LIFE-CYCLE RISK ANALYSIS FOR DEPARTMENT OF ENERGY (DOE) BURIED WASTES

Volume I

By

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To my amazing daughter, Sophia, wise beyond her years

and most of all

To my beloved wife, Chrisie, infinitely supportive

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DISCLAIMER

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TABLE OF CONTENTS

Appendix

LIST OF TABLES

LIST OF FIGURES

Figure **Page**

LIST OF EXHIBITS

LIST OF ABBREVIATIONS

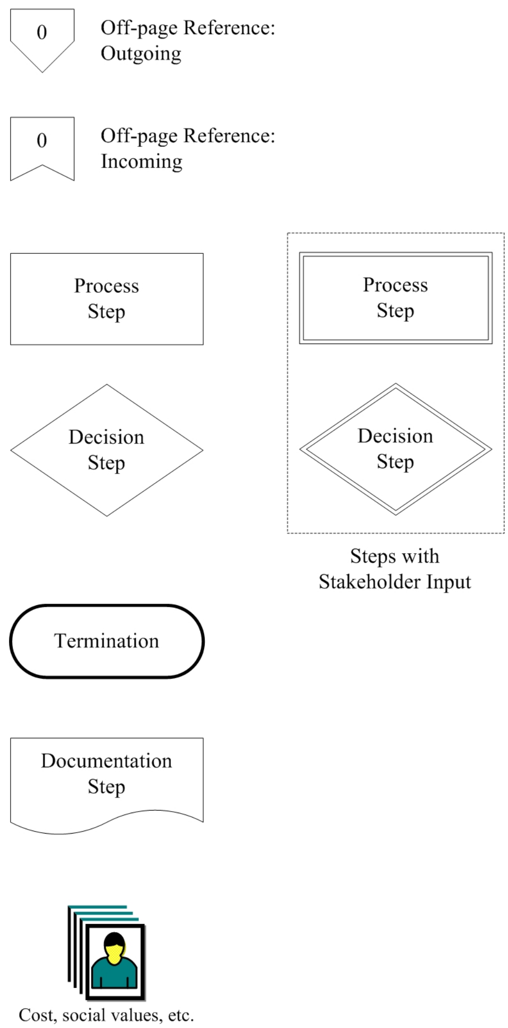
ABRA	Ancillary Basis for Risk Assessment
ALOHA	Areal Locations of Hazardous Atmospheres
AMWTF	Advanced Mixed Waste Treatment Facility
ANPP	Annual Net Primary Production
ARARs	Applicable or Relevant and Appropriate Requirements
ARP	Accelerated Retrieval Process
ATSDR	Agency for Toxic Substances and Disease Registry
BCBG	Bear Creek [Valley] Burial Ground
BCM	Bear Creek Marker
BCV	Bear Creek Valley
BLS	[U.S.] Bureau of Labor Statistics
BRA	Baseline Risk Assessment
CAB	Citizen's Advisory Board
CAIRS	Computerized Accident Incident Reporting and Recordkeeping System
CAMEO	Computer-Aided Management of Emergency Operations
CAS	Chemical Abstracts Service
CBSM	Conceptual Burial Site Model
CCA	Cause-consequence analysis
CCDF	Complementary Cumulative Distribution Function
CDI	Chronic daily intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act (also known as "Superfund")
CFR	Code of Federal Regulations
COC(s)	Contaminant(s) of Concern
COPC(s)	Contaminant(s) of Potential Concern
CRESP	Consortium for Risk Evaluation with Stakeholder Participation
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
DCF	Dose Conversion Factor
De minimus	*de minimus non curat lex* or "the law does not concern itself with trifles"
DNFSB	Defense Nuclear Facilities Safety Board
DOE	U.S. Department of Energy
DOE-EM	U.S. Department of Energy Office of Environmental Management
DOL	U.S. Department of Labor
DUST-MS	Disposal Unit Source Term – Multiple Species
EDTA	Ethylenediaminetetraacetic Acid
EIS	Environmental Impact Statement

EPA	U.S. Environmental Protection Agency
ERA	Ecological Risk Assessment
ERDF	[Hanford] Environmental Restoration Disposal Facility
ESHRAP	Environment, Safety, and Health Risk Assessment Program
ETA	Event Tree Analysis
FDA	U.S. Food and Drug Administration
FDCA	Food, Drug, and Cosmetic Act of 1938
FEP	Features, Events, and Processes
FGR	Federal Guidance Report
FMEA	Failure modes & effects analysis
FMECA	Failure modes & effects/criticality analysis
FRAMES	Framework for Risk Analysis in Multimedia Environmental Systems
FTA	Fault Tree Analysis
HAZOP	HAZard and OPerability
HDT	Historical Data Task
HEAST	Health Effects Assessment Summary Tables
HFA	Human factors analysis
HHEM	Human Health Evaluation Manual
HI	Hazard Index
HLW	High-level Waste
HQ	Hazard Quotient
IARC	International Agency for Research on Cancer *Centre International de Recherche sur le Cancer*
ICDF	Idaho Site CERCLA Disposal Facility
ICP	Idaho Cleanup Project
ICRP	International Commission on Radiological Protection
IN[EE]L	Idaho National [Engineering and Environmental] Laboratory
IRA	Interim Risk Assessment
IRIS	Integrated Risk Information System
ISG	*In Situ* Grouting
ISTD	*In Situ* Thermal Desorption
ISV	*In Situ* Vitification
LCA	Life-Cycle Assessment
LCF	Latent Cancer Fatality
LEFPC	Lower East Fork Poplar Creek
LET	Linear Energy Transfer
LOAEL	Lowest-observed-adverse-effect-level
MC	Monte Carlo
MIP	Manage In Place
NAS	National Academy of Sciences

NAICS	North American Industry Classification System
NCRP	National Council on Radiation Protection and Measurements
NIST	National Institute of Standards and Technology
NOAEL	No-observed-adverse-effect-level
NPL	National Priorities List
NRC	U.S. Nuclear Regulatory Commission
NT	North Tributaries (Bear Creek Valley, Oak Ridge Reservation)
ORNL	Oak Ridge National Laboratory
ORP	Office of River Protection
ORR	Oak Ridge Reservation
OSHA	Occupational Safety and Health Administration
P/CCRARM	Presidential/Congressional Commission on Risk Assessment and Risk Management
PA	Performance Assessment
PCB	Polychlorinated Biphenyl
PCE	Tetrachloroethylene
PDCF	Pathway Dose Conversion Factor
PDF	Probability Distrbution Function
PHA	Process Hazard Analysis
PM	Particulate Matter
PMF	Probability Mass Function
PPA	Probabilistic Performance Assessment
PRA	Probabilistic Risk Assessment
PRG(s)	Preliminary Remediation Goal(s)
PVC	Polyvinyl Chloride
QRA	Quantitative Risk Assessment
RAGS	Risk Assessment Guidance for Superfund
RAIS	Risk Assessment Information System
RCRA	Resource Conservation and Recovery Act
RfD	Reference Dose
RFP	Rocky Flats Plant
RI/FS	Remedial Investigation/Feasibility Study
RME	Reasonable Maximum Exposure
RSD	Relative Standard Deviation
RT	Radionuclide Transport
RTD	Retrieve, Treat, and Dispose
RWMC	Radioactive Waste Management Complex
SDA	Subsurface Disposal Area
SEAM	Superfund Exposure Assessment Manual
SNF	Spent Nuclear Fuel

SRM	Simplified Risk Model
SRPA	Snake River Plain Aquifer
SSL(s)	Soil Screening Level(s)
TCE	Trichloroethene
TEDE	Total Effective Dose Equivalent
TRU	Transuranic
TRUPACT	TRansUranic PACkage Transporter
VC	Vinyl Chloride
VSD(s)	Virtually Safe Dose(s)
VVE	Vapor Vacuum Extraction
WBS	Work Breakdown Structure
WILD	Waste Information and Location Database
WIPP	Waste Isolation Pilot Plant
YM	Yucca Mountain

FRAMEWORK SYMBOLS



CHAPTER I

INTRODUCTION

Dissertation Overview

Before 1970, hundreds of thousands of cubic meters of transuranic (TRU), low-level, and mixed low-level wastes generated from nuclear materials production were buried at various sites across the U.S. Department of Energy (DOE) Complex. Most of these wastes were buried in shallow unlined pits and trenches and covered with soil creating the potential for contaminant migration and exposure with concomitant safety and health concerns.

Not only is the buried waste inventory large but the waste types are highly variable and often inter-mixed with hazardous wastes making their retrieval, treatment, and disposal highly problematic. For example, many of the TRU constituents buried at these sites possess very long half-lives and can pose hazards through a variety of exposure pathways. Inconsistency in regulatory approach and agreements, including historic DOE management, concerning disposal alternatives (i.e., manage the wastes in-place or retrieve the wastes for treatment and disposal either on- or off-site) provides neither a consistent basis for site remediation nor transparency to a range of stakeholders.

There were two previous stages to developing the framework for assessing the life-cycle risks for the disposition of DOE buried wastes. The first stage, requested of the Consortium for Risk Evaluation with Stakeholder Participation (CRESP)[1] by the U.S. DOE Office of Environmental Management (DOE-EM), began in the summer of 2002

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[1] The CRESP website is http://www.cresp.org (accessed March 13, 2008) (CRESP-II 2006).

with the intent to formulate a framework for the consistent technical evaluation of the life-cycle risks associated with DOE buried transuranic (TRU) waste[2] disposition. Because TRU wastes are often not the primary risk drivers for DOE sites, the research topic was broadened in November 2004 to a framework for both radioactive and hazardous waste when CRESP elected to develop preliminary risk evaluations for two contaminated areas at the Idaho Site (Brown et al. 2005; Switzer et al. 2005).

