laboratory OUs) or for continued Laboratory operations (Laboratory OUs). Exposure parameters for long-term workers should generally reflect the RME conditions, which include periods of outdoor work (discussed further in Section 3.4.2.1). PRS-specific data are used to determine whether a construction worker scenario, which evaluates subsurface contamination, should be assessed.

The youth recreational scenario is used to evaluate current use of canyonside and bottom areas at extra-Laboratory sites and for trespassing on Laboratory-controlled PRSs. This receptor is assumed to hike or bike frequently in the area encompassing a given PRS.

It is assumed that long-term institutional controls are imposed on the MDAs at the Laboratory to prevent contact with subsurface contamination. These areas have been or are currently used for subsurface disposal of radioactive and nonradioactive waste, and it can be assumed that these areas would present a significant risk if excavation were to occur. For these MDAs, ~~only current and future exposure of long-term workers to surface contamination will be assessed.~~ **current exposures will be determined for workers, and long-term risks will consider workers and the potential for migration of contaminants to an offsite receptor.**

3.2.3.3.2 Receptors under Future Land Use Scenarios

For most extra-Laboratory OUs, the future receptor evaluated will be a resident, regardless of current land use. In a limited number of extra-Laboratory areas (e.g., canyon sides) in which physical restrictions preclude residential use, a youth recreational user will be assessed.

For both mesa top and canyon bottom OUs whose past, current, or likely future use is Laboratory operations, the RME receptor is likely to be the long-term worker or the construction worker, both of which will be assessed as current receptors. However, the receptor most often assumed for canyon bottoms on Laboratory land will be a future camper (because most canyon bottom areas have not been used for Laboratory operations).

In general, it is not expected that canyonside areas will have subsurface contamination (i.e., greater than 2 ft in depth). Because of erosion, the soil in these areas generally does not extend to depths of greater than 2 ft. Accordingly, no current or future exposure to subsurface contamination on canyon sides needs to be considered. Also, the most plausible land use identified for canyonside areas is recreational use, which does not involve exposure to subsurface contamination. Therefore, the receptor evaluated for both current and future land use of canyonside areas will be the recreational youth user (i.e., hiker or biker). Exceptions can be addressed in PRS-specific risk assessments, as necessary.

To facilitate evaluation of changed land use (particularly loss of institutional control) and to account for possible redistribution of subsurface contaminants to the surface, future receptors may be evaluated by assuming exposure to subsurface contamination that has been redistributed to the soil surface. Use of subsurface contamination levels estimates the RME for future receptors when subsurface contamination levels are higher than those at the surface. Appropriate use of surface or subsurface contaminant levels to assess future receptors will need to be determined on a PRS-specific basis.

3.2.3.3.3 Application of Current and Future Scenarios in Risk Assessments

In some instances, it may be sufficient to evaluate only one receptor for a given PRS; this receptor represents the maximally exposed individual under both current and future conditions. An example would be a PRS currently used residentially and at which no subsurface contamination exists. Conversely, depending on PRS-specific contaminants of concern, depths of contamination, and other factors, it may be necessary to evaluate up to three receptors for a single PRS. An example is an operating Laboratory PRS at which both surface and subsurface contamination are present and at which subsurface contamination levels are higher than surface levels. For this example, a "current" long-term worker is assessed for the current exposure scenario (using surface contamination data), a construction worker is used to assess possible current exposure to subsurface contamination if excavation should take place, and a "future" long-term worker, who is assumed to be exposed to former subsurface contaminants that have been redistributed to the surface, is assessed for the future land use scenario.

The risk associated with contamination of environmental media depends on the actual dose to receptors engaged in the activities that occur in the selected land use scenario. This dose is proportional both to the temporal frequency and duration of exposure and to ~~the integral of~~ the contaminant concentration over a spatial exposure unit within which these activities take place. For example, under the camping scenario, exposure may occur annually over several years for a period of 2 to 4 weeks each year (the exposure frequency and duration), during which time the receptor can be expected to roam over an area of one-half to several acres (the exposure unit). Exposure units for the land use scenarios used by the ER Program are defined below in Section 3.3.2.

3.2.4 Summary of Receptor-Specific Exposure Routes

Exposure routes evaluated for all potential receptors are inhalation of particulate and/or gaseous emissions from soils or structural surfaces, direct contact with soil

or dust from structural surfaces (ingestion and dermal contact), and external gamma irradiation from soils or structural surfaces (Figure K-1).

Potential contact with perennial surface water in canyon streams is most likely for the future camper and the youth recreational user. Contact with perennial surface waters may also be an exposure route for future residential scenarios in a limited number of extra-Laboratory canyon bottom PRSs. Dermal uptake from surface water and incidental ingestion of surface water while wading are appropriate exposure routes for quantitative assessment. Inhalation does not need to be assessed because volatile contaminants are present in surface waters for only short periods before being dissipated to the air. The appropriateness of the surface water pathway should be determined on a PRS-specific basis; surface water may not be present in quantities large enough to warrant evaluation of this pathway. For most PRSs, it is unlikely that worker contact with surface water will be a significant exposure route because infiltration occurs rapidly after storm events and areas of Laboratory operations are distant from perennial canyon streams.

The need for assessment of exposure to temporary standing water after storm events will be assessed on a PRS-specific basis. However, surface water pools created by storm events are generally present only for short periods (e.g., about 2 h) before infiltration occurs, and storm events capable of generating these pools occur approximately 3 to 11 times per year. Therefore, potential for contact with this standing water is limited.

If field investigations reveal the presence of measurable gaseous emissions directly to air (i.e., via subsurface vapor migration from MDAs), exposure to these emissions will be evaluated. Based on OU data available to date, the location of emissions is assumed to be distant from work and residential areas, which indicate that the receptor evaluated will be the youth recreational user or the future camper; inhalation will be the exposure route assessed.

The ingestion exposure route may be evaluated for current and future residents who ingest homegrown produce and possibly for future campers who ingest wild edible plants (e.g., berries and nuts). PRS-specific information is needed to determine whether quantitative assessment of this pathway is appropriate. For example, extra-Laboratory PRSs currently used for residences may have a concrete cover over the contaminated area, ruling out gardening. In this instance, evaluation of ingestion for the current resident would not be appropriate, although the pathway would be evaluated for a future resident to address the possibility that the concrete cover is removed. Similarly, ingestion of wild foods by future campers should be assessed only if edible species are identified in the vicinity of the PRS.

Although ingestion of game animals such as elk is a possible current or future exposure route, the large territory over which these animals graze in semiarid climates makes the probability of significant uptake of contaminants from a single PRS small. Similarly, although grazing livestock on Laboratory lands may be allowed in the future, these animals would not be expected to have significant uptake of contaminants from any single PRS. Because the overall uptake from multiple PRSs may result in significant bioaccumulation of contaminants in game animals, the risks from ingestion of contaminated meat may be evaluated on a Laboratory-wide basis. Therefore, evaluation of the game and/or livestock ingestion pathway is outside the scope of an OU-specific risk assessment.

Similarly, although contaminants from the Laboratory may have reached the Rio Grande River via canyon streams, the potential for contaminant intake as the result of ingesting fish will be evaluated (if needed) on a Laboratory-wide basis.

For Laboratory OUs with associated alluvial aquifers in canyon bottoms, it is hypothesized that a well is drilled in the future to support recreational use of the areas. It is assumed that water supply to these wells is adequate to provide drinking water for future campers but not to provide bathing facilities. Therefore, ingestion is the only significant exposure pathway associated with groundwater to be evaluated for future campers. For extra-Laboratory OUs, there is a possibility that canyon bottoms will be used residentially and that a domestic well will be drilled to access alluvial groundwater, although this source may be limited. If necessary, ingestion of and dermal contact with contaminants in alluvial groundwater may be assumed for future residents in extra-Laboratory canyon bottom OUs (only when contaminants of concern have been identified in alluvial groundwater). Volatile contaminants are not likely to be present in alluvial groundwater because the most likely mechanism of contamination, run-off from contaminated surface soil, results in volatilization. However, should any volatiles be identified, inhalation of these contaminants during household use (e.g., showering) should be evaluated.

3.3 Estimation of Exposure Concentrations

The media-specific concentration of a contaminant at the location of exposure (i.e., the exposure concentration) must be estimated in order to evaluate the amount of human exposure that might be associated with a contaminant source. This section provides guidelines for estimating exposure concentrations for all contaminants of concern in soil, air, produce, on structural surfaces, and in surface water and alluvial aquifers.

Exposure concentrations can be estimated directly by using media-specific environmental sampling and analysis data or by using a combination of sampling data and environmental fate and transport calculations and models. For most risk assessments performed under the ER Program, some combination of sampling data and fate and transport modeling will be required to estimate exposure concentrations.

3.3.1 Methods

3.3.1.1 ~~Direct Estimation~~ Use of Sampling Data

Insofar as possible, sampling data will be used to directly estimate exposure concentrations for ER Program risk assessments. Direct use of sampling data is particularly applicable when exposure involves direct contact with the environmental medium of concern (e.g., direct contact with contaminated soils) and in cases where monitoring has occurred directly at an exposure point (e.g., a drinking water well). For these example exposure scenarios, sampling data provide the best estimate of current exposure concentrations. ~~The data quality objectives (DQO) process applied to the design of sampling plans during the phases of the RFI will result in the collection of environmental data appropriate for use in the risk assessment.~~

~~3.3.1.2 Modeling Approaches~~

In some instances, it may not be appropriate to use ~~direct~~ sampling data ~~alone~~ **directly** to estimate exposure concentrations, and some form of environmental fate and transport calculations or computer modeling applications will be required. Instances in which fate and transport calculations or modeling may be required include the following:

- Fate and transport calculations for contaminants will be required when exposure points are remote from contaminant sources and when release mechanisms exist to transport contaminants to an exposure point (e.g., transport of contaminants in groundwater to a receptor).

- Modeling will be required when temporal distribution of contaminants is needed and monitoring data are lacking. Typically, environmental sampling data give an indication of current conditions. Although there are situations in which it is reasonable to assume that contaminant concentrations will remain constant over time, in many cases, the time span over which data are taken during one-time sampling are not adequate to predict future exposure concentrations.

- When direct measurements of contaminant concentrations in air are unavailable, fate and transport modeling will be needed to determine air concentrations based on contaminated soil concentrations.

The level of effort required to estimate exposure concentrations depends on the type and quantity of data available, the level of detail required in the assessment, and the expected level of risk to human health and the environment. In general, estimating exposure concentrations involves analyzing direct sampling data and applying simple environmental fate and transport calculations. **Section 3.3.2 describes simple methods for estimating exposure concentrations for the exposure routes of inhalation of particulates and gas, ingestion of contaminated plants, and external gamma radiation, on the basis of concentrations of contaminants in surface and subsurface soil.**

More sophisticated modeling approaches than those presented there may be required, based on PRS-specific needs. For those efforts requiring a complicated modeling approach, a wide variety of EPA-approved contaminant fate and transport models is available for use in exposure assessments. Contaminant fate and transport models are applied in ER Program risk assessments on a PRS-specific basis. The procedure ~~to conduct~~ **for** complicated contaminant fate and transport modeling includes

- identifying and bounding the modeling problem and approach,

- selecting qualified experts to identify the appropriate models to be applied (this step may also include developing models to account for unique conditions at the Laboratory), and

- communicating with regulatory agencies to obtain approval of the modeling approach before conducting the modeling exercise.

The data quality objectives (DQO) process applied to the design of sampling plans during the phases of the RFI will result in the collection of environmental data appropriate for use in the risk assessment, whether direct or indirect modeling approaches are required.

3.3.1.2 Estimation of Reasonable Maximum Exposure (RME)

EPA (1989, 0305) suggests using an "upper 95% confidence limit on the arithmetic average concentration of contaminants at a site (or portion thereof)" as the appropriate statistic to represent the RME. In general, LANL will use the upper 95% confidence limit for the average concentration within an exposure unit as an initial point estimate of RME. Selection of statistical estimators for this average and for the corresponding 95% upper confidence limit depends on the statistical distribution of the observations. In Bayesian analyses, 95% upper confidence limits would be replaced by upper 95% probability bounds.

A recent Science Advisory Board review (EPA 1993, 1013) of the *Human Health Evaluation Manual* recommended that EPA move toward a distributional approach for calculating RMEs. This approach would require the development of distributions for the parameters that enter the calculation of individual exposures. These distributions, together with Monte Carlo simulation methods, would then be used to determine the distribution of exposure, and an appropriate quantile of the distribution (e.g., the 95th percentile) would be used as the RME. The ER Program at Los Alamos endorses this distributional approach to exposure and risk assessment, and distributions for environmental data and parameters used in calculating dose/intake should be developed. However, this represents a substantial investment of time and energy, and initially such distributions will be used primarily for quantitative uncertainty analysis.

3.3.1.3 Uncertainty in Exposure Concentration Estimates

Evaluating uncertainty is a very important component of the exposure assessment. EPA (1989, 0305) specifies that a qualitative discussion of uncertainty in the exposure analysis should include (1) a tabular presentation of the ranges of estimates of exposure concentrations and exposure parameters consistent with the data, the point values used in calculating the RME dose or intake, and a brief description of the rationale used in selecting the point value; and (2) a summary of major sources of uncertainty in the risk assessment that specifically addresses monitoring data, the models used to estimate exposure concentrations, and the ~~values~~ **point estimates** selected for intake ~~variables~~ **parameters**.