The focus of one of the CRESP studies was the Idaho Site Subsurface Disposal Area (SDA) in which waste contaminated with both radioactive (including TRU) and hazardous materials were buried in pits, trenches, and soil vaults between 1952 and 1970. The initial risk evaluation for the SDA constituting the second research stage was completed in June 2005 (Brown et al. 2005). The SDA study has been an important contribution to this research because the SDA reflects many important characteristics of DOE buried waste sites (especially in arid regions). The SDA has been selected as one of two prototypic sites that will be used to demonstrate the flexibility, usefulness, and value of the proposed framework and methodology in providing the information needed to make a risk-informed decision[3].

To provide a foundation for risk-informed decision-making, a framework is developed here for the transparent and consistent technical evaluation of the life-cycle risks and risk trade-offs (both to the general public and workers) associated with buried waste disposition and site remediation. Risk is one of the inputs needed (along with costs, technical feasibility, cultural and societal impacts, etc.) to make a risk-informed decision.

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[2] Transuranic (TRU) waste contains more than 3,700 Becquerels (100 nanocuries) of alpha-emitting TRU isotopes (e.g., ^{238}Pu, ^{239}Pu, ^{241}Am, etc.) per gram of waste, with radioactive half-lives greater than 20 years.

[3] A "risk-informed" approach is a philosophy whereby risk information is considered with other, non-risk factors to better focus attention on issues commensurate with their importance to health and safety.

Use of this framework to provide the risk information needed will differ from existing approaches by providing a basis for evaluating relevant risk tradeoffs involving the general public and workers in a consistent and transparent manner. The framework is applied to the buried wastes areas from two DOE sites with very different climactic conditions.

Research Goals

The primary goal of this research is the development of a generalized life-cycle risk analysis framework and methodology for the consistent and transparent assessment of the risks and risk trade-offs associated with disposition of buried U.S. Department of Energy (DOE) wastes and site remediation. The buried wastes in question were generated from historic nuclear material production and are currently managed by the DOE. The results from the application of this framework can be used as one input, along with other non-risk factors, to a risk-informed decision-making process.

Hypothesis Testing

The hypotheses described below are tested to evaluate the effectiveness, flexibility, and value of the general approach developed here to assessing life-cycle risks for DOE buried waste sites[4]. The effectiveness of the approach is demonstrated by providing the risk and uncertainty information generated through application of the framework to DOE buried waste sites. The flexibility of the approach is demonstrated by

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[4] In this research, the terms "hypothesis" and "testing" are used in a somewhat less formal sense than usually found in the scientific method and statistics. The *hypotheses* defined in this research are indeed "testable statements;" however, the *tests* for the *hypotheses* are more subjective in nature and based not on quantitative data and statistical tests but instead on expert analysis and/or model predictions.

applying the framework to two different DOE buried waste sites in very different climatic and geologic settings. These sites are the Bear Creek Burial Grounds (BCBG) at the Oak Ridge Reservation (ORR) and the Subsurface Disposal Area (SDA) at the Idaho Site[5]. Finally, the value of the approach is demonstrated by showing that the risk and uncertainty information generated using the framework facilitates risk-informed decision making.

The information generated by application of the framework to a buried waste site is the human health risk and uncertainty inputs to the decision-making process. The non-risk factors (e.g., costs, technical feasibility, cultural and societal impacts, etc.) are not generated during application of the framework but will be needed to make an informed decision. However, having comprehensive and consistent risk information as input to the decision-making process is essential to making a truly informed decision concerning the disposition of DOE buried waste sites.

Retrieval versus Manage In-Place

Primary Research Hypothesis for SDA: For the Idaho Site Subsurface Disposal Area (SDA), the remedial alternative involving managing buried wastes using *in situ* techniques, barriers, etc. will result in lower life-cycle risks to potentially impacted receptors than the alternative whereby buried wastes are retrieved and treated for disposal off-site.

Primary Research Hypothesis for BCBG: The retrieve, treat, and dispose alternative for the Bear Creek Burial Grounds (BCBG) located in the humid conditions

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[5] The selection of these sites as prototypic sites and their characteristics is discussed in greater detail in Chapter IV.

on the Oak Ridge Reservation (ORR) will result in lower life-cycle risks than managing the wastes using *in situ* techniques.

At the most general conceptual level, there are two alternatives available for the disposition of buried wastes: 1) retrieve the wastes for treatment and disposal either on- or off-site or 2) manage the buried wastes *in situ*. The initial set of research hypotheses that is tested concern which of these two remedial alternatives will likely result in lower life-cycle risks based upon the types of contaminants and conditions at the buried waste site. For example, if the buried waste site is located near the water table and contains potentially mobile and persistent contaminants in a humid region with significant rainfall and infiltration[6], then retrieval of the buried wastes is likely more warranted than for a site in an arid region where mobilization of contaminants through infiltration and leaching can be minimized through the use of engineered barriers and institutional controls. Therefore, different hypotheses are warranted for the two sites.

Because there is currently no *feasible* method to render buried radioactive and many hazardous contaminants non-radioactive and/or non-hazardous,[7] ultimate disposal of these wastes must include isolation and containment to the extent possible while 1) wastes remain hazardous, 2) there are potential receptors that can be impacted, and 3) there is a possible pathway from the waste to a potential receptor that can result in unacceptable risks. Containment of contaminants can be effected either in-place (at the

[6] Water that percolates through the buried wastes is the primary driver for both contaminant release from the buried wastes and transport through the environment.

[7] Only with sufficient time and decay will radioactive contaminants become stable; whereas, hazardous metals will remain hazardous. Transmutation of radionuclides is not considered economically feasible at this time or at any time in the foreseeable future; therefore, transmutation is not considered a viable remedial option for this research.

original buried waste site in a new, engineered disposal cell) or at an alternative (and likely more stable) disposal location off-site; however, there are very real risks associated with waste retrieval, handling, and shipping that may make the retrieval alternative less attractive, especially considering that the retrieval alternative ultimately constitutes a risk transfer (albeit hopefully to a more stable and controlled environment).

Combination of Actions Provide Lowest Risk

Research Hypothesis: The remedial alternative that results in the lowest life-cycle risks to potentially impacted receptors is a combination of *in situ* techniques and targeted retrieval actions taken, if possible, in different areas of the disposal site.

Both the Idaho Site Subsurface Disposal Area (SDA) and the Oak Ridge Bear Creek Burial Grounds (BCBG) are very complicated buried waste sites containing both radioactive and hazardous wastes (often inter-mixed) in various forms. Because of the complexity of these sites, it is unlikely that a single remedial alternative is best for the entire buried waste site. This hypothesis involves investigating whether or not a combination of *in situ* techniques and targeted retrieval actions, if possible, will likely be the most effective remedy.

Furthermore, independent evaluation of separate areas within a buried waste disposal site may not result in the optimum risk management strategy for the site as a whole. Many buried waste sites in the DOE Complex are large and highly complex. Remedial actions that are appropriate in some areas may actually increase life-cycle risks in other areas and may lead to an overall increase in life-cycle risks for the area as a whole. For example, depending upon the time frame under consideration, radioactive constituents may not always be the primary risk drivers for remedial decisions at these sites and, therefore, both radioactive and hazardous constituents must be considered as part of the decision-making process that considers all significant, life-cycle risks.

All Significant Sources of Risk Considered to be Risk-Informed

Research Hypothesis: The significant sources of exposure and accident risks for both general public and workers (in addition to non-risk factors such as costs, technical feasibility, cultural and societal impacts, etc.) must be considered for each remedial alternative for the decision to be risk-informed.

This research hypothesis considers what sources of risk information for each remedial alternative should be incorporated into the decision-making process (in addition to other non-risk factors such as costs, technical feasibility, cultural and societal impacts, etc.). Temporal variations in the sources of risk are also important considerations in evaluating remedial alternatives. For example, consideration of only potential long-term public health impacts from radiological and hazardous chemical exposures while ignoring short-term worker risks from accidents during remedial activities does not provide a complete (and arguably equitable) picture of the true risks involved with buried waste disposition. Only the open consideration of all significant sources of risk to all potential receptors provides the comprehensive risk input needed for an equitable and transparent decision. Different decision makers and stakeholders may consider the different types of risks amongst the various potential receptors (e.g., workers, off-site public, current, and future generations) and non-risk factors differently, reflecting individual and organizational values; however, all significant sources of risk must be made available so that an informed decision can be made.

Research Objectives

The research objectives are described here proceeding from the general to the specific. The first objective is to develop a general life-cycle risk analysis framework for the disposition of buried DOE wastes. The framework is essentially the rational, graphical representation of the process for the consistent and transparent evaluation of the risks and uncertainties associated with buried waste site disposition. A corresponding methodology is developed describing how the framework is applied to buried waste sites in the DOE Complex. A screening risk tool is developed in the GoldSim simulation software (GTG 2005a; b) that implements the screening parts of the framework for DOE sites in both arid and humid climates. The framework, methodology, and screening risk tool are applied to two prototypic sites illustrating the effectiveness, flexibility, and value of the framework in providing the risk information needed to make an informed remedial decision.

Risk Analysis Framework

The first and foremost research objective is the development of a general life-cycle risk analysis framework for assessing the life-cycle risks and risk trade-offs associated with DOE buried waste disposition. The conceptual framework outlines the general process for estimating and comparing the risks and risk trade-offs involved with either 1) managing buried wastes in-place or 2) retrieving, treating, and disposing wastes in an appropriate on- or off-site area[8]. The risk analysis framework is both iterative and tiered so that each successive assessment tier builds on each preceding phase (when necessary) and represents an increase in sophistication (e.g., in terms of modeling) and required site-specific information to better characterize or reduce uncertainty and increase the accuracy in the risk input to the decision-making process.

Risk Analysis Methodology

A methodology for applying the framework to compare the risks and risk trade-offs associated with the two general disposition alternatives for buried waste sites is developed in concert with the framework. The steps and types of information (e.g., diagrams, screening tools, etc.) needed to produce meaningful risk analyses for potential remedial actions for buried waste sites are defined. A rational approach is developed for managing uncertainties and missing information in such a way that this additional dimension of risk can be incorporated when risks and risk trade-offs associated with potential remedial alternatives are compared. Guidelines for defining appropriate comparison metrics are provided and a set of test metrics will be defined for use in this research.