~~The same EPA guidance document suggests using an "upper 95% confidence limit on the arithmetic average [that is, the mean or expected value] concentration~~

~~of contaminants at a site (or portion thereof) [that is, within an exposure unit]" as the appropriate statistic for representing an RME. The words in brackets above interpret the phrases used in the EPA regulatory guidance in terms that are used in the Los Alamos ER Program. The arithmetic mean of the observations from an exposure unit is not the only reasonable estimator for the mean concentration in that exposure unit. Depending on observed distributional shapes, there may be substantially better estimators. The selection of an appropriate estimator of the mean is tailored to each situation.~~

~~The EPA guidance assumes a classical approach to statistical data analysis. If a Bayesian approach is taken instead, constructs such as confidence bounds can be replaced by probability intervals, and significance levels can be replaced by probabilities that hypotheses are true. Although the basic idea of estimating an RME with an upper bound for a mean concentration may remain, the Bayesian construction of such an estimate is quite different from the classical construction espoused in the current guidance. A Bayesian analysis demands construction of an exposure concentration distribution from which pertinent statistics (such as means, probability bounds, quantiles) may be calculated directly.~~

~~A recent Science Advisory Board review (EPA 1993, 1013) of the *Human Health Evaluation Manual* recommended that EPA move toward a distributional approach for calculating RMEs. This approach requires that the EPA develop distributions for the variables needed to calculate individual exposures; these distributions, together with Monte Carlo simulation methods, are used to determine distributions of exposure, and an appropriate quantile of the distribution (e.g., the 90th or the 95th percentile) could be used as the RME (e.g., the 90th percentile of the exposure distribution corresponds to an exposure concentration for which there is a 90% probability that an individual is exposed at no greater level). The ER Program at Los Alamos endorses this distributional approach to exposure and risk assessment, and, where possible, distributions for environmental data and parameters used in calculating dose/intake should be developed and presented. However, at present, EPA's policy is to use distributional approaches only as a means of assessing uncertainty.~~

~~In the following subsections, simple methods are described by which to estimate exposure concentrations for the exposure routes of inhalation of particulates and gas, ingestion of contaminated plants, and external gamma radiation, on the basis of concentrations of contaminants in surface and subsurface soil. More sophisticated modeling approaches than those presented may be required, based on PRS-specific needs.~~

In addition, particularly where a complex modeling approach to exposure estimation is required, quantitative assessments of the effect of uncertainty in all parameters, including both those estimated from site-specific data and those whose values are selected from other sources, is recommended. The Monte Carlo simulation method mentioned in the preceding section may indicate that the point estimates are in an extreme tail of the likely distribution of actual exposure concentrations. In this case a reevaluation of the selected point estimate may be required.

3.3.2 Exposure Concentrations for Various Exposure Pathways

3.3.2.1 Exposure Concentrations for Soil-Related Exposure Pathways

Soil data can be used to estimate exposure concentrations for the following pathways: incidental soil ingestion, dermal contact with soil, inhalation of particulates and gases, ingestion of contaminated plants, external gamma exposure, and inhalation of radon and its decay products. These pathways are addressed in the following subsections. Using measured air or food concentrations, gamma exposure rates, and radon concentrations (when available) is preferable to the use of soil data for the following pathways: inhalation of particulates and gases, ingestion of contaminated produce, external gamma exposure, and inhalation of radon.

Under the Laboratory's ER Program, surface soil samples are generally considered to include any data obtained within the top 6 in. of soil, although some surface samples may be obtained from up to 2 ft of soil (EPA 1989, 0088). In assessing the risk from incidental soil ingestion, dermal contact, and particulate inhalation, it is preferable to use soil samples taken from shallow depths (2 in.) because receptors are most likely to be exposed to near-surface soils. However, in instances in which contaminant levels in the upper 2 ft of soil are homogeneous or in which levels in the 0.5- to 2-ft horizon exceed those in the 0- to 0.5-ft horizon, use of data from the lower horizon will provide equal or higher estimates of dose or intake. Use of data from the 0.5- to 2-ft horizon also takes into account possible limited soil intrusion by receptors. Therefore, use of data from the 0.5- to 2-ft horizon to derive surface exposure concentrations is acceptable and should be used as necessary on a PRS-specific basis.

For the purposes of ER Program risk assessments, subsurface soil is considered to extend to a depth of 12 ft (EPA 1989, 0088). This depth is assumed under RCRA guidance because 12 ft is considered the maximum depth to which a basement is excavated, and therefore direct exposure to contaminants in soils deeper than 12 ft does not need to be assessed. (Contamination of soil or tuff found at depths of greater than 12 ft may be assessed by modeling potential for transport to groundwater.) For many mesa tops in the vicinity of the Laboratory, consolidated tuff is reached at depths of less than 12 ft. It is also uncommon for house foundations to be placed in the tuff in the Los Alamos area . Therefore, the total depth used to derive **direct** subsurface exposure concentrations may be adjusted on a PRS-specific basis.

An exposure unit is defined as the surface area over which a given receptor is assumed to receive a daily average exposure. Ideally, when data are collected for risk assessment, sampling plans are designed so that the samples collected represent the area over which they will be averaged. The statistics used to describe the contaminant distribution (the RME) for the exposure unit of concern are used to calculate the dose to a receptor (e.g., current EPA guidance suggests using a 95% upper-confidence bound of the arithmetic average). For example, a resident could be assumed to be randomly exposed to an area of soil equal to the average size of a residential property [about 1/8 acre (500 m^2) (EPA 1989, 0305)].

Precedent for using the exposure unit concept is given in EPA risk assessment guidance (EPA 1989, 0305). This EPA document states that "the area over

which the [receptor] activity is expected to occur should be considered when averaging the monitoring data.... For example, averaging soil data over an area the size of a residential backyard (e.g., 1/8 acre) may be most appropriate for evaluating residential soil pathways."

The land use scenario assumed for the PRS directly leads to the definition of the area over which a receptor receives a daily exposure. Thus, for ER Program risk assessments, each land use assumed (Section 3.2.2) has an associated exposure unit. Assumptions of appropriate exposure unit areas are made using site-specific information on land use patterns and likely activities for receptors being evaluated (EPA 1989, 0305). For ER Program risk assessments, assumed exposure unit areas are needed for four receptors: current and future residents, long-term workers, construction workers, and current and future recreational users. Assumed exposure unit areas for these receptors are presented in Attachment I and are discussed below. In general, PRSs on extra-Laboratory OUs are treated as residential and have an exposure unit of 500 m^2.

Recreational land use scenarios have exposure units equal to 1/2 acre (2,000 m^2). A 1/2-acre area is a reasonable estimate of the area covered by a child playing, a youth recreating, an adult exercising, or a family camping in the lands surrounding Laboratory work sites. The exposure unit concept for recreational users assumes that receptors locate a "preferred area" in which to spend the predominant portion of their recreational time and that this area encompasses the PRS being evaluated. The half-acre exposure unit is not necessarily assumed to be square but can be fit to the site-specific landscape of the PRS (e.g., a long rectangle for an outfall). A variable exposure unit shape for the recreational scenario is an appropriate assumption because potentially contaminated areas are formed over easily accessible land, where receptors tend to be exposed.

As part of the continued-Laboratory-operations land use scenario, commercial/industrial exposure scenarios have variable exposure unit sizes, depending on the type of PRS. PRSs modeled with a continued-Laboratory-operations scenario are generally on mesa tops and either have subsurface contamination or surface contamination only. For PRSs with a subsurface component for which a construction worker scenario is being evaluated, the exposure unit area equals the surface area of the contamination (i.e., it is assumed that the construction worker spends the entire exposure duration in the contaminated area and that no dilution from exposure to uncontaminated areas occurs). Construction worker receptors are likely to be exposed to a small area of contamination for a short time only. For PRSs with surface contamination, the appropriate receptors for the commercial/industrial exposure scenario are the long-term workers at a site. Under the long-term worker scenario, the exposure unit chosen equals 500 m^2, which is a reasonable estimate for a typical site worker who works indoors, taking lunch and breaks outdoors or spending up to 4 h outdoors.

When data have been collected without the appropriate exposure unit as a basis for averaging, an adjustment for the correct exposure unit must be incorporated before dose from soil sources can be calculated. Adjustment is made only for areas of contamination that are smaller than the exposure ~~unit because areas that are larger consider multiple exposure units (Appendix H describes proper methodology).~~ **units.** For PRSs that are smaller than the exposure unit, a correction factor can be applied to the dose or intake estimates to adjust them downward, accounting for the likelihood that, on average, the receptor will only

be exposed to the contaminated area for a fraction of the daily exposure time. Corrections for exposure unit size can be made with the "fraction from contaminated source" parameter (Section 3.4.2.8).

3.3.2.1.1 Exposure Concentrations for Incidental Soil Ingestion

To evaluate exposures to current receptors (e.g., long-term worker, youth recreational user, resident at extra-Laboratory PRS), contaminant concentrations in surface soils are used to calculate incidental soil ingestion intakes. An appropriate statistic is used as the RME for the incidental soil ingestion pathway (e.g., current EPA guidance suggests using the 95% upper confidence of the arithmetic average; **averages and the corresponding upper confidence bounds are evaluated for appropriate exposure units**).

Assessment of potential exposure to subsurface soil contaminants is needed for evaluating the construction worker scenario and future exposure scenarios. For the construction worker, an appropriate statistic is used as the RME for the incidental soil ingestion pathway (e.g., **again,** current EPA guidance suggests using the 95% upper confidence of the arithmetic average**, and averages are determined for the appropriate exposure units**).

Subsurface soil exposure concentrations can be used to evaluate exposures of future receptors exposed over the long term (i.e., workers, recreational users, residents of extra-Laboratory properties). To estimate exposures for these future receptors, it is assumed that subsurface soil contaminants have been redistributed to the soil surface before these receptors can be exposed. PRS-specific data, such as soil depth, volume of contaminated soil, likely future land use, and likely size of structure to be built on the property, are needed to determine the surface area over which the contaminated soil will be distributed. Assuming an appropriate soil thickness (e.g., 6 in.), area and depth of excavation corresponding to the likely type of structure to be built, and the appropriate exposure unit for the future receptor of interest, "derived" surface soil exposure concentrations for the future receptor can be estimated. This method is similar to one described by Reynolds et al. (1990, 1009) for estimating potential surface contaminant concentrations on the basis of measured subsurface levels. To distinguish these derived surface concentrations from measured surface concentrations in this text, the derived surface concentrations are termed "subsurface exposure concentrations."

In instances where receptor-specific exposure concentrations from surface soil data exceed exposure concentrations calculated from subsurface soil data, the risk assessor may decide to use surface soil data to assess risks to future receptors in order to ensure that risks are not underestimated. This decision is based on the likelihood of future excavation and should consider site-specific data and probable future land uses.

3.3.2.1.2 Exposure Concentrations for Dermal Contact with Soils

With few exceptions, the dermal absorption exposure route does not apply for most radionuclide contaminants in soil because the skin provides an effective barrier to the absorption of most radionuclides. Even absorption of tritium from soil water is likely to be much lower than absorption from immersion in a body of water, given equal concentrations of tritium. In the Los Alamos area, where soil moisture is generally low, dermal uptake of tritium from soil is assumed to be

negligible.

It is possible that the presence of radioactively tagged organic contaminants will result in a dermal exposure route for radionuclides. The significance, if any, of this exposure route for radionuclides should be evaluated on a PRS-specific basis.

For chemicals, the significance of dermal absorption from soils is highly dependent on the identity of contaminants of concern for specific PRSs. In general, organic contaminants are more easily absorbed through the skin than are inorganic contaminants. Chemical-specific data for the PRS-specific contaminants of concern should be used to determine whether evaluation of this pathway is appropriate. When evaluation is necessary, exposure concentrations are the same as those for the soil ingestion pathway (i.e., surface exposure concentrations for current receptors and subsurface exposure concentrations for future receptors, both in units of milligrams per kilogram).

3.3.2.1.3 Exposure Concentrations for the Inhalation of Soil-Derived Particulates and Gases

3.3.2.1.3.1 Particulates

The inhalation-of-soil-derived-particulates exposure route includes the inhalation of contaminants adhering to dust particles that become airborne as the result of wind erosion and mechanical disturbances of contaminated soil. Exposure concentrations of airborne contaminants depend to some extent on the exposure scenario being assessed. For example, the concentrations of particulates inhaled by an indoor office worker are expected to be significantly lower than the concentration of particulates inhaled by an outdoor construction worker because particulate levels are higher in construction zones. The concentrations of airborne contaminants may be obtained by direct measurement or by modeling the emission and dispersion from the source (i.e., soil).

Generally, measured air concentrations of particulate-associated radionuclide and chemical contaminants will not be available for use in ER Program risk assessments. Emissions resulting from suspension of soils by wind erosion or vehicular disturbance can be estimated using site-specific input parameters and any of a number of models (e.g., models described in EPA's *Superfund Exposure Assessment Manual* (EPA 1988, 0747) or models from the Air Superfund National Technical Guidance Series (EPA 1989, 1011). When necessary, models can also be used to estimate dispersion from the source to downwind receptor locations. The need for air emission and dispersion modeling is determined on an OU- or PRS-specific basis.

In some instances, it is sufficient to estimate emissions from the contaminant source simply by obtaining a site-specific value for mass loading of particulates to air and assuming that radionuclide and chemical contaminant levels in these particulates are equal to those measured in soils. Average particulate level values can be used for most exposure scenarios and may be obtained from the meteorologic station nearest the site. An average value for the Los Alamos area has been estimated as 0.09 mg/m^3 (Environmental Protection Group 1990, 0497). However, for the construction worker scenario, the assumed particulate level is higher because of receptor activities that disturb the soil. If data for construction sites at the Laboratory are not available, a particulate level of 15

mg/m^3, which is the Occupational Safety and Health Administration's limit for nuisance dust (OSHA 1991, 0610), can be assumed for construction worker scenarios. It may also be necessary to assume a particulate level higher than the average level for some recreational users (e.g., bikers), depending on whether their activities are expected to generate dusts. Appropriate assumptions based on PRS-specific particulate levels are made in individual ER Program risk assessments.

Not all particulate matter is respirable by humans; therefore, an adjustment is needed to account for the respirable portion of the particulates. Paustenbach (1989, 1007) gives a range of 30% to 50% as the respirable fraction of suspended particulates. Unless site-specific data on levels of respirable particulates are available, it is recommended that 50% of air particulate matter be assumed to be respirable. Air exposure concentrations for particulate-associated contaminants can be estimated as follows:

$$\text{Air EC} = \text{Soil EC} \times \text{PC} \times \text{CF} \times \text{RF} \quad , \qquad (1)$$

where

Air EC = exposure concentration in exposure concentration in air (picocuries per cubic meter for radionuclides, milligrams per cubic meter for chemicals),

Soil EC = appropriate surface or subsurface soil exposure concentration (picocuries per gram for radionuclides, milligrams per kilogram for chemicals),

PC = particulate concentration (milligrams per cubic meter),

CF = conversion factor (10^{-3} g/mg for radionuclides, 10^{-6} kg/mg for chemicals), and

RF = respirable fraction (0.5).

3.3.2.1.3.2 Gases

Inhalation of volatile contaminants from soils is not a pathway of concern for most receptors because these contaminants volatilize quickly from surface soils; therefore, exposure is relatively short-term. EPA guidance (EPA 1991, 0302) does suggest evaluating this pathway for workers exposed to subsurface contaminants [e.g., construction worker scenarios for ER Program risk assessments (EPA 1991, 0302)]. The only volatile radiological contaminants of concern are radon and tritium. Radon is addressed separately (Section 3.3.2.1.6). The concentration for tritium in air will be evaluated on a PRS-specific basis. Air exposure concentrations for volatile chemicals can be estimated as follows:

$$\text{Air EC} = \text{Soil EC} \times (1/\text{VF}) \ , \qquad (2)$$

where

Air EC = exposure concentration of volatile contaminant in air (milligrams per cubic meter),

Soil EC = appropriate subsurface soil exposure concentration (milligrams per kilogram), and

VF = soil-to-air volatilization factor, chemical-specific (cubic meters per kilogram).