Conceptual Burial Site Model and Screening Risk Analysis Tool

A novel conceptual burial site model is defined that describes both the exposure and standard industrial risks associated with remedial actions for DOE buried waste sites to be made. This conceptual model is novel because it allows for an integrated,

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[8] The risks and risk trade-offs associated with any needed transportation and final disposal of the retrieved wastes in a new site must also be included to provide a comprehensive and life-cycle-based comparison.

comprehensive, and transparent analysis of the significant risks confronting the disposition of a contaminated waste site. The conceptual model was implemented in the GoldSim simulation software and can be used for DOE sites in both arid and humid conditions.

Framework and Methodology Application

The framework, methodology, and screening risk tool are applied to two prototypic sites to illustrate the effectiveness, flexibility, and value of the approach to promote consistency in planning for the disposition of buried waste across the DOE Complex. Previous experience indicates that the Oak Ridge Reservation (ORR) Bear Creek Burial Ground and Idaho Site Subsurface Disposal Area (SDA) are useful candidates for prototype site selection (Brown et al. 2005). These sites appear to bracket the types of contaminants, hazards, and conditions that are expected from the various DOE sites.

Application of the framework, methodology, and screening risk tool on these prototype sites illustrates the effectiveness, flexibility, and value of the framework and methodology in providing the risk information needed to make an informed remedial decision. The results obtained from the framework, methodology, and screening risk tool differ from existing approaches by providing a basis for evaluating all relevant risk tradeoffs involving the general public and workers in a consistent and transparent manner.

Significance and Contribution of the Research

The goal of this research is the development of a comprehensive, life-cycle risk analysis framework and methodology for the disposition of Department of Energy (DOE) buried waste sites that is straightforward and efficient to apply and results in a consistent and transparent evaluation of the disposition risks. To truly improve the risk assessment process and its acceptance may be best effected by focusing efforts on consistency, transparency, and trust in the process rather than by attempting to improve the technical components that will likely never be fully understood (and perhaps trusted) by some regulators and many stakeholders alike. The risk analysis framework and methodology developed in this research provides a mechanism for providing the consistency and transparency not delivered by other such frameworks.

The methodology for applying the framework to the disposition of buried waste sites promotes consistency and transparency in developing the risk information needed for informed decision-making. The methodology requires that the following elements be developed:

- *Site conceptual models* are needed for the baseline site conditions linking contaminant sources to potentially impacted receptors (both public and occupational) and describing graphically why remedial action is likely required.
- *Comparison metrics* are defined that form a reasonable basis for how risks and other information (e.g., dose, hazard quotient, etc.) obtained from the analysis can be compared.
- For each acceptable remedial alternative, the following information is required:
    - *Task list* and corresponding *management flow diagram* outlining the steps required to execute the remedial alternative.
    - *Novel conceptual site models* relating the natures of the hazards and risks during remedial activities to potentially impacted receptors.

- *Hazard analysis* identifying (for each process step) the task frequency, elements of risk, potentially impacted population, basis for characterizing the risk, and contribution of the remedial task to overall risk.

- *Risk flow diagram* indicating the sequence of remedial and stewardship activities with potential to pose significant human health risks and incorporating *conceptual models* that indicate potential hazards, failure and release modes, transport pathways or media, exposure mechanisms, and impacted receptors.

- *Gap analysis* describing the key knowledge barriers, missing information, variabilities, and uncertainties involved in assessing risks for the remedial alternative.

- *Integrated hazard and gap analysis* summarizing the most important potential risks and information gaps for the remedial alternative.

- *Risk breakdown* indicating the risks associated with the proposed remedial alternative as they relate to types of risks (e.g., chemical exposure, radiation, traumatic injury, etc.) and potential receptors. Indications of the uncertainties associated with the risks and their potential impacts on the decision process must be included. *Comparison metrics* should also be evaluated.

- *Life-cycle risk breakdown* indicating the life-cycle risks for proposed remedial alternatives as they relate to types of risks and potential receptors. An explicit declaration of the value judgments and simplifying assumptions made by the risk assessor must be made as well as the likely impact of significant uncertainties on the risk estimates.

The above components of the risk analysis information help focus the assessment during subsequent phases and provide a basis for the consistent and transparent comparison of potential remedial alternatives.

There are a number of additional qualities of the risk analysis framework and methodology that lend consistency and transparency and ultimately trust to the risk results obtained. The risk assessment process is *tiered* so that the level of detail in the analysis of both risk and uncertainty and the types of simplifying assumptions tolerated are commensurate with the importance, complexity, and stage of the buried waste site disposition. The risk analysis framework is also *iterative* so that the risk assessment can be updated as new information is obtained, new questions are asked, or regulations are changed. Risk assessment should be thought of as a journey much more than a goal. This journey, to which all interested parties are invited by the framework and early, addresses all relevant types of risk and considers the impacts of uncertainty consistently and transparency so that trust can be engendered.

The risk assessment framework and methodology integrate the concepts of exposure *and* standard industrial risks to all potentially impacted receptors—both in the general public and workers. Despite advances in risk assessment techniques, there is a conspicuous absence of the consideration of standard industrial risks[9] in many risk assessment approaches despite indications that the predominant source of risk in site cleanup is industrial or occupational in nature (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). However, as important as the question of who is at risk is the question of when they are at risk. Workers tend to be most exposed and at risk during remedial actions; whereas, general public exposures may last for millennia. The temporal aspects of risk are also integral to the evaluations of risk in the framework.

A novel risk screening tool is developed that integrates many of the concepts in the life-cycle risk analysis framework and methodology. Although the primary product of this research is *not* software, the screening risk tool was developed to incorporate the basic concepts (i.e., integrating exposure and industrial risks, public and occupational receptors, temporal variation in risks, sensitivity and probabilistic capabilities, etc.) of the framework and methodology. The screening risk tool can thus be used to apply the

[9] Standard industrial risks are those non-exposure risks associated with falls, explosions, transportation accidents, etc.

concepts of the framework and methodology to a buried wastes site to evaluate the risks may be for potential remedial alternatives for a buried waste site.

No risk analysis framework or software tool can decide what should be done with a contaminated site. However, the life-cycle framework and methodology and the results from applying the risk screening tool can organize the evaluation process and assure that the evaluation is performed in a consistent and transparent manner. The risk and uncertainty results obtained from the evaluation can then be used as the risk input to the informed decision-making process.

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CHAPTER II

A REVIEW OF PRIOR USE OF RISK SUPPORTING LIFE-CYCLE RISK ANALYSIS FOR RISK-INFORMED REMEDIAL ACTION DECISIONS

Hundreds of thousands of cubic meters of hazardous and radioactive wastes generated from nuclear materials production have been—and still are being—buried at various sites across the U.S. Department of Energy (DOE) Complex. Until recently most of these wastes were buried in unlined pits and trenches and covered with soil creating the potential for contaminant migration and human exposure with concomitant safety and health concerns. Inconsistency in regulatory approach and agreements concerning disposal alternatives has provided neither a consistent basis for site restoration nor transparency to stakeholders.

One degree of consistency does appear to run through the regulatory approaches to evaluating potential risks to the public associated with buried waste disposition and site remediation. Recognizing that risk is one of the necessary inputs (along with costs, technical feasibility, cultural and societal impacts, etc.) to the decision-making process, use of the framework developed in this research can provide the relevant risk information to decision makers and stakeholders in a consistent and transparent manner. Risk assessments have been employed for decades to evaluate the "baseline" risks associated with the sites to determine whether the site requires remedial action and, if needed, often the extent to which remedial action must be taken. The basic process for risk assessment of human exposure to hazardous chemicals was formalized in *Risk Assessment in the Federal Government: Managing the Process* (a.k.a., the *Red Book*) (NAS 1983).

However, the concept of risk and its evaluation for contaminated sites were both around long before the *Red Book* formalized the process. The purpose of risk assessment has also evolved greatly since that time. Improvements have been made both in the technical processes that constitute the assessment; however, even more important strides have been made in how the public is informed and included in the evaluation. Perhaps most significantly it is now recognized that risk is but one of the necessary inputs (along with costs, technical feasibility, cultural and societal impacts, etc.) to a risk-informed decision-making process. This chapter describes the modern evolution of the concept of risk and its evaluation to provide the information necessary to make risk-informed decisions in a consistent and transparent manner.

The Meanings of "Risk" and "Risk Assessment"

The purpose of the current research is to provide the information necessary to assess the risks associated with the disposition of buried DOE wastes. So what is meant by the term *risk*? Many different definitions of risk can be found in the literature. One definition was provided by the influential Presidential/Congressional Commission on Risk Assessment and Risk Management (P/CCRARM) as the probability that a substance or situation will produce harm under specified conditions and is a combination of two factors: 1) the probability that an adverse event will occur and 2) the consequences of the adverse event (P/CCRARM 1997a). This definition is similar to that used in the 1975 U.S. Nuclear Regulatory Commission (NRC) Reactor Safety Study (NRC 1975):

$$Risk\left\{\frac{\text{consequence}}{\text{unit time}}\right\} = frequency\left\{\frac{\text{events}}{\text{unit time}}\right\} \times magnitude\left\{\frac{\text{consequence}}{\text{event}}\right\} \qquad [1]$$

The NRC currently uses the definition of risk provided by Kaplan and Garrick (1981) based on a risk-triplet that answers the following questions: 1) “What can go wrong?”, 2) “How likely is it?”, and 3) “What are the consequences?” (Callan 1998). A very different view of *risk* is held by Slovic and others who suggest that risk is a social construct based upon public perceptions and that whoever controls the definition of risk defines what is the rational solution to a given problem (Slovic 1987; 1999; 2000). Gephart (2003) provides an interesting fusion of these concepts by representing risk as:

$$Risk = probability\left(\text{likelihood}\right)\times consequence\left(\text{harm}\right)\times outrage\left(\text{fear}\right) \qquad [2]$$

although it would be difficult to quantify *outrage* (Sandman 1987) in the above expression.