Available soil-to-air volatilization factors for potential chemicals of concern are given in Appendix J. The equation for deriving volatilization factors is given in EPA guidance (EPA 1991, 0302).

3.3.2.1.4 Exposure Concentrations for Ingestion of Contaminated Foods

The ingestion of edible plants grown in contaminated soils presents a potential exposure route for current and future residents at extra-Laboratory PRSs and recreational users. When ~~available, an appropriate statistic will be used as the RME for ingestion of edible vegetation~~ **concentrations of contaminants in edible vegetation are available, an appropriate statistic will be used as the RME for ingestion** (e.g., current EPA guidance suggests using the 95% upper-confidence bound of the arithmetic average). Alternatively, concentrations in foods can be modeled from soil concentrations; food contamination resulting from root uptake is modeled separately from that resulting from foliar deposition.

3.3.2.1.4.1 Root Uptake

Root uptake will be the primary contributor to contaminant levels for nonleafy food species such as potatoes, carrots, and berries. Soil data can be used to predict contaminant concentrations resulting from root uptake in edible plants; however, considerable uncertainty is involved. Contaminant-specific data relating concentrations in edible produce to soil concentrations are generally unavailable, especially for chemical contaminants. Soil-to-plant transfer factors available in the literature are derived from data used for evaluating radiological exposure from weapons testing fallout and may not be appropriate for evaluating chemical toxicity. Additionally, the transfer of contaminants of concern from soil to edible plants depends on many factors, such as plant species, pH of the soil, and chemical form of the contaminant (Gough et al. 1979, 0998). For some substances, a certain soil level is toxic to some plant species and, beyond that level, growth does not occur.

Nonetheless, a general method for predicting contaminant concentrations in edible plants from soil concentrations is provided here for use in deriving baseline risk estimates. This method may result in overestimating concentrations in foods. If the resulting calculated risks are **unrealistically** high, a more thorough assessment may be required, based on PRS-specific contaminants of concern and data on soil types, growing conditions, and types of edible plants likely to

grow in the Los Alamos area.

Although not entirely applicable for every species of edible plant, soil-to-plant transfer factors have been assigned to many radioactive and inorganic substances (Yu et al. 1993, 1014; Ng et al. 1968, 1016; 1982, 1006). Plant uptake of organic compounds may be estimated using the procedure described by Briggs et al. (1982, 0995). Alternatively, soil-to-plant transfer factors for organic substances have also been compiled (Strenge and Peterson 1989, 0837).

Exposure concentrations in edible plants may be calculated as follows:

$$\text{Produce EC} = \text{Soil EC} \times \text{TF} \times \text{RDP} \quad , \qquad (3)$$

where

Produce EC = exposure concentration in edible portion of plant picocuries per gram for radionuclides, milligrams per kilogram for chemicals),

Soil EC = exposure concentration in soil (picocuries per gram for radionuclides, milligrams per kilogram for chemicals),

TF = soil-to-plant transfer factor, radionuclide- and chemical-specific (unitless), and

RDP = root depth parameter, PRS-specific (unitless).

The root depth parameter is used to assess the fraction of root length that is in contact with contaminated soil, which essentially provides a dilution factor for root uptake. For example, if contamination is found only in the top 15 cm of soil and the active root depth is 100 cm, the root depth parameter is 0.15 (i.e., the fraction of contaminant in soil transferred to the plant is estimated to be 15% of the amount that would be transferred to a plant with a root depth of 15 cm or less). Site-specific values of root depths for local edible plants should be used when available. If site-specific values for the species of concern are not available, generic values can be used (e.g., Zipparo et al. 1993, 1015). This root-depth dilution factor method is recommended for the produce ingestion pathway if only surface soil [the top 6 in. (15 cm)] is contaminated.

Another approach for estimating the exposure concentration in edible plants resulting from root uptake is to use the effective soil exposure concentration over the soil depth equivalent to the root length and to assume that the root depth parameter in Equation 3 is equal to 1. In cases where subsurface soil is contaminated, this weighted average method is recommended.

3.3.2.1.4.2 Foliar Deposition

Foliar deposition may be a significant contributor to contaminant levels for leafy edible species such as lettuce and spinach. A method has been developed that uses contaminant-independent "mass-loading" transfer factors from soil to air to edible portions of plants in estimating foliar deposition (Kennedy and Strenge

1993, 1002). These mass-loading transfer factors are recommended for a variety of plant types and are added to contaminant-specific root uptake transfer factors to obtain total soil-to-plant transfer factors. (Both factors must be based on dry plant weight). The factors are then multiplied by dry-plant-weight to wet-plant-weight conversion factors. Finally, multiplication by the soil concentration yields the contaminant concentration in edible portions of the plant. The equation for this calculation is

$$\text{Produce EC} = CF \times [(\text{Soil EC}_{eff} \times TF_{RT}) + (\text{Soil EC}_{surf} \times TF_{FD})] \times W \quad (4)$$

where

Produce EC = concentration in edible portions of plant (picocuries per kilogram or milligrams per kilogram),

CF = unit conversion factor (radionuclides only, 1,000 g/kg),

Soil EC_{eff} = effective soil exposure concentration over length of roots (picocuries per gram or milligrams per kilogram),

Soil EC_{surf} = surface soil exposure concentration (picocuries per gram or milligrams per kilogram),

TF_{RT} = soil-to-plant transfer factor from root uptake [picocuries (or milligrams) per kilogram dry plant per picocurie (or milligrams) per kilogram soil],

TF_{FD} = soil-to-plant transfer factor from foliar deposition [picocuries (or milligrams) per kilogram dry plant per picocurie (or milligram) per kilogram soil], and

W = dry-plant-weight to wet-plant-weight conversion factor (kilograms dry weight per kilogram wet weight).

The above methodology, although inherently simple, is based on fixed assumptions of airborne dust, crop yields, removal rates, etc. These assumptions should be evaluated for applicability to conditions at the Laboratory before using this model.

3.3.2.1.5 Exposure Concentrations for External Gamma Exposure

When possible, direct measurement of external gamma exposure rates should be used to estimate external gamma exposure concentrations. Exposure to gamma radiation in air is measured by the quantity of ion pairs (ionization) formed per unit mass of air by gamma rays emitted from contaminated air, soil, and structural materials. For onsite assessments, the contribution from gamma emitters in air is

generally negligible compared with the other two sources. Exposure to external radiation may be estimated from the concentrations of emitters in the source, taking into account attenuation and source/receptor geometries. However, directly measuring exposure via the air pathway at the receptor location (in units of milliroentgens per hour) results in a faster, more accurate assessment of dose than measuring the concentrations in the source medium.

When direct measurement of gamma exposure rates is not practical (e.g., for use in assessing future exposure from contaminated subsurface soil), soil concentrations of specific radionuclide contaminants of concern can be used to estimate gamma exposure rates (external gamma irradiation from structural surfaces is addressed in Section 3.3.2.3). Other minor exposure routes, such as external radiation from immersion in contaminated dust and water, do not contribute significantly to the external gamma exposure of any of the postulated receptors. Therefore, exposure concentrations in air and water are not needed to assess the risk to receptors from the external exposure route.

To assess external gamma irradiation from soils, soil exposure concentrations for the appropriate soil depths should be used. For current receptors, surface soil data (i.e., samples collected within 2 ft of the surface) should be used as the exposure concentrations. For future receptors, subsurface soil data will be needed to assess the risk both during activities involving soil excavation and after subsurface soil contaminants are redistributed to the surface.

If the receptor is postulated to be indoors a significant portion of the time (i.e., long-term worker or resident) and the source of contamination is soil, shielding by uncontaminated structural materials should be taken into account to more realistically assess receptor exposure. The recommended shielding factor is 0.7 (Yu et al. 1993, 1014). Also, the presence of any uncontaminated cover material should be accounted for in calculating the exposure rates. To conservatively assess the risks from the external exposure route, the receptor is assumed to be located at the center of the contaminated area.

3.3.2.1.6 Exposure Concentrations for the Inhalation of Radon

The presence of ^{226}Ra (and other members of the natural uranium chain) or ^{228}Th (and other members of the natural thorium decay chain) in soil or structural materials results in the generation of ^{222}Rn or ^{220}Rn gas, respectively. Exposure concentrations in air of either radon isotope may be estimated from the concentrations of radium or thorium in soil by using an appropriate model such as RESRAD (Attachment II to this appendix) or by measuring them directly. Modeling is recommended if radon precursors are detected and measurements of airborne radon are not available. Modeling is also required when estimating the dose in a future use scenario that involves indoor occupancy of buildings that do not currently exist (Section 3.4.3.6). In such cases, surface or subsurface soil concentrations for the appropriate radium or thorium isotope are used as exposure concentrations.

When radon measurements are available, they should be used in calculating exposure concentrations. Radon measurements are usually reported in units of picocurie per liter. To calculate the radon dose, however, these concentrations must be corrected for the ingrowth of radon decay products, which contribute most of the dose. The radon concentration unit that accounts for this contribution is defined as the working level (WL), which is equal to the

concentration of short-lived decay products that release 1.3 x 10^5 MeV of potential alpha energy per liter of air. The concentrations of decay products in air are seldom in equilibrium with radon. To convert concentrations of radon from picocuries per liter to concentration of daughters in working levels, the degree of equilibrium between the former and the latter must first be estimated. Equilibrium factors depend on factors such as radionuclide half-lives, radon emanation and ventilation rates, fraction of decay products attached to particulates, plateout of decay products, and particulate deposition rates. The following equation is used to convert the radon concentrations to decay product working levels (i.e., the exposure concentrations):

$$C_{WL} = C_{pCi/l} \times CF\ EF\ , \qquad (5)$$

where

C_{WL} = concentration of radon decay products (working level),

$C_{pCi/L}$ = radon concentration (picocuries per liter),

CF = conversion factor (working level per picocurie per liter), and

EF = equilibrium factor (unitless).

The conversion factors for ^{222}Rn and ^{220}Rn are 0.01 and 0.13 WL per picocurie per liter, respectively (ICRP 1981, 1000). In the absence of site-specific data, an equilibrium factor of 0.5 is recommended (National Research Council 1991, 1005).

3.3.2.2 Exposure Concentrations for Water Pathways

3.3.2.2.1 Ingestion of Groundwater or Surface Water

When contaminants of concern are identified in potable alluvial groundwater, it is appropriate to assess ingestion of groundwater in canyon bottoms. Water is considered potable if there is sufficient yield to support the assumed use (e.g., domestic or recreational use) and adequate quality (e.g., the water is low in total dissolved solids). The receptor to be evaluated for this pathway depends on the location of the PRS; a future resident is evaluated for extra-Laboratory PRSs, and a future camper is evaluated for Laboratory PRSs. No actual current use of alluvial groundwater is known.

When sufficient numbers of samples exist, an appropriate statistic is used to represent the RME in alluvial groundwater. For example, current EPA guidance suggests using the 95% upper-confidence bound of the arithmetic average. However, it is expected that data for alluvial groundwater will be limited to one or two rounds of sampling from only a few monitoring wells for most OUs. Given such limited data, it may be necessary to use the maximum level of each contaminant of concern in any monitoring well as the exposure concentration. The units are picocuries per liter for radionuclides and milligrams per liter for chemicals.

Perennial surface water bodies are generally limited to streams in a few canyon

bottoms. Assessment of potential exposure to contaminants in these surface waters includes an assumption of limited ingestion during recreational use (e.g., wading) by the youth recreational user, future camper, or future resident. The exposure concentrations are an appropriate statistic to represent the RME in alluvial groundwater. The units are picocuries per liter for radionuclides and milligrams per liter for chemicals.

3.3.2.2.2 Dermal Absorption from Groundwater or Surface Water

Dermal absorption of most radionuclides from water is negligible compared with other exposure pathways and is not assessed. However, absorption of tritium from water may be significant (Pinson and Langham 1957, 1008). The tritium exposure concentration used to evaluate dermal absorption from the water pathway is simply the concentration of tritium in water in units of picocuries per liter.

After chemical contaminants of concern have been identified for a PRS, their potential for dermal absorption from water should first be evaluated qualitatively. If a significant potential for dermal absorption exists, either an appropriate statistic is used to represent the RME in water, or the maximum contaminant levels detected are used as exposure concentrations, depending on the numbers of samples available. The units are milligrams per liter.

3.3.2.2.3 Inhalation from Household Use of Groundwater

In general, this pathway is unlikely to be applicable for ER Program assessments. The most likely mechanism for contamination of alluvial groundwater in canyon bottoms is via run-off from Laboratory operations on mesa tops. This mechanism would lead to volatilization rather than to leaching of volatile contaminants to groundwater. However, because there may be some sources of volatile contaminants in canyon bottoms, this pathway cannot be completely ruled out.

If volatile contaminants are detected in alluvial groundwater, inhalation exposure to volatiles will only be a potentially significant pathway for extra-Laboratory areas in which future residential use is possible (future campers in canyon bottoms would not use the groundwater for bathing or showering). For future residents, showering is likely to be the most significant exposure source during household use. Because of the limited application of this pathway, equations for the derivation of air exposure concentrations and dose/intake while showering are not presented. The need for evaluating this pathway should be determined on a PRS-specific basis. If needed, methods for evaluating exposure while showering are available [e.g., Byard (1989, 0996)].

3.3.2.3 Exposure to Contaminated Building Surfaces

3.3.2.3.1 Radiological Contaminants

Building surfaces that have been contaminated as a result of past Laboratory operations may result in worker exposures to these contaminants. It is assumed that these buildings will remain in use for industrial/commercial purposes in the future~~. Therefore, only the long-term-worker scenarios need be considered.~~ **or be demolished during D&D. Long-term-worker scenarios need to be considered for buildings that remain in operation.** Demolishing and renovating contaminated buildings may result in higher short-term exposures

to construction personnel; however, **these** health impacts are likely to be lower than exposures from long-term building use. Worker exposure to radiological contaminants in buildings may occur from one or more of the following pathways (Kennedy and Strenge 1993, 1002):

- dermal exposure to and inhalation of gamma radiation emitted from building surfaces,
- ingestion of dust from contaminated building surfaces, and
- inhalation of air contaminated by the entrainment of radionuclides from building surfaces.

For the external exposure route, actual measurement of exposure rates at locations where individuals are likely to work provide the most precise estimate of worker doses. Alternatively, surface or volumetric concentrations of total radioactivity (loose plus fixed) in disintegrations per minute per 100 cm^2 or picocuries per gram may be used in conjunction with shielding models or external dose rate factors to estimate exposure rates at receptor locations.

For ingestion of radionuclides, the exposure concentration is the measured concentration of loose surface contamination in disintegrations per minute per 100 cm^2.

The exposure concentration for the inhalation pathway resulting from the building occupancy scenario may be obtained by direct measurement of airborne contaminants (picocuries per cubic meter for radionuclides). When no measurements are available, the exposure concentration may be obtained by modeling the resuspension of contaminants from building surfaces (Kennedy and Strenge 1993, 1002).

3.3.2.3.2 Chemical Contaminants

Exposure to chemical contaminants on indoor surfaces may occur via ingestion, dermal contact, and inhalation. Ingestion and dermal contact are possible only when surfaces are accessible for direct contact. Potential for inhalation depends on the extent of dust resuspension. For ER Program risk assessments, volatile contaminants are presumed to have dissipated.

A recent review of the literature found no correlation between chemical contaminant concentrations in surface wipe samples (for which methods are specified under the Occupational Safety and Health Act) and air concentrations of those chemicals (Caplan 1993, 0997). Caplan attributed the lack of correlation to the dependence of resuspension on several factors, including properties of dusts (e.g., adhesiveness, particle size distribution, and density); properties of surfaces (e.g., macro- and microstructure, adhesive properties, porosity); and variable activities in buildings (e.g., foot traffic, vehicle traffic, vibrations, air exchange). Because an accurate resuspension factor for chemicals from surfaces to air is not available, monitoring data will likely be needed for estimating air exposure concentrations caused by surface contamination. Wipe samples will be used to identify contaminants that might be present in air.

Estimates of potential ingestion and/or dermal contact exposure via contaminated indoor surfaces can be made using wipe sample data and assumptions on extent

of contact. The receptor of concern for this exposure pathway in ER Program risk assessments is a long-term worker. The need for assessing this pathway should be evaluated on a PRS-specific basis. Examples of risk calculations for contaminated building surfaces in the literature are scarce, although such an analysis has been conducted for dermal contact with PCBs on contaminated surfaces (Rosenbaum et al. 1990, 1010). For the ingestion pathway, methods used for assessing radiological contamination in buildings (Kennedy and Strenge 1993, 1002) could be adapted to address chemical contamination.

3.4 Estimation of Radiological Dose and Chemical Intake

3.4.1 Methods

3.4.1.1 General Exposure Parameters

Estimates of exposure are based on the contaminant concentrations at exposure locations (Section 3.3) and scenario-specific assumptions and exposure parameters. Scenario-specific assumptions include factors such as frequency and duration of exposure to a contaminated medium by a potential receptor. Exposure parameters are associated with the route of exposure (e.g., ingestion and inhalation). The assumptions and intake factors for the exposure scenarios applicable for ER Program risk assessments are presented in Section 3.4.2. The approach and equations for determining contaminant doses and intakes at the potential exposure locations are given in Sections 3.4.3-3.4.5.

The EPA's risk assessment guidance recommends the quantification of intakes for each route of exposure from an exposure medium (EPA 1989, 0305). Exposure estimates are based on the RME expected to occur under current and future land use conditions (Sections 3.2.2. and 3.2.3). The RME is defined as the highest exposure that can reasonably be expected to occur at a site. Scenario assumptions and intake parameters used to estimate the RME are based, to the extent possible, on values provided in the *Exposure Factors Handbook* (EPA 1989, 0304), in *Human Health Evaluation Manual* (EPA 1989, 0305), and in a supplement to that manual (EPA 1989, 1011). Section 3.3 provides further discussion of appropriate methods for estimating the RME.

3.4.1.2 Estimates of Radiological Dose

Internal exposure to radioactive contaminants is expressed in terms of the 50-yr committed effective dose equivalent (CEDE). To calculate the CEDE, the contaminant concentration is multiplied by dose conversion factors (DCFs) and by the environmental transport factors appropriate for each medium and receptor scenario. Dose conversion factors, which are specific to the radionuclide and exposure pathway, are used to determine the CEDE per unit intake of the radionuclide. Derivation of DCFs incorporates the following consideration for each radionuclide:

- the radiosensitivity of each internal organ,
- the type of radiation emitted by the radionuclide (alpha, beta, and gamma), and
- the solubility class or gastrointestinal absorption fraction of the radionuclide following inhalation or ingestion,

respectively.

The concept of committed dose applies only to internal dose pathways. For the external dose pathway, there is no long-term residence of radionuclides in the body. In this case, the appropriate measure of radiological exposure is the effective dose equivalent (EDE). Dose conversion factors for external exposure from contaminated soil are based on continuous occupancy and a contaminated area of infinite extent and depth. The calculation of EDE at the receptor point must therefore be corrected for appropriate occupancy factors and shielding and geometry for each exposure scenario (Sections 3.4.2-3.4.5).

The sum of the CEDE (internal dose pathways) and the EDE (external dose pathway) is termed the total effective dose equivalent. For purposes of simplification, both CEDE and EDE are referred to as dose (expressed as millirem) in the following sections.

Separate DCFs are available for the various exposure routes (i.e., external gamma, inhalation, ingestion) for the radionuclides potentially present at the Laboratory. When several inhalation or ingestion DCFs are available for each radionuclide, the DCF resulting in the highest dose estimate will be applied. DCFs are available from a DOE (1988, 0266) report for internal dose and from Yu et al. (1993, 1014) for external dose. The dose contributions from short-lived decay products (having half-lives of less than 6 mo) are incorporated in the DCF for the parent radionuclide. For example, the dose contributions from ^{90}Y and ^{137}Ba are included in the DCFs for ^{90}Sr and ^{137}Cs, respectively.

Radiation doses can be calculated with RESRAD, a computer program used to estimate doses to onsite receptors at radioactively contaminated sites (Yu et al. 1993, 1014). The code incorporates the DCFs discussed above and allows the user to enter values for site- and scenario-specific parameters (contaminant concentrations, area and thickness of the contamination at the PRS, exposure pathways, occupancy factors, intake rates, etc.). Use of the RESRAD code to calculate radiological doses is discussed in Attachment II.

3.4.1.3 Estimates of Chemical Intake

Exposure to chemical contaminants is expressed in terms of intake, which is the amount of contaminant taken into the body per unit body weight per unit time (generally expressed as milligrams per kilogram per day).

3.4.2 Scenario-Specific Assumptions and Exposure Parameters

The assumptions used to estimate radiological doses and chemical intakes for the receptors described in Section 3.2.3 are discussed in Sections 3.4.2.1 through 3.4.2.8 and are summarized in Attachment I.

Some exposure parameters depend on the age of the receptor (e.g., ingestion and inhalation rates, body weight, skin surface areas). For long-term receptors for whom some portion of exposure occurs during childhood (e.g., residents and future campers), variable exposure parameters may be appropriate for the portions of exposure that occur during childhood and those portions that occur during adulthood. However, use of variable exposure parameters generally does not alter calculated intake significantly because, for example, decreased body weight corresponds with decreased ingestion and inhalation rates. The only

pathway for which incorporation of both child and adult intake parameters alters calculated intake substantially is the soil ingestion pathway because children have higher ingestion rates than adults (and lower body weights). Under EPA guidance, variable exposure parameters should be assumed for the soil ingestion pathway but are not necessary for inhalation and dermal contact pathways (EPA 1991, 0746; EPA 1992, 1012).

3.4.2.1 Exposure Time, Frequency, and Duration

Exposure time, frequency, and duration together define the total amount of time a receptor may be in contact with a given distribution of contaminants in the exposure unit. The exposure time is the number of hours per day that a receptor is present at a specific exposure point, the exposure frequency is the number of days per year that exposure occurs, and the exposure duration is the total number of years over which exposure occurs. For each scenario, the exposure time is further divided into time spent indoors and time spent outdoors.

Long-term workers at the Laboratory are assumed to be onsite 8 h/day for 250 days/yr over 25 yr. EPA risk assessment guidance recommends these exposure time, frequency, and duration values for workers (EPA 1991, 0746). In general for ER Program risk assessments, assumptions of increased worker time spent outdoors result in higher intake and dose estimates than assumptions of no or little time spent outdoors. These results occur because intake from particulate inhalation and dose from external gamma irradiation are greater for outdoor exposures, and the assumed soil ingestion rate is also higher for outdoor workers. Therefore, an assumption of a substantial fraction of the day spent outdoors (e.g., 4 h) generally represents the RME for a long-term worker. An exception to this assumption occurs when radon is the contaminant of concern; radon build-up inside buildings might lead to a higher calculated risk for workers who spend most of their time indoors. Specific exposure assumptions for long-term workers in ER Program risk assessments should be made on a PRS-specific basis, depending on the contaminants of concern and the likelihood of outdoor work.

Youth recreational users of canyon sides and bottoms (e.g., hikers and bikers) are assumed to spend 2 h/day outdoors in canyon areas for 170 days/yr (e.g., about 5 days/wk for 8 mo/yr, which are reasonable estimates, considering the climate in the Los Alamos area). The exposure duration is estimated as 9 yr, which is the median time spent at one residence (EPA 1989, 0088) and is also consistent with the age range assumed for this receptor (10-18 years). Although the youth recreational user is more likely to be a concern in the future, the current use scenario is included to address potential trespassers.

The current and future residents of PRS areas in the townsite are assumed to have an exposure frequency of 350 days/yr and an exposure duration of 30 yr, as recommended in EPA guidance (EPA 1991, 0746). The exposure time assumes 20 h/day, allowing for 4 h/day spent away from the property. Two hours per day are assumed to be spent outdoors on the residential property.

Construction workers are assumed to be onsite 8 h/day, 90 days/yr. An exposure duration of 1 yr is used, assuming the time spent at any single PRS is limited. ~~Cumulative exposures from several PRSs for construction workers are discussed in Section 5. These~~ **Construction** workers' activities are assumed to occur outdoors only. In general, construction is currently not allowed in PRS areas.

However, because schedules for required construction work cannot be predicted, the construction worker scenario is considered possible for both the present and the future. **Cumulative exposures from several PRSs for construction workers will be evaluated when this scenario is possible.**

For PRS areas where future recreational use is assumed (i.e., canyon bottoms), camper receptors are assumed to be onsite 24 h/day, 28 days/yr over 20 yr. The future camper is assumed to spend the entire 24 h outdoors.

For the dermal-absorption-from-soil-pathway, all the scenarios except the residential scenario assume the number of soil contact events is equal to the exposure frequency (days per year). For residents, it is assumed that dermal contact occurs seasonally (e.g., during gardening), approximately 5 days/wk for 8 mo/yr over 30 yr. For the youth recreational user, future camper, and future resident scenarios, dermal exposure to compounds in surface water (e.g., during wading) is assumed to occur during daily 1-h exposure events. The future camper is assumed to wade in surface water daily while present at the PRS (i.e., 28 days/yr), whereas the youth recreational user and the future resident are assumed to wade 40 days/yr (it is assumed that these receptors are unlikely to wade daily). For the future resident in canyon bottoms, dermal exposure to compounds in groundwater (e.g., during bathing) is assumed to occur for 15 min/day, 350 days/yr over 30 yr.

3.4.2.2 Body Weight

The standard assumption for adult body weight is 70 kg (EPA 1989, 0305). Therefore, a body weight of 70 kg is used for long-term worker, construction worker, resident, and future camper scenarios and for that portion of the current or future resident and future camper scenarios for which an adult is assessed (applicable to soil ingestion pathway only). A body weight of 50 kg for youths of ages 10 to 18 (EPA 1989, 0305) is considered representative for the youth recreational user. A body weight of 15 kg for children of ages 1 to 6 (EPA 1991, 0746) is assumed for both the child portion of the future camper scenario and for the child portion of the current and future resident scenarios for evaluation of the soil ingestion pathway.

3.4.2.3 Inhalation Rates

EPA recommends the use of an inhalation rate of 0.83 m^3/h (20 m^3/day) for the assessment of resident adult scenarios (EPA 1989, 1011). This inhalation rate (which is based on the average inhalation rate over an entire day, including periods of rest, and light, moderate, and heavy activity) should be used for indoor exposures for the current and future resident scenarios. It is also appropriate for the long-term worker scenario for indoor activities, which are assumed to involve light office, laboratory, or maintenance work. [EPA (1989, 0304) defines activity levels as follows: resting includes watching television, reading, and sleeping; light activity includes most domestic work, hobbies, and conducting minor home repairs; moderate activity includes heavy indoor cleanup, conducting major home repairs, and climbing stairs; and heavy activity includes vigorous physical exercise and climbing stairs carrying a load.]

The inhalation rate for other scenarios is adjusted to account for greater exertion while working outdoors or engaging in recreational activities. The outdoor

inhalation rate assumed for worker and resident scenarios is 1.7 m^3/h, assuming that half of the exposure time is spent at moderate activity and half at light activity. For the youth recreational user, an inhalation rate of 3.2 m^3/h is used; this rate is based on a moderate activity level for a 10-yr-old child (EPA 1989, 0304). The inhalation rate of 1.3 m^3/h for a future camper is based on adult exposures of 8 h/day resting, 12 h/day light activity, 2 h/day moderate activity, and 2 h/day heavy activity.

3.4.2.4 Ingestion Rates for Soil and Dust

Soil ingestion rates are based on EPA guidance (EPA 1991, 0746) to account for incidental ingestion of soil and dust. For most scenarios, the recommended soil ingestion rate of 100 mg/day for adults and youths more than 6 yr of age is appropriate. The assumed soil and dust ingestion rate for youth recreational users is 100 mg /day (EPA 1991, 0746).

The soil ingestion rate recommended for workers spending most of their work day indoors is 50 mg/day (EPA 1991, 0746); this rate should be used for long-term workers who are assumed to perform little or none of their work outdoors. For workers assumed to spend their entire exposure duration (i.e., 8 h/day) indoors, a factor of 0.4 (Alzona et al. 1979, 0994) is applied to the ingestion rate to account for the fact that not all indoor dust is composed of contaminated soil (i.e., smoking and other indoor activities). Long-term workers who spend some portion of their workday outdoors (e.g., 1 h or more) are assumed to have a higher ingestion rate of 100 mg /day [which is also consistent with EPA guidance to assume 100 mg/day soil ingestion for adults; (EPA 1991, 0746)].

Based on a study by Hawley (1985, 0999), EPA recommends 480 mg/day as a default soil ingestion rate for short-term, outdoor activities in a commercial/industrial setting. Therefore, for the construction worker scenario, an ingestion rate of 480 mg/day is assumed to account for extended daily exposure to contaminated material and ingestion of inhaled material that is not retained in the lungs.