The risk-triplet concept provided by Kaplan and Garrick (1981) is used in this research because it adequately captures other concepts of risk, is simple to understand, and defines risk at a fundamental level that can be applied to a broad range of activities. The risk-triplet can be used to describe both exposure and occupational hazards (e.g., traumatic injuries, falls, heat stress, noise, etc.) needed to evaluate life-cycle disposition risks. Usage of the risk-triplet concept does not conflict with Slovic and others’ views on risk being a social construct best considered during the decision-making process (Slovic 1987; 1999; 2000; 2003).

Whereas questions concerning likelihood and consequences may be considered subjective and value-laden by Slovic (2003), a quantitative framework for evaluating these risk elements has been developed (Garrick 2007; Kaplan and Garrick 1981; Kaplan et al. 2001). Furthermore, evaluation of the risks associated with those things that can go wrong during waste disposition and site remediation and their consequences does not preclude consideration of other factors (e.g., costs, technical feasibility, cultural and societal impacts, etc.) during the decision-making process. In fact, risks are one—albeit a critical—input to the decision-making process. The risk information provided from the approach taken here (based on the risk-triplet) provides consistency and transparency.

Elements of Risk: Including the Human and Temporal Dimensions

The risk-triplet concept has been applied most frequently to the area of nuclear power plant safety and so, in this context, the risk-triplet concept captures the needed information. However, in the context of human health risks typically associated with the disposition of buried wastes, two additional questions become critical[9]; these questions are: “Who are at risk?” (a.k.a., receptor identification) and “When are they at risk?” (a.k.a., temporal analysis). These final questions are closely related. Each of the (now five) questions must be answered when evaluating the disposition of buried waste sites across the DOE Complex.

The receptors potentially impacted by the hazards can vary greatly depending upon the remedial actions taken, as does the time frame that receptors would be at risk. For example, if buried wastes are managed in-place, then those potentially at risk tend to be large, off-site (often future) populations after long-lived contaminants have moved through the subsurface. Furthermore, any workers involved with maintaining or monitoring the waste site might be exposed. Thus, impacted populations may include both current and future members of the general public and on-site workers.

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[9] One can argue that the risk-triplet concept may account for these “additional” human and temporal dimensions. However, these issues are considered important enough to warrant specific mention.

The time to effect for buried wastes can be lengthened significantly if engineering barriers are used and/or the wastes are treated. If the wastes are to be retrieved and treated for disposal elsewhere, then the remedial workers tend to be those experiencing the most direct—and often acute—hazards. Significant hazards (including traffic accidents possibly involving the general public) also may be involved with transporting retrieved wastes to an off-site disposal area. The retrieved wastes would be transferred to a more stable disposal environment; however, the hazards associated with the wastes would not be eliminated entirely and most likely would impact a different set of receptors although the exposure risks would be reduced.

The types of hazards related to either managing the wastes in-place versus retrieving, treating, and disposing the wastes either on- or off-site can differ significantly. The types of hazards associated with managing the wastes in-place tend to be chronic effects from exposure to contaminated food and water; impacts that can take many years to manifest themselves, that may impact varying receptors differently, and that may be difficult to separate from other causes. Uncertainties associated with these types of long-term impacts tend to increase the level of discomfort with these types of effects and to foster continuing debate.

In comparison, hazards associated with the retrieval alternative involve not only long-term, chronic exposure effects but also acute, potential traumatic effects from construction-type and traffic accidents (e.g., explosions, fires, etc.) during retrieval, treatment, and transport operations. A waste site during retrieval activities may resemble a heavy construction site with its particular types of hazards; however, these sites hold additional exposure hazards because of the types of wastes being handled (Applegate and

Wesloh 1998; Gerrard 2002). Therefore, all types of hazards should be considered during the risk assessment and remedial selection processes to provide a comprehensive and transparent evaluation of risks.

The final—and often overriding—piece of the risk-triplet is the set of consequences presented by buried waste disposition. The potential consequences associated with disposition include illness, injury, genetic changes, latent cancer, and death. Because of the often large uncertainties involved, lay people often focus on "dread" and "unknown" risks[10] when deciding those hazards that should, be addressed (Slovic 1987). However, to provide a consistent and inclusive basis for comparisons of the risks associated with the disposition of buried wastes, the focus should be on all aspects of risk and not just the potential consequences and their magnitudes.

Elements of Risk Assessment: The *Red Book*

The risk-triplet has been shown to be a useful framework for assessing risks for nuclear power plants, space systems, and chemical processes (Garrick 1988) and thus bridges the concept of risk and the process of risk assessment. However, the customary basis for assessing human health risks for radiation and chemical hazards is the *Red Book* (NAS 1983). The process of risk assessment in the *Red Book* was comprised of all or parts of the following four steps (where emphases in *italics* and implicit assertions are added):

1) *Hazard identification*—determination of the link between a *chemical* and particular [human] health effect,

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[10] "Dread" risk is characterized by hazards with a perceived lack of control; catastrophic, fatal consequences; and an inequitable distribution of risks and benefits. "Unknown" hazards are judged to be unobservable, not known, new, and delayed in their manifestation of harm (Slovic 1987).

2) *Dose-response assessment*—determination of the relation between the magnitude of exposure [to a chemical] and likelihood of the [human] health effect,

3) *Exposure assessment*—determination of *human* exposure before and after action is taken, and

4) *Risk characterization*—description of nature and magnitude of *human* risk including uncertainty.

The definition of human health risk assessment in the *Red Book* made sense at the time and in the context for which it was defined—when primary concerns were the potential *human* health effects from exposure to toxic *chemicals* and the risk assessment process was perceived by many as a scientific, value-free practice. However, risk assessment has roots that are much older than the *Red Book*.

Human Health Risk Assessment: A Condensed History

Covello and Mumpower (1985) trace the nascence of risk assessment to the Asipu people of the Tigris-Euphrates valley about 3200 B.C.[11]; however, the current research here is focused on more formal and modern risk assessment techniques. Paustenbach (1995a; 1995b) indicates that modern health risk assessment began in about 1975; this is also around the same time that the first probabilistic risk assessment (PRA) was performed (Rechard 1999). The basis for performance assessment (PA), which in its probabilistic form is very similar to PRA, was not available until 1981 (albeit in draft form and later issued in 1987 (Rechard 1999)) and the first full PA was performed in

[11] Covello and Mumpower (1985) provide an excellent review of the history of risk assessment and risk management before the 20th Century. Other excellent sources of historical information include a series of papers by Paustenbach (1995a; 1995b; 2000; 2002) (where the focus of these papers is on dose-response analysis) and an article by Rechard (1999) examining the relationship between performance assessment and other types of risk assessment for radioactive waste disposal. Rhomberg (1997) provides a comprehensive survey of health risk assessment methods for various U.S. Federal agencies. Lester et al. (2007) provide a review of risk-related literature since 2000. Garrick (2007) provides a summary of the development of *quantitative* risk analysis techniques over several hundred years that have roots in probability theory.

1990 for the Waste Isolation Pilot Plant (WIPP) for which the PA methodology was developed (Rechard 1999). The initial Yucca Mountain (YM)[12] PA was issued later in the same year (Rechard 1999).

Whereas probabilistic risk analysis began in the 1970s for evaluating safety concerns for nuclear reactors, Friess (1987) suggests that the quantitative (health) risk assessment process as it is now understood began much earlier in the 20th Century, roughly in the 1930s[13]. This condensed history also begins around the same time. Figure 1 provides timelines of selected events (on the top timeline) that have influenced the development of risk and performance assessments and the regulatory and non-regulatory actions (on the bottom timeline) that illustrate the development of and influences on risk and performance assessments since 1930.

Figure 1 indicates some of the relationships that exist among the various events and regulatory decisions and the development of modern risk assessment techniques. However, even the numerous items illustrated in Figure 1 (which are admittedly only a small cross-section of those that could be shown[14]) do not capture the complete flavor of the breadth and complexity of the development of risk assessment techniques. A few of the major events and regulatory decisions and their relationships to modern risk assessment methods will be described here.

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[12] Yucca Mountain is the only site currently being studied (per the Nuclear Waste Policy Amendments Act of 1987 (USPL 1987)) for the deep geologic disposal of commercial spent nuclear fuel and defense high-level waste.

[13] The U.S. EPA suggests that risk assessment began as late as 1940 (USEPA 2001c).

[14] Rechard (1999) provides an excellent series of detailed illustrations showing the development of the various branches of science that contributed to modern risk assessment methods.

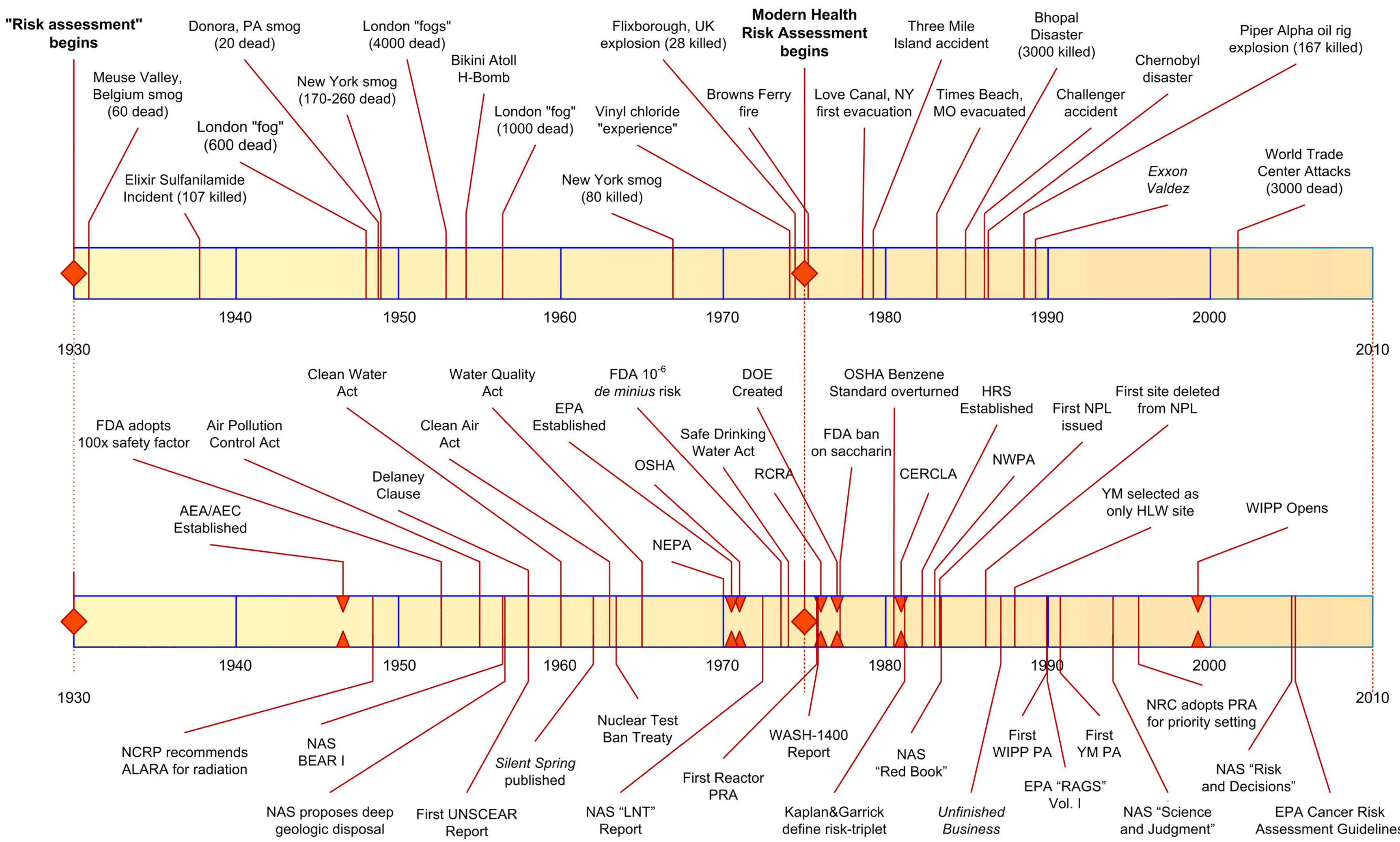

Figure 1. Top: Events that influenced the development of risk and performance assessments. Bottom: Regulatory and non-regulatory actions that illustrate the development of risk assessment since 1930. Adapted from Rechard (1999).

Most of the major events depicted in Figure 1 that resulted in major losses of life and that can be tied to regulatory actions are related to poor air quality (e.g., smog, "fog", etc.). Regulatory action for such problems began around the turn of the 14th Century when King Edward I of England attempted to forbid the burning of "soft" or "sea" coal in London because of the noxious fumes produced (Covello and Mumpower 1985; Paustenbach 2002). However, most health concerns before the 1940s tended to instead focus on infectious diseases (e.g., bubonic plague, influenza, etc.) and their causes because these diseases presented the greatest direct threats to human life (Paustenbach 1989; 2002). By the late 1940s, public health attention had turned to more "subtle and insidious" hazards because many serious infectious diseases had been eliminated or controlled due to better understanding and drugs (Eisenbud 1978; Paustenbach 2002).

Two major changes in the 20th Century led to the development and refinement of modern risk assessment techniques: nuclear power and exposure to man-made chemicals (MacDonald et al. 2004). The large increase in the production of chemicals since World War II was thought possible, if not probable to some, to have serious health impacts on both humans and the environment (Carson 1962). These concerns led to the development of more advanced techniques to attempt to establish the relationship between long-term chemical and radiation exposures and their potential health impacts (e.g., the dose-response relationship in health risk assessment) (Edler and Kitsos 2005; MacDonald et al. 2004). Alternatively, concerns with nuclear power safety led to the development of advanced quantitative techniques (with roots in the reliability and system analysis fields (Rechard 1999) and culminating with probabilistic risk assessment) to evaluate the likelihood of acute, catastrophic events related to safety system and reactor and support system component failures (Garrick 2007; MacDonald et al. 2004; Rechard 1999). Concerns with the potential human health impacts of acute and chronic exposures to the wastes resulting from both civilian and defense applications of nuclear technologies led to the development of performance assessment techniques[15].

No direct causal link has been definitively established between the large number of man-made chemicals that have been introduced into the environment and the overall rate of cancer and malignancy (Newby and Howard 2005; Paustenbach 2002). This lack of a direct link does not mean that no such link exists; it may instead indicate the difficulty of establishing such a link when large uncertainties and competing effects are involved. Despite the lack of establishing a general, causal link between the many new man-made chemicals that have been introduced in the environment since World War II and cancer rates, links between specific chemicals and corresponding health effects have been well established over the years and a few important ones will be discussed in relation to their impacts on the development of health risk assessment techniques.

While interest in toxicology can be traced to ancient times, Paracelsus in the 16th Century established the basic tenet of toxicology: "the dose makes the poison" (Graham 1995). Much of the effort to protect human health since then has been focused on

[15] Rechard (1999) provides an excellent discussion of the historical relationship among performance assessment and other types of risk assessment techniques.

determining "safe" levels of chemical exposure[16]. The concept of "the dose makes the poison" prevailed until the mid-1970s when some believe that experience with vinyl chloride marked a fundamental change in health risk assessment (Paustenbach 2002).

The Vinyl Chloride Experience

Vinyl chloride (VC), introduced into American commerce in 1927 (Wagoner 1983) and whose primary use is in the manufacture of polyvinyl chloride (PVC), was at one time considered safe enough for human use as an anesthetic agent. However, evidence was available by 1970 that exposure to VC may result in severe human health problems including hepatic abnormalities, toxic angioneuropathy, neurologic disorders, liver degeneration, and acroosteolysis (Wagoner 1983). These findings appeared to arouse little in the way of public response because most were interpreted as representing unrealistically high doses.

The general public was first introduced to the highly carcinogenic nature of VC in 1974 when industry representatives announced finding a rare form of liver cancer (i.e., liver angiosarcoma) in three workers at the same U.S. vinyl chloride polymerization facility (Creech and Johnson 1974). Further epidemiological study of workers involved in the manufacture and polymerization of VC indicated that exposure could cause cancer at concentrations that were odorless, tasteless, and produced no discernable side effects

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[16] The U.S. Food and Drug Administration (FDA) is an agency with a long history of significant impact on defining "safe levels" of chemicals and health risk assessment. The Delaney Clause prohibiting the addition to foods of any additive or colorant that was shown to a human of animal carcinogen was added to the Federal Food, Drug, and Cosmetic Act of 1938 by amendments in 1958 and 1960. Because most data on possible carcinogenic effects are taken from animal studies, Lehman and others (Lehman and Fitzhugh 1954) at the FDA proposed the first safety factor (of 100) to be employed in relating animal to human effects (Dourson and Stara 1983; Friess 1987). The FDA was also the first agency to adopt a *de minimus* [*non curat lex*] risk level that ranged from 10^{-8} in 1973 to 10^{-6}, which was finally adopted in 1977 (Rodricks et al. 1987). In fact, the FDA was the first agency formally to use health risk assessment in the decision-making process (Rodricks et al. 1987).

(Paustenbach 2002). These results indicated that background cancer levels largely due to life-style choices could possibly mask severe health effects from even low-level exposure to chemicals in the environment and led to the development of new approaches for determining safe levels of exposure (Paustenbach 2002).

However, the concepts that “the dose makes the poison” and that a threshold or safe dose to carcinogens might exist at which no adverse human health effects were anticipated were first challenged more than a decade before vinyl chloride impacts were coming under public scrutiny (Mantel and Bryan 1961; Paustenbach 2002). Soon after Mantel and Bryan (1961) issued their landmark paper describing an approach to determine virtually safe doses (VSDs), human exposures to carcinogens were considered for regulatory purposes to have some degree of risk despite the degree of exposure and cancer formation mechanism (Crump et al. 1976; Friess 1987; Paustenbach 2002).

The regulatory notion of a "no threshold" response to carcinogenic exposure led to the widespread use of extrapolation models to estimate potential upper-bound human cancer risk (even at very low exposure levels) based upon results from animal studies. The regulatory acceptance of either “acceptable” or *de minimus* risk estimated using these “no threshold” models was a clear departure from the Delaney Clause (Paustenbach 2002). Despite continued regulatory acceptance of these models, there is no *a priori* method of determining which extrapolation model provides the best upper-bound risk estimate for the carcinogen and receptor in question; the various models normally considered can provide risk levels for low doses that vary over several orders of magnitude (Munro and Krewski 1981; Paustenbach 2002). These issues have led to increased research in the areas of low dose health effects and more accurate models including those that are biologically-based (Moolgavkar 1986; Moolgavkar and Knudson 1981). However, better extrapolation models may not truly help in the public acceptance of health risk assessment; therefore, other methods including the benchmark dose approach based on a predefined level of effect attempt to avoid the problems associated with extrapolation to low doses (Crump et al. 1995; Paustenbach 2002; USEPA 2000a). However, the process of health risk assessment must be examined in greater detail before cogent suggestions for improvements can be made.

Elements of Human Health Risk Assessment

Significant advances have been made in the four areas comprising risk assessment (Edler and Kitsos 2005; Goldstein 2005), especially in the area of evaluating exposure of humans to hazardous chemicals and the corresponding human health impacts. Each of the steps will be discussed in light of applying it to evaluation of a buried waste site and its disposition.

Hazard Identification

The purpose of this step is to determine the causal link (if one exists) between an agent and a particular health effect (where only human health effects were discussed) (NAS 1983). This purpose appears somewhat narrow from a modern risk assessment perspective. The concept underlying health risk assessment has been expanded to include not only chemicals but also radiation[17] and other stressors (e.g., physical disturbances, biological invasions, etc.) that might produce health effects. The receptors that might be

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[17] In fact some believe that "risk assessment" *per se* began not by evaluating chemicals but instead in the field of radiation control (Graham 1995).

impacted have also been expanded by some to include not only humans but any organism, population, community or ecosystem of concern (Cirone and Duncan 2000). The focus in this research, however, is on human health effects.