EPA guidance recommends considering both early childhood (i.e., ages 1 to 6, when intake is greater) and adult exposures when evaluating soil ingestion for a residential scenario. For ER Program risk assessments, this approach is used for evaluating both current and future resident and future camper scenarios. Therefore, for the soil ingestion pathway, exposure for these receptors is evaluated assuming 6 yr of exposure as a young child, with an ingestion rate of 200 mg/day, and either 24 yr of exposure (resident scenario) or 14 yr exposure (future camper scenario) at the lower ingestion rate of 100 mg/day.

3.4.2.5 Ingestion Rate for Water

Future residents in canyon bottoms and future campers are the only receptors for which the pathway of groundwater ingestion is evaluated. For this scenario, the standard default value of 2 L/d, recommended by EPA (1991, 0746) for adult residents, is assumed to be the water ingestion rate. It is also assumed that incidental ingestion of surface water might occur during wading for the youth recreational user, future camper, and future resident scenarios. The surface water ingestion rate of 0.05 L/d is based on EPA's (1989, 0088) recommended incidental ingestion level of 50 mL/h for a 1-h exposure time.

3.4.2.6 Produce Ingestion Rate and Fraction of Ingested Produce Grown on Contaminated Area

Current and future residents and future campers are the receptors for which the ingestion of contaminated produce and wild plants is evaluated. EPA (1989, 0304; 1991, 0746) presents figures for "typical" consumption of fruit and vegetables as 140 g/d and 200 g/d, respectively. The reasonable worst-case proportion of ingested produce that is considered to be homegrown (i.e., grown on the contaminated area) ranges from 30% to 40% for fruits and vegetables. Therefore, for the resident scenario, the produce ingestion rate is assumed to be 340 g/d, of which the fraction of ingested produce grown on the contaminated area is estimated to be 40%. The future camper is assumed to ingest wild plants at a rate of 140 g/d, of which 100% is assumed to grow on the contaminated site. Data specific to the Los Alamos area should be used to determine the fractions of produce assumed to be leafy and nonleafy (Section 3.3.2.1.4).

3.4.2.7 Skin Surface Area (Soil and Water Pathways)

Assumptions for total body surface area and the surface area of component body parts are both scenario- and age-dependent (EPA 1992, 1012). For ER Program risk assessments, dermal exposure is evaluated only for adult receptors. This simplification is warranted because dermal exposure is proportional to surface area; thus, intakes calculated for adult and child receptors are similar.

For the showering/bathing scenarios for future residents using canyon bottom groundwater, the mean total adult body surface area is approximately 20,000 cm^2, which is in EPA's recommended default range. For soil contact and wading in surface water scenarios, dermal exposure is expected to occur on roughly 25% of the total surface area (i.e., hands, lower legs, forearms, neck, and head). Twenty-five percent of the mean total body surface area is 5,000 cm^2 for adults. The soil contact surface area for the youth recreational user is increased to 5,000 cm^2 (i.e., adult value) based on the potential dust-generating nature of the recreational activity. For long-term workers who perform some of their work outdoors and future construction workers, protective clothing probably limits the exposure to hands, forearms, neck and head; therefore, the mean surface area for these upper extremities of adults is 3,200 cm^2.

3.4.2.8 Fraction from Contaminated Source

Sampling plans are generally designed so that samples collected for use in a risk assessment represent an exposure unit area. The statistic used to describe the contaminant distribution, the RME, within an exposure unit area is used to calculate receptor dose. An adjustment is recommended in cases for which the area of the PRS is smaller than the appropriate exposure unit area and for which data have not been obtained outside the PRS. This adjustment can be made by prorating the values obtained in the PRS according to the fraction of the exposure unit area corresponding to the PRS. This fraction is termed the "fraction from contaminated source" (FI) and is set to 1 if the PRS is not smaller than the exposure unit.

In effect, the FI parameter adjusts the concentration parameters used in the equations presented in the following sections. This adjustment is equivalent to an assumption that the area outside the PRS but inside the exposure unit is free of contamination. However, considering that the potential impact of nonzero

contamination in this area is likely to be small, averaging with an assumption that this area is free of contamination is unlikely to cause gross decision-making errors. Other assumptions can be made about the area outside the PRS; however, if other assumptions are made, the adjustment becomes more complex.

3.4.3 Equations for Exposure to Soil

This section describes the methodology for estimating chemical intakes and radiological doses for exposures based on concentrations in soil. Equations for estimating human intakes are presented for the following exposure routes: incidental soil ingestion, dermal contact with soil, inhalation of particulates, ingestion of contaminated produce, external gamma exposure, and inhalation of radon. With the exception of the external gamma exposure and radon inhalation pathways, the equations presented are based on equations given in EPA's *Human Health Evaluation Manual* (EPA 1989, 0305); analogous equations are used for the external gamma exposure and radon inhalation pathways.

3.4.3.1 Incidental Soil Ingestion

For both radiological dose and chemical intake estimates, scenario-specific assumptions on exposure time, frequency, and duration, and the assumed ingestion rates for the various receptors are given in Attachment I. The intake for a long-term worker whose work is exclusively indoors is multiplied by a factor of 0.4 to account for the fact that all exposure occurs indoors, where not all ingested dust originates from contaminated soil (Section 3.4.2.4).

3.4.3.1.1 Radiological Dose

The basic equation used to calculate radiation doses from ingesting contaminated soil is

$$D_{is} = R_{is} \times IR_s \times CF \times EF \times FI \times DCF_i(Ing) \times ED \ , \qquad (6)$$

where

D_{is} = dose from ingestion of radionuclide, i, in soil (millirem),

R_{is} = soil exposure concentration of radionuclide, i, in soil (picocuries per gram) (Section 3.3.2.1.1),

IR_s = soil ingestion rate (milligrams per day),

CF = conversion factor (10^{-3} g/mg),

EF = exposure frequency (days per year),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

$DCF_i(Ing)$ = dose conversion factor for ingestion of radionuclide, i (millirems per picocurie), and

ED = exposure duration (years).

To account for increased soil intake rates during childhood years, the ingestion rate can be calculated as the time-weighted average over the exposure duration:

$$IR_s = \frac{(IR_{sc} \times ED_c) + (IR_{sa} \times ED_a)}{ED_c + ED_a}, \qquad (7)$$

where

IR_{sc} = child soil ingestion rate (milligrams per day),

IR_{sa} = adult soil ingestion rate (milligrams per day),

ED_c = child exposure duration (6 yr), and

ED_a = adult exposure duration [24 yr or 14 yr (scenario-specific)].

The soil ingestion dose may be calculated with the RESRAD code using the methodology in Attachment II.

3.4.3.1.2 Chemical Intake

The basic equation used to calculate chemical intake via ingestion is

$$\text{Intake (mg/kg-d)} = \frac{C_i \times IR_s \times CF \times FI \times EF \times ED}{BW \times AF \times AD}, \qquad (8)$$

where

C_i = soil exposure concentration of chemical, i (milligrams per kilogram) (Section 3.3.2.1.1),

IR_s = soil ingestion rate (milligrams per day),

CF = conversion factor (10^{-6} kg/mg),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

EF = exposure frequency (days per year),

E = exposure duration (years),

BW = body weight (kilograms),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for carcinogens and 70 yr for carcinogens)].

A modified equation is used to model 6 yr of childhood exposure and 24 yr of adult exposure for the resident scenario and 6 yr of childhood exposure and 14 yr

of adult exposure for the future camper scenario. The modified equation is

$$\text{Intake (mg/kg-d)} = \frac{C_i \times IR_{sc} \times CF \times FI \times EF \times ED_c}{BW_c \times AF \times AD} + \frac{Ci \times IR_{sa} \times CF \times FI \times EF \times ED_a}{BW_a \times AF \times AD} \quad (9)$$

where

C_i = soil exposure concentration of chemical, i (milligram per kilogram) (Section 3.3.2.1.1),

IR_{sc} = child soil ingestion rate (milligrams per day),

IR_{sa} = adult soil ingestion rate (milligrams per day),

CF = conversion factor (10^{-6} kg/mg),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

EF = exposure frequency (days per year),

ED_c = child exposure duration (6 yr),

ED_a = adult exposure duration [24 yr or 14 yr (scenario-specific)],

BW_c = child body weight (15 kg),

BW_a = adult body weight (70 kg),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

Because chemical intakes for carcinogenic risk calculations are averaged over a lifetime of 70 yr, the intakes calculated for use in carcinogenic risk estimates differ somewhat from those calculated for noncarcinogenic endpoint estimations.

3.4.3.2 Dermal Contact with Soil

Dose estimates and chemical intakes for the dermal exposure pathway are calculated using the exposure concentrations in soil (Section 3.3.2.1.1). Scenario-specific assumptions on the dermal exposure factors (exposure time, frequency, duration, skin area, and soil adherence) are given in Attachment I.

3.4.3.2.1 Radiological Dose

The dermal absorption of radionuclides depends on the chemical compound in which the radioactive element is incorporated. Most radionuclides found in the environment have relatively low dermal absorption fractions, and the contribution

to dose from dermal absorption is typically much smaller than the dose from soil ingestion. In addition, dose conversion factors have not been developed for the dermal absorption pathway for most radionuclides (tritium is an exception). If the dermal absorption pathway results in a significant radiation dose because of the high dermal absorption potential of the chemical compound, the dose must be calculated on a case-by-case basis and requires the derivation of dose conversion factors for dermal absorption.

3.4.3.2.2 Chemical Intake

The following equation is used to estimate the dermal absorption of chemicals from soil:

Intake (mg/kg-day)

$$= \frac{C_i \times CF \times FI \times ADF \times ABS \times EF_d \times ED \times SA_s}{BW \times AF \times AD}, \quad (10)$$

where

C_i = soil exposure concentration of chemical, i (milligrams per kilogram) (Section 3.3.2.1.1),

CF = conversion factor (10^{-6} kg/mg),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

ADF = adherence factor of soil to skin (milligrams per square centimeter per event),

ABS = absorption fraction (unitless, chemical-specific),

EF_d = exposure frequency for soil dermal contact (events per year),

ED = exposure duration (years),

SA_s = skin surface area available for soil contact (square centimeter),

BW = body weight (kilograms),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

Estimation of the dermal absorbed dose for compounds in soil requires identification of appropriate chemical-specific absorption fractions. Although the use of soil permeability coefficients may be preferable, EPA currently recommends the use of absorption fractions until this issue is investigated

further. In estimating absorption fractions, preference is given to experimentally derived values; otherwise, predictive models must be used. Some theoretical approaches to determining an appropriate absorption fraction are discussed in EPA guidance for assessing dermal exposure (EPA 1992, 1012); that document also lists absorption fractions for several compounds.

An EPA review (1992, 1012) of experimental data on soil adherence indicates a conservative range of values from 0.2 mg/cm^2 to 1.5 mg/cm^2 per event. Although the uncertainties in the studies make it difficult to recommend a default value, EPA suggests that 1 mg/cm^2 is a reasonable upper value. Therefore, 1 mg/cm^2 should be used as the soil-to-skin adherence factor for all applicable scenarios (Attachment I).

3.4.3.3 Inhalation of Particulates

Scenario-specific assumptions on exposure time, frequency, and duration and assumed inhalation rates for use in calculating dose and intake from the inhalation pathway are given in Table 3.2.

3.4.3.3.1 Radiological Dose

The following equation is used to calculate radiation doses from the inhalation of contaminated dust when all exposure is outdoors:

$$D_{ia} = R_{ia} \times IR_a \times ET \times EF \times FI \times DCF_i(Inh) \times ED \ , \qquad (11)$$

where

D_{ia} = dose from inhalation of radionuclide, i, in air (millirem),

R_{ia} = concentration of radionuclide, i, in air (picocuries per cubic meter), based on soil exposure concentration (Section 3.3.2.1.3),

IR_a = inhalation rate (cubic meters per hour),

ET = exposure time (hours per day),

EF = exposure frequency (days per year),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

$DCF_i(Inh)$ = dose conversion factor for inhalation of radionuclide, i (millirems per picocurie), and

ED = exposure duration (years).

For receptors assumed to spend some portion (or all) of the exposure time indoors (i.e., long-term workers and current and future residents), the concentration of particulates indoors is assumed to be reduced by 60% (Alzona et al. 1979, 0994). To account for this assumption, the product of inhalation rate

and exposure time in the above equation may be substituted by the following expression:

$$IR_a \times ET = (IR_o \times ET_o) + (0.4 \times IR_i \times ET_i) \ , \qquad (12)$$

where

IR_o = outdoor inhalation rate (cubic meters per hour),

IR_i = indoor inhalation rate (cubic meters per hour),

ET_o = exposure time outdoors (hours per day), and

ET_i = exposure time indoors (hours per day).

The dust inhalation dose may be calculated with the RESRAD code using the methodology in Attachment II.

3.4.3.3.2 Chemical Intake

The following equation is used for calculating chemical intake via inhalation when all exposure is outdoors:

$$\text{Intake (mg/kg-day)} = \frac{C_i \times IR_a \times FI \times ET \times EF \times ED}{BW \times AF \times AD} \ , \qquad (13)$$

where

C_i = concentration of chemical, i, in air (milligrams per cubic meter), based on soil exposure concentration (Section 3.3.2.1.3),

IR_a = inhalation rate (cubic meters per hour),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

ET = exposure time (hours per day),

EF = exposure frequency (days per year),

ED = exposure duration (years),

BW = body weight (kilograms),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

For receptors assumed to spend some portion (or all) of the exposure time

indoors (i.e., long-term workers and current and future residents), the concentration of particulates indoors is assumed to be reduced by 60%. The above equation is modified to account for this assumption, as follows:

$$\text{Intake (mg/kg-day)} = \frac{C_i \times IR_o \times FI \times ET_o \times EF \times ED}{BW \times AF \times AD} + \frac{C_i \times IR_i \times FI \times ET_i \times 0.4 \times EF \times ED}{BW \times AF \times AD}, \quad (14)$$

where

C_i = concentration of chemical, i, in air (milligrams per cubic meter), based on soil exposure concentration (Section 3.3.2.1.3),

IR_o = outdoor inhalation rate (cubic meters per hour),

IR_i = indoor inhalation rate (cubic meters per hour),

FI = fraction from contaminated source, calculated using scenario-specific exposure units (unitless) (Section 3.4.2.8),

ET_o = exposure time outdoors (hours per day),

ET_i = exposure time indoors (hours per day),

EF = exposure frequency (days/year),

ED = exposure duration (years),

BW = body weight (kilograms),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

3.4.3.4 Ingestion of Contaminated Produce

3.4.3.4.1 Radiological Dose

The following equation is used to calculate the radiation doses from the ingestion of contaminated produce:

$$D_{ip} = R_{ip} \times IR_p \times F_{HG} \times EF \times DCF_i(Ing) \times ED, \quad (15)$$

where

D_{ip} = dose from ingestion of radionuclide, i, in produce (millirems),

R_{ip} = concentration of radionuclide, i, in produce (picocuries per kilogram), based on soil exposure concentration (Section 3.3.2.1.4),

IR_p = produce ingestion rate (kilograms per day),

F_{HG} = fraction of ingested produce that is grown on or collected from the contaminated area (unitless) (Section 3.4.2.6),

EF = exposure frequency (days per year),

$DCF_i(Ing)$ = dose conversion factor for ingestion of radionuclide, i (millirems per picocurie), and

ED = exposure duration (years).