The hazards typically addressed in health risk assessment can be categorized (especially as they are regulated) as cancer and non-cancer effects. For non-cancer effects, the focus historically has been on identifying reference or "safe" levels of exposure to the harmful agent in question[18]. For carcinogenic effects, the prevailing U.S. regulatory position is that any exposure to a carcinogen is considered to present a finite risk to the receptor. However, there is great uncertainty when actually attempting to link a specific chemical to a particular human health impact especially at the very low levels that humans would tend to be exposed. This uncertainty may help to explain why even through there are tens of thousands of man-made chemicals in the environment, only hundreds have been shown to be animal carcinogens (often at exposure levels many orders of magnitude higher than humans would likely be exposed) and approximately 100 are known human carcinogens[19]. The difficulty in establishing links between chemicals and human health effects also likely results from other factors including types of tumors, inter-species differences, metabolism, mechanism of action, etc. (Paustenbach 2002).

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[18] The reference dose is established via modification of a "no-observed-adverse-effect-level" (NOAEL) or "lowest-observed-adverse-effect-level" (LOAEL) using a series of uncertainty and modifying factors. The benchmark dose approach is an improved alternative to this procedure for non-cancer risk assessment (Crump et al. 1995).

[19] A total of 102 known, human carcinogens (Group 1) are identified by the International Agency for Research on Cancer (IARC) available at http://monographs.iarc.fr/ENG/Classification/crthgr01.php (accessed March 13, 2008). The known animal carcinogens are deduced from examining the other classifications and their listings. Two items should be noted: 1) not all the hundreds of thousands of man-made chemicals in the environment have been tested and thus rates of incidence may be more meaningful than raw numbers, but 2) the chemicals tested first are those that may most suspect and thus likely to be found carcinogenic and thus the rates of incidence reflected by item 1 may not be generally applicable.

The concept of being precautionary in the introduction of new chemicals may seem warranted from the difficulties presented here; however, without some degree of risk tolerance, there will not only be no unforeseen hazards presented by the introduction of new chemicals—there will also be no potential benefits or innovation. This risk-benefit trade-off must be approached by assessing all risks (including those from doing nothing) while acknowledging the uncertainties in a rational manner and then deciding based on all the information available.

Dose-Response Assessment

This step involves determining the relation between the magnitude of exposure to an agent and likelihood of an adverse health effect identified during the hazard identification step (NAS 1983). This determination often takes into account factors such as sex, genetic sensitivity, and extrapolation to very low doses (often from animal studies at much higher exposures) (Paustenbach 2002). There are known differences in the development of tumors (i.e., *tumorigenesis*) between animals (e.g., mice and rates) and humans (Cunningham 2002; Trosko and Upham 2005). Human exposure to chemicals in the environment is often two to five orders of magnitude lower than the minimum dose tested in the corresponding animal study used to establish whether or not the chemical is a carcinogen (Paustenbach 2002), which requires extrapolation to low doses and resulting in large uncertainties in the dose-response relationship. This last factor is often that which introduces the largest uncertainties into the dose-response assessment.

Despite known uncertainties, regulatory preference has been to define single or point values to represent either a safe exposure level (e.g., acceptable daily intake, reference dose, etc.) for non-carcinogens or risk per unit exposure (e.g., slope factor) for carcinogens[20]. These point values are often derived from an assessment of the quality and uncertainties in the data and methods. However, formal probabilistic techniques have not been introduced into the development of these point values (Edler and Kitsos 2005; Richardson and Burmaster 1996), and their use often leads to upper-bound (i.e., not expected values or "true") estimates of risk, with an unknown degree of "conservatism" (i.e., by how much it may or may not overstate the expected or "true" risk).

Therefore, even if one can measure or predict the exact level of exposure of a receptor to an agent known to have heath impacts, the expected or "true" risk cannot be obtained from the current point values and methods. It is hoped that the risk values obtained can be shown to overstate the expected or "true" risk. Some attempts have been made to represent reference values and slope or potency factors using probabilistic distributions and techniques (Baird et al. 1996; Crouch 1996; Richardson and Burmaster 1996); however, there has been no indication that regulatory agencies are going to adopt these concepts. The dose-response information used in this research is taken from the most recent versions of the appropriate Federal Guidance Reports (Eckerman et al. 1999; Eckerman and Ryman 1993; Eckerman et al. 1988), EPA Integrated Risk Information System (IRIS) database (USEPA 2006), EPA Health Effects Assessment Summary Tables (HEAST) (USEPA 2001b), International Commission on Radiological Protection (ICRP)[21] or dose-response information of similar or better pedigree.

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[20] The U.S. Nuclear Regulatory Commission (NRC), whose purpose is to regulate civilian use of nuclear materials, regulates based upon dose instead of risk. The U.S. Environmental Protection Agency (EPA) and NRC, which often both have responsibility at contaminated sites, are working to find common ground on this issue of regulation (NRC/EPA 2002).

[21] The International Commission on Radiological Protection (ICRP) website can be found at http://www.icrp.org (accessed March 6, 2008).

Exposure Assessment

Once an agent is known (or suspected) to produce an adverse human health effect and the dose-response relationship (represented by a point value) has been established, then the degree of human exposure before and after remedial action is taken must be estimated (NAS 1983). Without exposure to the agent, there is no risk from the agent. Considerable attention has been paid and significant improvements have been made since the *Red Book* was issued in the areas of monitoring and predicting human exposures using mathematical and computer-based models (Edler and Kitsos 2005; Goldstein 2005; Paustenbach 2002). These advances appear to be the logical result of responding to the concerns related to conservative exposure and risk estimates resulting from compounding conservative assumptions[22] from studies performed since the 1970s (Paustenbach 2002).

The results from the exposure assessment should include the magnitude, duration, and frequency of human exposure to the agent of concern (Paustenbach 2002). In its most comprehensive form, the exposure assessment should also provide the route(s) of exposure and characterize the population exposed (i.e., size, nature, and classes including sensitive individuals) and address all significant uncertainties in the exposure estimates (NAS 1983). Because of the highly uncertain nature of buried waste sites[23] and the fact that potential remedial actions must be evaluated, the application of conceptual and mathematical models (for release, fate and transport, uptake, etc.) is critical to assessing the potential for exposure of and risk to receptors to contaminants over the long time

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[22] The literature concerning the deleterious effects of compounding conservative assumptions in risk assessment is copious (Bogen 1994; Burmaster and Harris 1993; Cullen 1994; Finkel 1989; Maxim 1989; McClellan 2003; Nichols and Zeckhauser 1988).

[23] Wastes have been buried at many of the larger DOE sites since the 1940s and 1950s with little in the way of detailed records of what was buried and where. Furthermore, even if detailed internment records are available, many of the contaminants have likely migrated since burial.

periods that must be considered. The uncertainties associated with the use of such models must be addressed in an open and transparent fashion for the results to be trustworthy[24].

The single agent-pathway-receptor concept that prevailed in the *Red Book* for describing source to receptor pathways has been supplanted by further considerations of how to estimate risks for aggregate and cumulative exposures[25] to chemicals, radiation, and other stressors (Goldstein 2005; USEPA 2003). Thus the applicability and complexity of the risk assessment process has been increasing since the *Red Book* was published. Responding to these changes in the basic fabric of the risk assessment process in an understandable and transparent manner is a significant challenge to the first three steps (i.e., hazard identification, dose-response assessment, and exposure assessment) in the health risk assessment process; however, it may be most challenging to final step, risk characterization.

Risk Characterization

The primary purpose of this step is to combine the exposure and dose-response assessment information for identified hazardous agents to describe the nature and magnitude of *human* risk including a discussion of uncertainty (NAS 1983). This step may be the most critical because this is where and when the scientific results of the health risk assessment process are ultimately translated into a final form that must be understandable to those who must manage the risks identified and the stakeholders to

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[24] As Richard L. Postles, formerly of E. I. DuPont, attributed to G. E. P. Box of Wisconsin University, "No model is correct, but some are useful."

[25] Aggregate risk assessment involves the evaluation of risks resulting from a single substance via multiple pathways (Goldstein 2005; USEPA 2001a). Cumulative risk assessment is used to evaluate simultaneous exposures to multiple contaminants via multiple routes (Goldstein 2005; USEPA 2003).

whom they answer. However, the fact that the risk information must be described in a consistent and transparent manner means that the impact of characterizing risk must be considered throughout any risk assessment process and that the entire risk assessment process must be open for the results to be transparent (NAS 1996). These issues have lead to the idea that risk managers and stakeholders be kept "in the loop' throughout the process (NAS 1996; P/CCRARM 1997a).

To some, health risk assessment is synonymous with "quantitative [health] risk assessment"[26] and necessarily involves *quantification* of health risks during the characterization process. However, the broader context originally suggested in the *Red Book* includes *qualitative* evaluations of human health risk (NAS 1983). Despite the qualitative-quantitative context of the risk assessment process, certain fundamental information must be provided with the risk results including characterizing the number and nature of potentially exposed receptors (including temporal variations in exposure and risk) and uncertainties.

The level of detail applied to the analysis of risk should be commensurate with the nature of the contaminated site and the importance of the decision to be made. Often it is appropriate to begin with a qualitative analysis of available site information to determine the nature of the risks presented by the contaminants and their potential migration from the site. This qualitative assessment can provide the foundation for the quantitative analyses to follow and, perhaps more significantly, a convenient approach for introducing the problem and its scope to risk managers, regulators, and other stakeholders while reducing the potential for getting ensnared in technical and esoteric details concerning the

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[26] For an example of such a sentiment, please refer to Paustenbach (2002), p. 42.

dose-response and exposure portions of the risk assessment. The basic approach to site evaluation is outlined at the beginning of the process with additional communications as the assessment process unfolds to allow for transparency to interested parties.