The produce ingestion dose may be calculated with the RESRAD code using the methodology in Attachment II.

3.4.3.4.2 Chemical Intake

The following equation is used for calculating chemical intake via ingestion of contaminated produce:

$$\text{Intake (mg/kg-day)} = \frac{C_i \times IR_p \times F_{HG} \times EF \times ED}{BW \times AF \times AD}, \qquad (16)$$

where

C_i = concentration of chemical, i, in produce (milligrams per kilogram), based soil exposure concentration (Section 3.3.2.1.4),

IR_p = produce ingestion rate (kilograms per day),

FI_{HG} = fraction ingested produce that is grown on or collected from the contaminated area (unitless)(Section 3.4.2.6),

EF = exposure frequency (days per year),

ED = exposure duration (years),

BW = body weight (kilograms),

AF = averaging frequency (365 days/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

This equation is used for the current and future resident and future camper scenarios.

3.4.3.5 External Gamma Exposure

If direct measurements of the external radiation exposure levels from contaminated soil are available, the dose to current receptors may be estimated as follows:

$$D_x = EL \times CF \times [(ET_i \times SF) + ET_o] \times EF \times ED \ , \qquad (17)$$

where

D_x = dose from external gamma radiation (millirems),

EL = outdoor exposure level at 1 m above ground averaged over the exposure unit (milliroentgens per hour),

CF = conversion factor (a function of the average gamma-ray energy) (millirems per hour per milliroentgens per hour),

ET_i = exposure time indoors (hours per day),

ET_o = exposure time outdoors (hours per day),

SF = shielding factor for indoor occupancy (0.7),

EF = exposure frequency (days/year), and

ED = exposure duration (years).

The RESRAD computer code (Attachment II) may be used to calculate the external gamma dose if direct measurements are not available.

3.4.3.6 Inhalation of Radon

The working-level month (WLM) is the unit of exposure used to calculate doses from direct measurement of radon (Section 3.3.2.1.6). It is defined as the exposure to 1 WL for 170 h (2,000 working hours per year). The WLM was used historically to report exposures to uranium miners. An exposure to 1 WLM of ^{222}Rn and ^{220}Rn decay products is equal to a committed effective dose equivalent of 1,000 mrem and 350 mrem, respectively (ICRP 1981, 1000). The ratio of inhalation rates for various scenarios (e.g., long-term worker or resident vs uranium miner) must be applied to correct for the implicit assumptions upon which the WLM is based.

The equation used to calculate dose from directly measured radon is

$$WLM = \frac{C_{Rn} \times IR \times ET \times EF \times ED}{CF} \ , \qquad (18)$$

where

WLM = working-level month,

C_{Rn} = indoor or outdoor air concentration of ^{222}Rn and ^{220}Rn decay products (WL),

IR = inhalation rate (cubic meters per hour),

ET = exposure time (hours per day),

EF = exposure frequency (days per year),

ED = exposure duration (years), and

CF = conversion factor, 204 m^3/mo [CF is the product of the inhalation rate assumed for uranium miners (1.2 m^3/h) and the number of working hours in 1 mo (170 h/mo)]

To estimate the dose from radon and radon decay products based on soil contaminant levels, the RESRAD code can be used (Attachment 1).

3.4.4 Equations for Exposure to Groundwater and Surface Water

This section describes the methodology for estimating chemical intakes and radiological doses for exposures based on concentrations in water. The following exposure routes are addressed: ingestion of groundwater or surface water, dermal absorption from groundwater or surface water, and inhalation from household use of groundwater. The equations presented are based on equations given in EPA's *Human Health Evaluation Manual* (EPA 1989, 0305).

3.4.4.1 Groundwater and Surface Water Ingestion

Radionuclide doses and chemical intakes from ingestion of alluvial groundwater and surface water are calculated based on exposure concentrations in groundwater and surface water (Section 3.3.2.2.1). The youth recreational user, future camper, and the future resident in a canyon bottom are considered the relevant scenarios for this exposure pathway (however, the youth recreational user would not ingest groundwater). Scenario-specific assumptions on exposure time, frequency, and duration, and the assumed ingestion rate are given in Attachment I.

3.4.4.1.1 Radiological Dose

The dose associated with intake of radioactive contaminants resulting from ingestion of groundwater or surface water can be calculated as follows:

$$D_{iw} = R_{iw} \times (IR_{gw} \text{ or } IR_{sw}) \times (EF \text{ or } EF_{sw}) \times ED \times DCF_i(Ing) \quad , \qquad (19)$$

where

D_{iw} = dose from ingestion of radionuclide, i, in water (millirem),

R_{iw} = exposure concentration of radionuclides in groundwater or surface water (picocuries per liter) (Section 3.3.2.2.1),

IR_{gw} = groundwater ingestion rate (liters per day),

IR_{sw} = surface water ingestion rate (liters per day),

EF = exposure frequency (for groundwater ingestion) (days per year),

EF_{sw} = exposure frequency (for surface water ingestion) (days per year),

ED = exposure duration (years), and

$DCF_i(Ing)$ = dose conversion factor for ingestion of radionuclide, i (millirems per picocurie).

3.4.4.1.2 Chemical Intake

The following equation is used for calculating chemical intake via ingestion of groundwater or surface water:

$$\text{Intake (mg/kg-day)} = \frac{C_i \times (IR_{gw} \text{ or } IR_{sw}) \times (EF \text{ or } EF_{sw}) \times ED}{BW \times AF \times AD}, \quad (20)$$

where

C_i = exposure concentration of chemical, i, in groundwater or surface water (milligram per liter) (Section 3.3.2.2.1),

IR_{gw} = groundwater ingestion rate (liters per day),

IR_{sw} = surface water ingestion rate (liters per day),

EF = exposure frequency (for groundwater ingestion) (days per year),

EF_{sw} = exposure frequency (for surface water ingestion) (days per year),

ED = exposure duration (years),

BW = body weight (kilograms),

AF = averaging frequency (365 d/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

Because chemical intakes for carcinogenic risk calculations are averaged over a lifetime of 70 years, the intakes calculated for use in carcinogenic risk estimates differ somewhat from those calculated for noncarcinogenic endpoint estimations.

3.4.4.2 Dermal Absorption from Groundwater and Surface Water

Dose estimates and chemical intakes for the dermal exposure pathway are calculated using the exposure concentrations in groundwater or surface water (Section 3.3.2.1.2). Scenario-specific assumptions on the dermal exposure factors (exposure time, frequency, duration, and skin surface area) are given in Attachment II.

3.4.4.2.1 Radiological Dose

Only dermal absorption of tritium is evaluated (Section 3.3.2.1.2). Dermal absorption of tritium has been investigated for both air and water immersion. According to Pinson and Langham (957, 1008), the absorption rate of water vapor by skin is approximately 0.014 L/h (24°C, 100% humidity). Dermal absorption when skin is immersed in water that is the same temperature as the skin is estimated to be 2.5 times faster (0.035 L/h). These estimates are based on a total skin surface area of 1.9 m^2 and should be reduced in cases where only partial contact with the skin occurs (e.g., wading). The tritium uptake rate is simply the concentration of tritium per liter of water times the water absorption rate, as shown in the following equation:

$$D_{H\text{-}3} = C_{H\text{-}3} \times U \times (SA_{gw} \text{ or } SA_{sw}) \times (ET \text{ or } ET_{sw}) \times (EF \text{ or } EF_{sw}) \times ED \times DCF_{H\text{-}3}\ , \qquad (21)$$

where

$D_{H\text{-}3}$ = dose from dermal absorption of tritium (millirems),

$C_{H\text{-}3}$ = exposure concentration of tritium in water (picocuries per kilogram),

U = dermal absorption rate from submersion in water (1.8×10^{-6} L/h-cm^2),

SA_{gw} = skin surface area available for groundwater contact (square centimeters),

SA_{sw} = skin surface area available for surface water contact (square centimeters),

ET = exposure time (for groundwater contact) (hours per day),

ET_{sw} = exposure time (for surface water contact) (hours per day),

EF = exposure frequency (for groundwater contact) (days per year),

EF_{sw} = exposure frequency (for surface water contact) (days per year),

ED = exposure duration (years), and

DCF_{H-3} = dose conversion factor for tritium uptake (6.3×10^{-8} mrem/pCi).

3.4.4.2.2 Chemical Intake

The following equation is used to estimate the dermal absorption of inorganic chemicals from groundwater or surface water:

$$\text{Intake (mg/kg-day)} = \frac{C_i \times CF \times K_p \times (ET_{gw} \text{ or } ET_{sw}) \times EV \times (EF \times ET_{sw})\ ED \times (SA_{gw} \text{ or } SA_{sw})}{BW \times AF \times AD}, \quad (22)$$

where

C_i = exposure concentration of chemical, i, in water (milligrams per liter),

CF = conversion factor (10^{-3} L/cm^3),

K_p = permeability coefficient, chemical-specific (centimeters per hour),

ET_{gw} = dermal contact time for exposure to groundwater (hours per event),

ET_{sw} = dermal contact time for exposure to surface water (hours per event),

EV = event frequency [events per day (equal to 1 event/day for youth recreational user, future resident, and future camper scenarios)],

EF = exposure frequency (days per year),

EF_{sw} = exposure frequency (for surface water contact) (days per year),

ED = exposure duration (years),

SA_{gw} = skin surface area available for contact with groundwater (square centimeters),

SA_{sw} = skin surface area available for contact with surface water, (square centimeters),

BW = body weight (kilograms),

AF = averaging frequency (365 d/yr), and

AD = averaging duration [years (equal to ED for noncarcinogens and 70 yr for carcinogens)].

The above equation applies to the youth recreational user, future camper, and future canyon bottom resident scenarios and assumes an absorbed dose that occurs during water contact events (i.e., wading and bathing). This equation is recommended for evaluating inorganic contaminants using the traditional steady-state approach for estimating the dermally absorbed dose from water. Chemical-specific permeability coefficients for selected inorganics are summarized by EPA (1992, 1012); for other inorganics, the default assumption of 10^{-3} cm/h is used.

For organics, EPA is cautiously recommending a newer, non-steady-state approach EPA (1992, 1012). The agency feels that the model more accurately reflects normal human exposure conditions and that it accounts for the dose that may occur after the actual exposure event resulting from absorption of contaminants in skin lipids. However, it appears that the model may be overly conservative, and it is difficult to validate because data are lacking. Alternatively, the steady-state equation 23 may be used for organics by using permeability coefficients obtained from EPA (1992, 1012) in which experimentally measured or calculated values (i.e., based on octanol/water partition coefficients) are summarized for over 150 common organic compounds.

3.4.4.3 Inhalation from Household Use of Groundwater

Section 3.3.2.2 provides a discussion of this pathway.

3.4.5 Assessment of Exposure to Contaminated Building Surfaces

Radiation doses to current and future long-term workers resulting from contaminated building surfaces may be estimated using National Research Council methodology for dose assessments from residual radioactive contamination from decommissioning (Kennedy and Strenge 1993, 1002). That document provides detailed equations for estimating the dose to workers from external gamma radiation from contaminated surfaces, from inhalation of suspended surface contamination, and from inadvertent ingestion of surface contamination.

Section 3.3.2.3 provides a discussion of the evaluation of building surfaces contaminated with chemical substances.

4.0 TOXICITY ASSESSMENT

The toxicity assessment involves gathering information on potential health effects and/or toxicological properties of the chemicals of concern for subsequent comparison with estimated intake levels. Both noncarcinogenic and carcinogenic effects are considered in the toxicity assessment.

4.1 Types of Toxicological Information Considered in the Toxicity Assessment

In collecting toxicity information (much of this information will have been collected in the screening assessment phase but may need to be updated), the hierarchy established by Superfund guidance for the appropriate sources of toxicity values will be followed. The first choice for information is the Integrated Risk Information System (IRIS) (EPA 1993, 1062), which is a data base maintained by the EPA

that contains the most up-to-date toxicity information and regulatory values for a number of chemicals. If the chemical of potential concern is not in IRIS, the Health Effects Assessment Summary Tables (HEAST) (EPA 1992, 0833) will be consulted. These tables are not as current as IRIS; however, unlike IRIS, they contain interim as well as verified health-based values. If values are not available in IRIS or HEAST, toxicity values will be derived from alternate sources such as regulatory levels (e.g., maximum concentration levels in drinking water or comparative potency factors for PAHs). The basis and confidence for all toxicity values will be provided.

The outcome of this analysis will be a table of toxicity values for the chemicals of potential concern. Available toxicity information will be reviewed and summarized for each chemical of potential concern. This information will focus on health effects (both noncarcinogenic and carcinogenic) from chronic and/or subchronic exposure because these time spans are typically the exposure durations of concern when estimating human health risk. However, acute toxicity will also be considered. A chronic toxicity table will follow the table of toxicity values in the text to summarize critical effects and target organs associated with each chemical.

4.2 Toxicity Assessment for Noncarcinogenic Effects

The goal of this step is to identify reference doses (RfDs) for the various compounds. The RfDs, established by the EPA, are estimates of the daily exposure to the human population below which exposure is not likely to incur appreciable risk of noncarcinogenic deleterious effects during a lifetime. Reference concentrations are analogous to RfDs and are typically used to evaluate toxicity resulting from exposure via inhalation. These values are converted to a daily inhalation dose, an inhalation RfD. The RfD and/or reference concentration are used to evaluate the potential for noncarcinogenic health effects from a particular chemical.

4.3 Toxicity Assessment for Carcinogenic Effects

4.3.1 Nonradiological Contaminants

The cancer slope factor is an upper-bound estimate of cancer risk associated with exposure to a particular chemical and is used to estimate the lifetime excess cancer risk associated with the estimated chemical intake for that chemical. The EPA's weight-of-evidence classification will be provided to indicate the confidence in the evidence used to classify a chemical as a carcinogen. The EPA classifies chemicals in Groups A, B1, B2, C, D, and E. Group A chemicals are those chemicals considered to be human carcinogens because sufficient evidence has been obtained to demonstrate carcinogenicity in humans. Group B chemicals are probable human carcinogens. A classification of B1 indicates limited evidence of carcinogenicity in humans, and B2 indicates demonstrated evidence of carcinogenicity in animals with inadequate or lack of evidence in humans. Group C suggests only the possibility of human carcinogenicity because evidence of carcinogenicity in animals is limited and data on humans are inadequate or lacking. Group D compounds are not classifiable as to human carcinogenicity because of inadequate or no evidence, and Group E chemicals show evidence of noncarcinogenicity. The cancer slope factors are used to estimate the excess cancer risk associated with exposure to potential carcinogens.

4.3.2 Radiological Contaminants

The toxic effects of radiological contaminants are distinctly different from those of nonradiological contaminants. Therefore, a separate section will be included in the analysis to discuss the chronic toxicity of radionuclides. It is recognized that exposure to radionuclides can have several health effects, such as teratogenesis and mutagenisis; however, carcinogenicity will be the focus of toxicity assessment and risk characterization because it is the principal adverse biological effect following chronic exposure and is well documented.

4.4 Identifying Appropriate Toxicity Values for Risk Assessment

RfDs and cancer slope factors are EPA's preferred toxicity values; therefore, when available, they will be used for the quantitative portion of the toxicity assessment. Alternate sources will be used (also when available) for those compounds that do not have published toxicity values, and toxicity values will be derived from these data, if possible. Additional information may include controlled epidemiologic investigations, clinical studies, and experimental animal studies. A toxicologist will review this information and, if it is determined that the information is adequate to establish a valid toxicity value, the toxicologist will derive a toxicity value.

4.5 Uncertainties Related to Toxicity Information

For those compounds that have established RfDs, the confidence in the RfD and the data base of toxicological information will be provided. The weight of evidence for the carcinogenic potential of compounds will also be discussed. Uncertainty is increased when alternate means are used to derive toxicity values. The uncertainty generated from the various methods used to estimate toxicity values will be examined. When no toxicity information is available, the limitations and impact inherent to excluding contaminants of concern can be great. The cumulative consequences of uncertainty from the different sources of toxicity information (or lack thereof) will be described and quantified to the extent possible.

5.0 RISK CHARACTERIZATION

5.1 Quantifying Risks

Risk characterization involves integrating the exposure and toxicity assessments in quantitative and qualitative expressions of potential health risk. The data from the initial tasks are reviewed, and the carcinogenic and noncarcinogenic risks are quantified for individual and multiple chemicals.

To characterize potential noncarcinogenic effects, chemical intakes and toxicity values are compared. The hazard quotient is the ratio of a chemical exposure level over a specified amount of time to an RfD for that chemical. The hazard quotient is not a statistical probability that a noncarcinogenic effect will occur but is rather a comparison of the exposure level and the appropriate toxicity value. The hazard quotients for several chemicals are summed for a particular pathway to yield a hazard index for that pathway. All hazard indices in an exposure scenario will be totaled. Hazard indices are not mathematical probabilities of the incidence or severity of an adverse health effect but a numerical index of a recommended

safety threshold. A hazard index or hazard quotient greater than 1 indicates a potential health risk. If a hazard index exceeds 1, the hazard index is segregated by target organ on a chemical-by-chemical basis. A hazard index of less than 1 suggests that site-related contaminants are not likely to present a health risk under the given exposure scenario.

Carcinogenic risk is characterized by estimating the probability that an individual will develop cancer over a lifetime as a result of exposure to a potential carcinogen. The probability of developing cancer is estimated by multiplying the cancer slope factor by the calculated intake. The carcinogenic risk estimate is generally an upper-bound estimate of risk, which means that the "true risk" probably does not exceed the risk estimates generated for the assessment and is likely to be less than the predicted risk. Again, risk is summed by chemical and pathway to determine the scenario cancer risk.

Exposure to radiological contaminants of concern are calculated by using the RESRAD computer code, which generates annual doses for potential receptors. The dose estimates are compared with the DOE's dose limit of 25 mrem/yr, which is based on exposure to an individual from a single site. This dose estimate is not combined with nonradiological risk. The contribution to adverse carcinogenic health effects from exposure to radionuclides will be discussed in the risk assessment in conjunction with carcinogenic risk from nonradiological constituents.

5.2 Combining Risk Across Exposure Pathways

The resulting carcinogenic and noncarcinogenic risks are added across exposure pathways in an exposure scenario. Additionally, combined impacts from multiple exposure scenarios are assessed if exposure to more than one scenario is possible (e.g., a child exposed in both recreational and residential scenarios).

5.3 Assessing and Presenting Uncertainty

The risk assessment will also discuss the uncertainties inherently associated with all components of the risk assessment process. The uncertainty analysis will highlight the areas in which assumptions were incorporated because sufficient data were not available to accurately characterize the risks associated with a particular exposure. The implication of these uncertainties on the overall risk results will be discussed fully. Whenever possible, uncertainty analysis will include a quantitative evaluation. The general approach, when it is necessary to make assumptions about the frequency or magnitude of exposures, will be to use conservative assumptions ~~that are likely to overestimate risk.~~ **so that risk is never underestimated.**

5.4 Summarizing and Presenting the Results of Baseline Risk Characterization

Once the carcinogenic and noncarcinogenic risks have been assessed for a site, a determination will be made of which contaminants, media, and pathways present the greatest risks for each scenario. The conclusions will be summarized in the text, and the risk values will be in tabular form.

ATTACHMENT I

RISK ASSESSMENT FOR LOS ALAMOS NATIONAL LABORATORY: SUGGESTED VALUES FOR SCENARIO PARAMETERS

Parameter	Unit	Long-Term Worker[a]	Construction Worker[b]	Youth Recreational User[c]	Resident[d]	Future Camper[e]
General Exposure Parameters						
Total Exposure Time (ET)	h/d	8	8	2	20	24
Exposure Time Outdoors (ET_o)	h/d	0-8	8	2	2	24
Exposure Frequency (EF)	d/yr	250	90	170	350	28
Exposure Duration[f] (ED)	yr	25	1	9	30	20
Area of Exposure Unit[g] (A_{EU})	m^2	500	Area of Contamin-ation	2000	500	2000
Body Weight[h] (BW)	kg	70	70	50	70 Adult 15 Child	70 Adult 15 Child
Indoor Shielding Factor for External Gamma (SF)	%	70	NA[i]	NA	70	NA
Pathway-Specific Exposure Parameters						
Inhalation Pathway						
Inhalation Rate[j] (IR_a)	m^3/h	0.83 Indoors 1.7 Outdoors	1.7	3.2	0.83 Indoors 1.7 Outdoors	1.3
Particulate Concentration in Air[k] (PC)	mg/m^3	0.09	15	PRS-specific	0.09	0.09
Amount of Outdoor Dust Present Indoors[l]	%	40	NA	NA	40	NA
Ingestion Pathways						
Soil Ingestion Rate[m] (IR_s)	mg/d	50 or 100	480	100	100 Adult 200 Child	100 Adult 200 Child
Groundwater Ingestion Rate[n] (IR_{gw})	L/d	1	1	NA	2	2
Surface water Ingestion Rate[n] (IR_{sw})	L/d	1	1	0.05	0.05	0.05
Exposure Frequency for Surface Water Ingestion[o] (EF_{sw})	d/yr	NA	NA	40	40	28
Produce or Berry Ingestion Rate[p] (IR_p)	g/d	NA	NA	NA	340	140
Fraction Ingested Produce Grown on Contaminated Area[p] (F_{HG})	%	NA	NA	NA	30	100
Soil Dermal Contact Pathway						
Exposure Frequency for Soil Dermal Contact[q] (EF_d)	events/yr	NA	90	170	170	28
Skin Surface Area Available for Soil Contact[r] (SA_s)	cm^2/event	NA	3200 (arms & hands)	5000	5000	5000
Soil-to-Skin Adherence Factor[r] (ADF)	mg/cm^2	NA	1	1	1	1
Groundwater Dermal Contact Pathway						
Dermal Contact Exposure Time for Groundwater (ET_{gw})	h/event	NA	NA	NA	0.25	NA
Skin Surface Area Available for Groundwater Contact[r] (SA_{gw})	cm^2/event	NA	NA	NA	20,000	NA

Parameter	Unit	Long-Term Worker[a]	Construction Worker[b]	Youth Recreational User[c]	Resident[d]	Future Camper[e]
Surface Water Dermal Contact Pathway						
Dermal Contact Exposure Time for Surface Water (ET_{sw})	h/ event	NA	NA	1	1	1
Exposure Frequency for Surface Water Dermal Contact[o] (EF_{sw})	d/yr	NA	NA	40	40	28
Skin Surface Area Available for Surface Water Contact[r] (SA_{sw})	cm^2/ event	NA	NA	5000	5000	5000

a. Current and future long-term workers; usually will assume 4 h/d working outdoors (represents reasonable maximum exposure). Primarily for mesa top areas but may be used for canyon bottoms, as appropriate.

b. Construction worker; for evaluation of exposure to areas of surface and subsurface contamination as appropriate. Evaluation limited to contaminants in soils at depths of 12 ft or less.

c. Youth recreational user of canyon sides and bottoms. Receptor is a youth age 10 to 18 using canyon sides and bottoms for hiking or biking. This is the only land use scenario applicable for areas of contamination on canyon sides. Although unlikely as a current use scenario because of institutional controls on Laboratory property, it is possible that limited trespassing occurs.

d. Current or future resident is a resident of extra-Laboratory areas (may include some canyon bottom areas in the future). This scenario does not apply to Laboratory property, for which future land use will be controlled by DOE.

e. Future camper on Laboratory mesa top or canyon bottom areas, assuming the site is released for recreational use (e.g., released to National Park Service).

f. Exposure duration for occupational and residential scenarios (i.e., 25 and 30 yr) recommended by the EPA (1991, 0746). For youth recreational user, 9 yr is the age-range duration and is also the median time at one residence (EPA 1989, 0305). Construction worker exposure duration is chosen based on assumption that projects will be of limited scope. Future recreational users are assumed to use site as long-term vacation area.

g. Area of residential exposure unit based on EPA guidance (EPA 1989, 0305). Other areas based on assumptions on likely scenario-specific activity patterns.

h. Body weights: 70 kg for adult scenarios (EPA 1991, 0746); 50 kg for youth scenarios, 12 to <15 yr old (EPA 1989, 0304); 15 kg for child 1-6 yr old (EPA 1991, 0746).

i. Not applicable.

j. Inhalation Rates: standard default value of 0.83 m^3/h used for indoor residential and indoor long-term worker exposures (EPA 1991, 0746). Inhalation rates accounting for resting, light, moderate, and heavy activity (EPA 1989, 1011) are calculated for other scenarios as follows:

- Long-term or Construction Worker, Resident—outdoor inhalation rate of 1.7 m^3/h equal to 0.5 exposure time at light activity + 0.5 exposure time at moderate activity, adult male inhalation rate.
- Youth Recreational User—inhalation rate of 3.2 m^3/h equal to entire exposure time at moderate activity, inhalation rate for 10-yr olds.
- Future Camper—inhalation rate of 1.3 m^3/h equal to 8 h/d resting, 12 h/d light activity, 2 h/d moderate activity, and 2 h/d heavy activity, adult male inhalation rate.

- Definitions—resting: reading, sleeping, watching television; light activity: domestic work, personal care, minor indoor repairs and home improvements; moderate activity: heavy indoor cleanup, major indoor repairs, climbing stairs; heavy activity: vigorous physical exercise, climbing stairs carrying a load (EPA 1989, 1011).

k. Reference for 0.09 mg/m^3 particulate concentration in air is Environmental Protection Group (1990, 0497). Value of 15 mg/m^3 considered maximum for construction worker scenarios (OSHA 1991, 0610).

l. Based on value given in Alzona et al. (1979, 0994). Only applies for inhalation pathways and for soil ingestion pathway where entire exposure occurs indoors.

m. Standard default soil ingestion rates recommended by EPA (1991, 0746). For the resident and future camper scenarios, the soil ingestion pathway assumes an ingestion rate of 200 mg/d and a body weight of 15 kg for 6 yr exposure as a child and an ingestion rate of 100 mg/d and a body weight of 70 kg for 24 yr and 14 yr, respectively, for exposure as an adult. Long-term worker rates of 50 or 100 mg/d are rates recommended for commercial/industrial scenarios by EPA (1991, 0746). Rate of 480 mg/d for the future construction worker based on EPA guidance (1991, 0746) to account for substantial soil contact and potential ingestion of inhaled material that is not retained in the lungs.

n. For groundwater, standard default water ingestion rate recommended in EPA (1991, 0746). For surface water ingestion, 0.05 L/d based on EPA recommended ingestion level of 50 mL/h for the 1-h exposure time (EPA 1989, 0305).

o. Exposure frequency for surface water ingestion and dermal exposure equal to total exposure frequency (28 d/yr) for the future camper; equal to 40 d/yr for the youth recreational user and future residents (based on best professional judgment of likely wading frequency.

p. Produce ingestion rate and fraction produce grown on or collected from the contaminated area: 340 g/d, 30% for residential scenario includes ingestion of vegetables and fruits; 140 g/d, 100% for future recreational user for fruits only (EPA 1991, 0746).

q. Number of soil dermal contact events equal to exposure frequency, except for residential scenario. For residents, it is assumed that dermal contact occurs seasonally (e.g., during gardening), about 5 d/wk for 8 mo/yr.

r. Skin surface area available for contact and soil-to-skin adherence factor assumptions, as recommended in EPA (1992, 0833).

ATTACHMENT II

METHODS FOR USING RESRAD IN ER PROGRAM RISK ASSESSMENTS

1.0 BACKGROUND

DOE Order 5400.5 (DOE 1990, 0080) requires that the methodology incorporated in the RESRAD computer code (Gilbert 1987, 0312; Yu et al. 1993, 1014) be used to establish soil cleanup guidelines for radionuclide contamination at DOE sites. With some minor modifications of input data and output results, which are outlined in this attachment, the RESRAD code can also be used in implementing the proposed risk assessment methodology for soil contaminated with radionuclides. The use of a computer code, such as RESRAD, carries the additional advantage of minimizing data input and calculational errors.

RESRAD is a pathway analysis code that calculates radiation doses to an onsite individual from chronic exposure to soil contaminated with radionuclides. The RESRAD code allows the user to define up to nine pathways and three exposure routes: external gamma dose from radionuclides in soil; inhalation dose from contaminated dust and radon gas; and ingestion dose from intake of contaminated plants, meat, milk, aquatic foods, water, and soil. A variety of scenarios, including residential, industrial, and recreational, can be modeled by adding or suppressing pathways and entering appropriate values for occupancy and consumption rates.