Selected Hazard and Safety Assessment Techniques

While many changes have been proposed to the health risk assessment process, lessons learned and potential changes from other types of risk assessment practices should also be examined and factored into the process. Historically, both hazards and risks have been evaluated using a wide variety of techniques. In essence, different techniques have different targets, objectives, and advantages and thus constitute tools in the risk assessor's toolbox that may be used to evaluate hazard and risk. Some techniques may be used to identify significant hazards while others link the identification of hazards with consequences. Some approaches address all elements of the risk-triplet including hazard identification, event probability, and consequences while others still also address issues of who may be at risk and when. Some techniques provide only qualitative information concerning hazards and/or consequences, while others provide quantitative or semiquantitative risk information.

Process Hazard Analysis Techniques

The first set of techniques considered here are those that can be broadly classified as Process Hazard Analysis (PHA) techniques, which form the foundation of the Occupational Safety and Health Administration (OSHA) standard for process safety management of highly hazardous chemicals (Bahr 1997; USDOL 1997a). The techniques mandated in the OSHA standard (USDOL 1997a) include

- Expert or "what if" analysis (Bahr 1997; Nolan 1994; Rasche 2001)
- Hazard and operability (HAZOP) studies (Bahr 1997; Nolan 1994; Rasche 2001)
- Failure modes and effects analysis/Failure modes, effects, and criticality analysis (FMEA/FMECA) (Bahr 1997; Rasche 2001)
- Fault tree analysis (FTA) (Bahr 1997; Louvar and Louvar 1997; Rasche 2001)

The above techniques are primarily safety analysis techniques and provide a subset of the information needed for a risk assessment. A brief discussion is provided below for each of the PHA techniques in light of their potential use in the health risk assessment process. Table 1 provides a summary of the safety analysis techniques identified in this research.

Expert or "What-If" Analysis. This technique can be useful during a screening assessment phase to consider (albeit qualitatively) the consequences of unexpected events on a system by asking "*What* are the consequences *if* this event occurs…?" (Bahr 1997). The analysis can be performed quickly and inexpensively and can be integrated with other techniques to provide the qualitative information needed to evaluate hazards and risks. However, the technique is inadequate for the analysis of highly complex systems, such as DOE buried waste sites, except in the initial or screening assessment phase.

HAZard and OPerability (HAZOP) Study. A HAZOP study is a systematic approach taken by a group (typically of engineers) to identify hazards based upon deviations from expected operation and inefficiencies in a system (Bahr 1997). The primary use of this technique is to focus on measures (engineering or administrative) to mitigate risks that are identified and considered unacceptable. This technique is thorough and can be good for complex, well-defined systems such as chemical process analysis. Typically, the detailed level of knowledge required for a *HAZOP* analysis would preclude it from being recommended as a reasonable, general technique for the screening assessment for a buried waste site. On the other hand, if the results of *HAZOP* analyses are available, the results would certainly be of considerable use in all assessment phases.

Failure Modes and Effects Analysis (FMEA)/Failure Modes, Effects, and Criticality Analysis (FMECA). *FMEA* identifies the ways that a component in a system can fail and corresponding effects (of component failure) on the system (Bahr 1997). *FMECA* takes *FMEA* a step further in that it identifies the criticality of the component that might fail. These tools were originally developed as reliability tools (i.e., not as safety analysis tools *per se*) but have been applied to safety analysis as recognized by OSHA (Bahr 1997; USDOL 1997a). The results from either a *FMEA* or *FMECA* can provide some useful information concerning failure modes and consequences but will not provide all the information necessary for a screening risk assessment.[27] Existing *FMEA*/*FMECA* results could be useful during the screening assessment phase.

[27] For example, there may be significant *hazards* even if there are no failures identified at the time.

Table 1. Process Hazard Analysis (PHA) Techniques —Basic Information (Holdren et al. 2006)

Risk Analysis Technique	Primary Industries and Applications	Hazards Considered	Potential Receptors	Types of Consequences	Time Frames Considered	Risk Expression	Uncertainties Considered	References
Expert and “what if” analyses	All industries & government for all risk assessment phases	All types including substances, conditions, or practices	Human & environmental	All types of consequences including ultimate	All time frames	Qualitative	Not taken into account	(Bahr 1997; Rasche 2001)
Hazard & operability (HAZOP) studies	All industries & government but primarily chemical & processing to identify hazards & assess control during design, operations, & disposal	Substances & conditions arising from deviations from design specifications, practices deviating from procedures	Often facility, can extend to humans	Process-related (e.g., releases) & ultimate effect (e.g., traumatic injury, illness, death, etc.)	No time dependencies	Qualitative	Not taken into account	(Bahr 1997; Rasche 2001)
Failure modes & effects/criticality analysis (FMEA/FMECA)	All industries & government but primarily aerospace & nuclear for reliability of single components & accident analysis after design is completed	Conditions related to component failures	Other components & system	Impacts to other components & system	No time dependencies	Qualitative/ Quantitative	Not taken into account	(Bahr 1997; Rasche 2001)
Fault tree analysis (FTA)	Conditions & practices deemed significant	Conditions & practices deemed significant	Implicit in hazard def. including both workers & general public	Implicit in hazard definition often process related	No time dependencies	Qualitative/ Quantitative	Input parameters, scenario variations	(Bahr 1997; Rasche 2001)

Fault Tree Analysis (FTA). *FTA* is a systematic, graphical, top-down approach in which a significant fault is postulated and then all the faults or events (and logical ordering) are defined that must occur for the postulated fault to occur. This technique is very thorough (for known faults) and provides both a useful, visual system model and probability estimates for faults (Rasche 2001). However, this technique relies upon the correct definitions of failures, mechanisms, and interactions, and this detailed information may not be available for the initial assessment stage. Thus a *FTA* would not be recommended during the screening assessment phase for a buried waste site; however, any existing *FTA* results should be taken into consideration during all phases of risk assessment.

Other Safety Analysis Techniques

Additional safety analysis techniques have been identified that have been used to supply some of the information required for the health risk assessment process. These techniques include

- Human factors analysis (HFA) (Bahr 1997; Rasche 2001)
- Event tree analysis (ETA) (Bahr 1997; Rasche 2001)
- Cause-consequence (or expanded event tree) analysis (CCA) (Bahr 1997)
- Dispersion (or consequence) analysis (Bahr 1997; Louvar and Louvar 1997)

As with the aforementioned process hazard analysis techniques, the above approaches are primarily safety analysis techniques and may only provide a subset of the information needed for a risk assessment. A brief discussion is provided below for each of these safety analysis techniques as it relates to the assessment of a buried waste site.

Human Factors Analysis. *HFA* is a thorough, qualitative or quantitative technique used to identify and mitigate or correct human errors that could lead to significant hazards and/or consequences (Bahr 1997). Because system failures are most often attributable to human error, the impact of human error on the disposition of buried wastes must be considered in all phases of the risk assessment process. However, quantitative *HFA* results can be misleading (unless pertinent data are available), and it is often preferable to include the pertinent human errors and their impacts as part of the *expert analysis* described above (especially during the screening risk assessment phase).

Event Tree Analysis (ETA). *ETA* is a graphical, bottom-up approach to model dependencies and escalation of catastrophic events. This technique is very thorough; however, *ETA* is highly dependent on the correct definitions of failures, mechanisms, and interactions. *ETA* is not recommended during screening risk assessments for buried waste sites; however, any existing *ETA* results should be taken into consideration.

Cause-Consequence (or Expanded Event Tree) Analysis. *CCA* is a blend of fault and event tree analyses used to identify chains of events that lead to undesired consequences (Bahr 1997). Although *CCA* is a good technique to model dependencies and temporal escalation of events, *CCA* relies upon often scarce information and can only be applied to one (initiating) event at a time. This technique is not recommended for use in the risk assessment of DOE buried wastes. The impracticality of employing sophisticated techniques such as fault and event tree analysis and *CCA* to complicated, geologic systems such as WIPP has been previously acknowledged (Rechard 1999).

Dispersion Analysis. Dispersion analysis uses well-established regulatory models and tools[28] to provide quantitative estimates of exposure due to the dispersion of chemicals typically via the air and water pathways (Bahr 1997; Louvar and Louvar 1997).

Summaries of the process hazard and safety analysis techniques are provided in Table 1 and Table 2, respectively. The strengths and weaknesses of these techniques are summarized in Table 3. Of the aforementioned process hazards and safety analysis techniques, *expert analysis* and the results of historic hazard and safety studies are considered the best types of input to the screening risk assessment phase.

For DOE sites, it is likely that a number of safety analyses would have been performed as standard practice. It is unlikely, however, that costly and time consuming techniques such as fault or event tree or cause-consequence analyses would have been performed or would be required for a buried waste site—it is more likely that early remedial actions would be taken to address specific hazards or that fault-tolerant remedial actions would instead be selected. More detailed health risk assessments that would be required after screening analysis should be based on human health risk assessment techniques and use site-specific information wherever possible (McClellan 2003).

[28] For example, Areal Locations of Hazardous Atmospheres (ALOHA) and Computer-Aided Management of Emergency Operations (CAMEO) are two systems that have extensive use for planning and responding to chemical emergencies. Refer to http://www.epa.gov/emergencies/content/cameo/index.htm (accessed March 13, 2008) for additional information on these systems.