The following exposure pathways will be considered:

- external exposure from gamma-emitting radionuclides in soil,
- inhalation of airborne particulates originating from contaminated soil,
- inhalation of radon progeny caused by radon emanations from soil contaminated with radon precursors,
- ingestion of contaminated soil, and
- ingestion of produce grown on contaminated soil.

In the discussion that follows, it is assumed that the reader is familiar with the operation of the RESRAD code (i.e., is experienced in entering data, running the code, and obtaining results) or has access to the user's guide.

To provide conservative results that are compatible with the calculations performed for chemicals, leaching of radionuclides from the contaminated zone is neglected for all pathways, which also has the effect of suppressing the water-dependent pathways. To set the leach rate to 0, the distribution coefficients for the radionuclides in the contaminated zone should all be set to a very large number (e.g., 106). This way, only the radioactive decay is taken into consideration. Also, the erosion rate of the contaminated zone should be set to 0.

The RESRAD code calculates doses on an annual basis. The maximum dose from most radionuclides occurs at time 0. However, for some radionuclides, the ingrowth of decay products may result in higher dose rates at a later time. To obtain a conservative total dose estimate, the maximum annual dose calculated for each radionuclide should be multiplied by the exposure duration (in years). However, for radionuclides such as ^{60}Co whose half-lives are short relative to the exposure duration, the user may choose to have the code calculate doses at regular intervals throughout the period of exposure. For example, if the exposure duration is 30 yr, the user could have the code calculate doses at 3-yr intervals (t=0, 3, 6, ..., 27 yr). The resulting annual doses are then summed and multiplied by 3 to obtain the total dose received during the exposure period.

With the exception of the external exposure pathway and the plant ingestion pathway, the concentrations entered in the code should cover the entire exposure unit area. The appropriate depth of the contamination should be entered as input to the code (e.g., 0.15 m for surface soil contamination). When exposure to future receptors from subsurface contamination is evaluated, a zone of surface contamination is created by assuming that the contaminated subsurface soil is excavated and distributed over the exposure unit area. The average depth of the contaminated layer to be entered in the code is calculated by dividing the volume of the excavated soil by the exposure unit area. If this thickness exceeds 0.15 m, then ~~the latter~~ **0.15 m** should be entered.

For external exposure and plant ingestion pathways, the contaminant concentrations entered in the code should be derived from measurements taken over the area of contamination rather than the exposure unit area.

The RESRAD code calculates area factors that are used to take into account uncontaminated areas when the contaminated zone is small relative to the size of the exposure unit. However, RESRAD does not allow the user to change the size of the exposure unit parameter. For all pathways except external exposure, it is recommended that the area of the contaminated zone be set to the RESRAD default of 10,000 m^2, which results in an area factor of 1 calculated by the code. Because the concentrations entered in the code are averaged over the exposure unit area, no further adjustment may be needed for certain pathways.

2.0 EXPOSURE TO EXTERNAL RADIATION

The RESRAD computer code allows the user to estimate the exposure from gamma-emitting radionuclides in soil. A conservative assumption made in the RESRAD code is that the receptor is located roughly at the center of a homogeneously contaminated area. Because the dose contributions from external gamma exposure are not a linear function of area size, the exposure unit area concept is not valid for this pathway. The RESRAD code adjusts the dose automatically as a function of area size. Therefore, the user should enter the concentrations of contaminants that are present in the area of contamination rather than dilute the concentrations over the receptor-specific exposure unit.

The following parameters are important in calculating the external dose:

- area, thickness, and bulk density of soil in the contaminated zone;

- thickness and soil density of the cover (for subsurface contamination);
- fraction of time during the year that the receptor spends outdoors, indoors, and away from the contaminated area; and
- shielding from external gamma radiation afforded by buildings during indoor occupancy.

Adjusting the dose to reflect the shape of the contaminated zone is recommended only when the contaminated area is less than 1,200 m^2 and when the shape deviates considerably from that of a circular area. Appendix A of the RESRAD manual provides instructions on how to calculate the shape factor for a noncircular contaminated area.

In the case of subsurface contamination, two cases may be run. To assess current exposures, the exposure scenario is modeled by entering the thickness and density of the material covering the subsurface contamination. For future exposures, the contaminated soil is assumed to be redistributed over an area encompassing the exposure unit. In this case, no cover is assumed to be present, and the average thickness of the contaminated zone covering the exposure unit area will be equal to the excavated volume divided by the exposure unit area or 15 cm, whichever is smaller.

The fraction of time spent indoors or outdoors entered in the RESRAD code is calculated based on annual averages. These fractions can be calculated as follows:

$$f_i = \frac{ET_i \times EF}{8{,}766}, \text{ and}$$

$$f_o = \frac{ET_o \times EF}{8{,}766},$$

where

f_i = fraction of year spent indoors (dimensionless),

f_o = fraction of year spent outdoors (dimensionless)

ET_i = exposure time indoors (hours per day),

ET_o = exposure time outdoors (hours per day),

EF = exposure frequency (days per year), and

8,766 = number of hours in a year (hours per year).

The RESRAD default value for the shielding factor during indoor occupancy is 0.7 (i.e., indoor exposure levels are 30% lower than outdoor levels).

3.0 INHALATION OF CONTAMINATED DUST

The RESRAD code uses a mass-loading model to estimate the concentration of contaminants in air resulting from resuspension of contaminated soil. The mass-loading model assumes that the contaminant concentration in airborne dust is the same as the concentration in soil adjusted for the presence of a ground cover (e.g., grass, pavement), depth of the contaminated soil, and area size.

The following parameters are important in calculating the dust inhalation dose:

- inhalation rate;
- mass loading of respirable particulates;
- fraction of time during the year that the receptor spends outdoors, indoors, and away from the contaminated area;
- reduction in outdoor dust levels afforded by buildings during indoor occupancy;
- thickness of soil in the contaminated zone; and
- thickness of the cover (for subsurface contamination).

Inhalation rates entered in RESRAD must be converted to units of cubic meters per year. If separate inhalation rates are given for outdoor and indoor activities, a time-weighted average inhalation rate should be entered. (Time spent away from the contaminated area should not be included in the time-weighted average.) In obtaining this average, any reductions in the dust levels during indoor occupancy must also be considered. The following equation should be used to derive the time-weighted indoor/outdoor inhalation rate:

$$IR = EF \times \frac{(0.4 \times ET_i \times IR) + (ET_o \times IR_o)}{(0.4 \times ET_i) + ET_o},$$

where

IR = weighted average annual inhalation rate (cubic meters per year),

EF = exposure frequency (days per year),

0.4 = reduction factor for indoor dust that is of outdoor origin (dimensionless),

ET_i = exposure time indoors (hours per day),

IR_i = indoor inhalation rate (cubic meters per hour),

ET_o = exposure time outdoors (hours per day), and

IR_o = outdoor inhalation rate (cubic meters per hour).

Time fractions spent indoors and outdoors are calculated in the same way as that for the external exposure pathway.

A cover and depth factor calculated by RESRAD accounts for mixing of contaminated soil that may occur with underlying or overlying uncontaminated soil. If no cover is present and the contaminated layer is thicker than the mixing depth (set to 1 cm for the inhalation pathway, assuming no gardening), the cover and depth factor calculated by the code is 1 (i.e., no dilution). This will be the case for most surface contamination scenarios. If subsurface contamination is present and is assumed to be redistributed over the exposure unit area, RESRAD calculates a cover and depth factor less than 1 if the thickness of contaminated soil, averaged over the entire exposure unit area, is less than 1 cm.

In RESRAD, the area of the contaminated site and a dilution length are used to calculate an area factor that accounts for dilution from uncontaminated sources. This area factor is calculated as

$$FA = \frac{\sqrt{A}}{\sqrt{A} + DL},$$

where

FA = area factor for inhalation pathway,

A = area of contaminated zone (square meters), and

DL = dilution length (meters).

Conservative results that neglect dilution with uncontaminated dust from outside the exposure unit can be obtained by forcing the area factor to equal 1, which is accomplished by setting the value of DL to 0.

4.0 INHALATION OF RADON

The RESRAD code allows the user to estimate the dose to an onsite receptor resulting from inhalation of ^{222}Rn and ^{220}Rn (and their short-lived progeny) emanating from soil contaminated with radionuclides in the natural uranium and thorium decay series. Except for the mass-loading factor and the dust reduction factor for indoor occupancy, the same parameters that are important in the dust inhalation pathway are also important in the radon inhalation pathway. In addition, the following parameters are also important:

- density and porosity of cover and contaminated soil;
- density, thickness, and porosity of building foundation;
- diffusion coefficient or volumetric water content in cover, contaminated soil, and foundation;
- building height and air exchange rate;
- average annual wind speed; and

- radon emanation fraction.

In the absence of site-specific data for the above parameters, the RESRAD defaults may be used because they are representative of national average values.

For combinations of indoor and outdoor occupancy, the time-weighted average inhalation rate is calculated in the same way as for the dust inhalation pathway, except that the dust reduction factor for indoor occupancy is set to 1 rather than to 0.4 (i.e., no reduction).

The maximum radon dose may not occur at time 0 unless only the immediate precursors are present in the soil. Instead, the radon dose may increase over time as the result of ingrowth of radionuclides in the uranium or thorium decay series.

5.0 INGESTION OF CONTAMINATED SOIL

Doses from soil ingestion can be calculated using the RESRAD code. In the RESRAD model, the cover and depth factor derived for the inhalation models also applies to the soil ingestion model. However, RESRAD uses an area factor that is set to 1 if the contaminated area is greater than or equal to 1,000 m^2; otherwise it is the area divided by 1,000 m^2. Because the concentrations used in the RESRAD input will have already been adjusted for the appropriate exposure unit, an area of 1,000 m^2 or greater should be entered. Other important parameters include the soil ingestion rate and the fraction of time during the year that the receptor spends outdoors, indoors, and away from the contaminated area.

Because RESRAD allows the input of a single soil ingestion rate, an age-weighted average value is required to account for increased intakes during childhood years:

$$IR = 10^{-3} \times EF \times \frac{(IR_c \times ED_c) = (IR_a \times ED_a)}{ED_c + ED_a},$$

where

IR = age-weighted average soil ingestion rate (grams per year),

10^{-3} = conversion factor (grams per year),

EF = exposure frequency (days per year),

IR_c = child soil intake rate (milligrams per year),

IR_a = adult soil intake rate (milligrams per year),

ED_c = child exposure duration (years), and

ED_a = adult exposure duration (years).

Because the intake rate calculated above already factors in the exposure frequency, no correction is required for time spent inside, outdoors, or offsite. To prevent RESRAD from further correcting the soil intake rate, the parameter for time fraction outdoors onsite should be set to 1 (the time fraction indoors should be set to 0).

For workers that spend all their time indoors, the intake rate in milligrams per day should be multiplied by the exposure frequency in days per year and converted to grams per year. The RESRAD code automatically reduces the result by a factor of 0.4 (using the shielding factor for inhalation) to account for the fraction of soil ingested that originates outdoors. In this case, the fraction of time spent indoors should be set to 1 (the time fraction spent outdoors should be set to 0).

For a combination of outdoor and indoor activities, the time fractions are calculated as

$$f_i = \frac{ET_i}{ET_i \times ET_o} \text{ , and}$$

$$f_o = 1 - f_i \text{ ,}$$

where the above parameters have been previously defined.

6.0 INGESTION OF CONTAMINATED PRODUCE

The RESRAD code can be used to estimate the radiation dose from ingestion of produce grown on contaminated soil. The model used in RESRAD accounts for the transfer of radionuclides in plants resulting from root uptake and foliar deposition. The contribution from contaminated irrigation water is set to 0, explicitly by setting the irrigation rate to 0 or implicitly by setting the radionuclide distribution coefficients in soil to a very large number (thus suppressing groundwater transport).

Because a relatively small area is required to grow produce for personal consumption, the exposure unit concept does not apply to the plant ingestion pathway. Concentrations entered in the code should be derived from the actual area of contamination. To prevent the RESRAD code from applying an area correction factor for small areas, the area of contamination should be entered as 1,000 m^2 or larger.

Other parameters in RESRAD important in calculating doses from ingestion of produce are

- produce consumption rates (leafy vegetables and other vegetables, fruit, and grain);
- soil-to-plant contaminant transfer factors (for root uptake);
- depth of plant roots;
- soil mixing layer; and
- mass loading of dust in air (for foliar deposition).

The ingestion rates entered in the code should be the annual average consumption of produce assumed to be grown or picked onsite and should not include any imported produce. Although it does not make a difference in the calculations which category of produce is selected (leafy vegetables or other vegetables, fruits, and grains), it is recommended that the consumption rate of leafy vegetables be set to 0.

The RESRAD code uses a mass-loading parameter for foliar deposition of resuspended contaminants that is not consistent with the approach presented in Section 3.3.2.1.4. This parameter should be set to 0 in the code input. To account for the differential transfer of contaminants through root uptake or foliar deposition, the transfer factor libraries should be modified using the following procedure.

In Section 3.3.2.1.4, a methodology is developed to account for the presence of surface and/or subsurface zones of contamination, each with potentially different concentrations of contaminants. Contaminants transferred to plants through foliar deposition originate from the surface layer of soil, whereas the contaminants transferred to plants through root uptake may originate from surface as well as from subsurface layers, depending on root depth.

Two separate runs are required to calculate the transfer of contaminants from soil to roots and from soil to leaves. In each run, the concentrations entered in the code are as follows:

- for root uptake, the concentrations may be entered either as the weighted average concentrations over the length of the roots (setting the thickness of the contaminated zone equal to the root depth) or as the surface concentrations if no subsurface contamination is present (setting the thickness of the contaminated zone to 15 cm), and

- for foliar deposition, the concentrations in the topmost 15 cm of soil should be entered.

For root uptake, the RESRAD default parameters for root uptake transfer factors may be used, or they may be changed to site-specific values if such data are available. For foliar deposition, values for a mass-loading transfer factor may be obtained for different types of vegetation from NUREG/CR-5512; these are not contaminant-specific but differ according to plant type. Before using them in RESRAD, these mass-loading transfer factors must be converted from a dry-plant- weight basis to a wet-weight basis using conversion factors provided in NUREG/CR-5512.

The dose contributions from the root uptake and foliar deposition runs are summed to obtain the total dose from ingestion of produce. However, RESRAD automatically assumes that only half of the produce ingested is grown onsite. Therefore, this total dose must be multiplied by a factor of 2.

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