Table 2. Safety Hazard Analysis Techniques —Basic Information

Risk Analysis Technique	Primary Industries and Applications	Hazards Considered	Potential Receptors	Types of Consequences	Time Frames Considered	Risk Expression	Uncertainties Considered	References
Human factors analysis	All industries but primarily those where humans play a major part (nuclear & aviation) during all phases but esp. during operations	"Safety critical" conditions & practices resulting from human error	Generally workers & general public	Injuries or death of humans often from acute effects	No time dependencies	Qualitative/ Quantitative	Not taken into account	(Bahr 1997; Rasche 2001)
Event tree analysis	Bottom-up approach for all industries & government but primarily nuclear & process to model catastrophic events & emergency response for all phases	Conditions & practices deemed important	Implicit in hazard def. including both workers & general public	Implicit in hazard definition often process related	Explicit consideration	Qualitative/ Quantitative	Scenario variations	(Rasche 2001)
Cause-consequence (or expanded event tree) analysis	Blend of fault/event tree approaches for all industries & government but primarily nuclear & process to identify chains of events leading to undesirable consequences	Conditions & practices deemed important	Implicit in hazard def. including both workers & general public	Implicit in hazard definition often process related	Explicit consideration	Qualitative/ Quantitative	Scenario variations	(Bahr 1997)
Dispersion (or consequence) analysis	All industries & government but primarily chemical & process to model impacts from dispersion of chemicals via water or air often during operations	Chemicals, gases, and vapors that can be dispersed & resulting conditions	Human (both workers & general public) & the environment	Traumatic injuries, fatalities, & environmental damage	A single event is modeled, acute & chronic effects can be considered	Quantitative	Not taken into account	(Bahr 1997; Louvar and Louvar 1997)

Table 3. Comparison of Hazard Analysis Techniques—Strengths and Weaknesses [Modified after (Bahr 1997; Rasche 2001; Rouvroye and Brombacher 1999)]

Risk Analysis Technique	**Primary Industries and Applications**	**Strengths**	**Weaknesses**	**References**
Expert and "what if" analyses	All industries & government for all risk assessment phases	Inexpensive, fast implementation, can use little data or information, can be easily integrated with other techniques	Not rigorous or adequate for highly complex systems, cannot identify dependencies	(Bahr 1997; Rasche 2001)
Hazard & operability (HAZOP) studies	All industries & government but primarily chemical & process to identify hazards & assess control during design, operations, & disposal	Thorough technique, evaluates existing safeguards, good for complex systems & identifying deficiencies	Requires committed team & well-defined system	(Bahr 1997; Rasche 2001)
Failure modes & effects/criticality analysis (FMEA/FMECA)	All industries & government but primarily aerospace & nuclear for reliability of single components & accident analysis after design is completed	Thorough technique, useful for single point failures	Failure/reliability not safety tool, performed late in design phase, no consideration of human error	(Bahr 1997; Rasche 2001)
Fault tree analysis	Top-down approach for all industries & government but primarily aerospace & nuclear to identify events leading to a defined hazard after requirements defined	Thorough technique, good for complex systems & accident investigation, identify redundancies and fault tolerance, provides visual system model	Does not model all possible faults, relies on correct fault, failure mechanisms, & interactions, data often unavailable, unable to model temporal events	(Bahr 1997; Rasche 2001)
Human factors analysis	All industries but primarily those where humans play a major part (nuclear & aviation) during all phases but especially during operations	Thorough technique identifying human errors and ways to mitigate, good at evaluating safety of procedures	Quantification often misleading, very difficult to model human behavior, data difficult to gather, data unavailable for specific purpose, can be costly	(Bahr 1997; Rasche 2001)
Event tree analysis	Bottom-up approach for all industries & government but primarily nuclear & process to model catastrophic events & emergency response for all phases	Good technique to model dependencies & temporal escalation of events, can use used to model catastrophic risk, can be supplemented by other techniques	Needed data often scarce, relies on correct event escalation, only one event can be examined at a time	(Rasche 2001)
Cause-consequence (or expanded event tree) analysis	Blend of fault/event tree approaches for all industries & government but primarily nuclear & process to identify chains of events leading to undesirable consequences	Good technique to model dependencies & temporal escalation of events, multiple outcomes analyzed, end events do not need to be know ahead of time	Needed data often scarce, relies on assuming correct event escalation, only one event can be examined at a time	(Bahr 1997)
Dispersion analysis	All industries & government but primarily chemical & process to model impacts from dispersion of chemicals via water or air often during operations	Quantitative estimates provided based upon well-accepted models & programs (e.g., CAMEO), only have to examine likely worst-case scenarios, fits nicely into other safety & risk analyses	Often based upon worst-case scenarios, quantitative estimates can be misleading, standard programs often do not allow for uncertainty analysis	(Bahr 1997; Louvar and Louvar 1997)

Other Relevant Risk Analysis Techniques

Health risk analysis techniques represent a small fraction of those available to analyze and manage risks. Techniques have been developed to evaluate risks for financial institutions (Alexander 2005), pest management (PMRA 2000), medical devices (Eisner 2000), prescription drugs (FDA 1999), marine transport (IMO 2002a; b; USCG 1996a; b; c), emergency response (DCDEP 2000), and industrial machine development (Anderson 2004). Two additional approaches will be described to illustrate the breadth of available risk analysis techniques and the various targets they address and goals they possess.

One approach, *ecological risk assessment* (ERA), is used to evaluate impacts on ecological receptors from exposure to stressors in much the same way that *human health risk assessment* is employed to evaluate adverse effects on human receptors (USEPA 1992a; 1997a; 1998a). ERA techniques have generally been based on those for human health risk assessment; however, changes have been made to account for the additional complexity of ecological systems, sensitivities of ecological receptors, diversity in stressors, and difficulties in defining meaningful endpoints (e.g., death, lack of diversity, etc.) (Suter 1999; USN 1999). Relevant information (e.g., management of sparse data, integration of carcinogenic and non-carcinogenic effects, etc.) from ERA techniques should be integrated into the human health risk analysis.

Another general approach is *life-cycle assessment* (LCA). LCA is a "cradle-to-grave" approach that evaluates the environmental impacts of a product, service, or activity in a holistic fashion from its origin to its end. There has been a recent focus on integrating LCA and risk assessment techniques (Cowell et al. 2002; Evans et al. 2002; Nishioka et al. 2002; Ozawa 2002; Tukker 2002); however, this integration is focused on improving LCA techniques and their acceptance—not providing an integrated framework that produces better decisions. A common approach to integrating risk and LCA techniques is by including toxicological parameters and models in LCA impact assessments (Flemström et al. 2004) instead of integration into a single framework. Sanne and Widheden (2005) suggest that integrating these approaches into a single framework may not be worthwhile because of the time-consuming nature of both and the very different perspectives on which they are focused. The approach of adding risk-based indicators to an LCA framework is not sufficient for the purpose of evaluating buried waste sites; however, the life-cycle and holistic perspectives of the LCA analysis are important considerations to a risk-informed decision-making process.

Occupational Hazards and Life-Cycle Considerations in Health Risk Assessment

Despite significant advances in risk assessment techniques, there is a conspicuous absence of the consideration of *standard industrial* risks[29] in many health risk assessment approaches. Many studies indicate that the predominant source of risk in site cleanup is industrial or occupational in nature and not from the exposure risks that are more typically and comprehensively evaluated (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). Most complicated site cleanups, especially those involving excavation and retrieval of hazardous and radioactive wastes, resemble heavy construction sites and the primary risk drivers are the same. If wastes must be transported off-site over long distances, then transportation accidents, without radionuclide or hazardous chemical releases, may be a significant or dominant risk component.

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[29] Standard industrial risks are *non-exposure* risks associated with falls, explosions, transportation accidents, etc.

Thus, even if occupational risks from cleanup activities are judged differently than involuntary exposure risks to the general public, occupational risks should at least be considered when selecting remedial alternatives. Furthermore, consideration of *life-cycle* risks is often missing from those approaches calling for the explicit consideration of short-term risks associated with remedial activities (USEPA 1991b). For example, the DOE Health Risk Evaluation Methodology (Blaylock et al. 1995) can be used to assess both exposure and standard industrial accident risks to workers performing environmental restoration and waste management activities at DOE sites but does not address the concomitant risks to the general public.

Numerous methodologies and tools exist that consider various aspects of the risk assessment process; however, few integrate—even in a screening sense—all aspects necessary to address the life-cycle risks posed by the disposition of buried wastes. One exception is the INEEL Environment, Safety, and Health Risk Assessment Program (ESHRAP) (Eide and Nitschke 2002; Eide et al. 2002; Eide and Wierman 2003) that integrates both the exposure and standard industrial risks associated with the ultimate disposition of wastes[30]. ESHRAP can be used to estimate worker and general public risks from exposures to both radioactive and hazardous chemicals as well as standard industrial risks from activities associated with cleanup and management activities over the entire waste management program (Eide and Wierman 2003). The point estimate risks obtained using ESHRAP are meant to be "best estimate" rather than bounding or conservative. However, ESHRAP contains neither probabilistic nor sensitivity analysis facilities and thus provides no ability to assess the uncertainties associated with nor defend the risk

[30] ESHRAP was formerly called the Simplified Risk Model (SRM) (Eide et al. 1998; Eide et al. 1996; Peatross and Eide 1996). Exposure risks are estimated using the methodology developed for the disposal of DOE mixed low-level wastes (Waters et al. 1996a; b; c).

estimates provided. Despite the existence of ESHRAP and other tools, a screening risk analysis tool is needed that can provide *defensible* life-cycle risk estimates for both baseline and remedial options as one input to the risk-informed decision-making process.

Probabilistic Risk Assessment and Performance Assessment

Risk assessment techniques have been developed either to evaluate human health effects due to chemicals in the environment (i.e., health risk assessment) or to evaluate low probability and high consequence events and related safety concerns for nuclear reactors (i.e., probabilistic risk assessment (Rechard 1999)). A more recent addition to the risk assessment landscape, *performance assessment* (Ewing et al. 1999; Rechard 1999), can be seen as entrenched in both probabilistic risk assessment (i.e., concerning the failure of engineered systems) and health risk assessment (i.e., transport of contaminants from failed engineered systems and concomitant human health effects)[31].

Various definitions for probabilistic risk assessment and performance assessment can be found in the literature (Ewing et al. 1999; NAS 2005; Rechard 1999); however, definitions consistent with those in the 2005 NAS *Risk and Decisions* Report (NAS 2005) and Rechard (1999) will be employed here. *Probabilistic risk assessment* (PRA), originally developed to help regulate risks found in the nuclear industry (Keller and Modarres 2005; Rechard 1999), is a systematic approach to transforming potential failures into risk profiles (as suggested by Kaplan and Garrick (1981)) taking explicit account of uncertainties to evaluate reliability, availability, and accident scenarios (NAS 2005). Results of the PRA indicate the probability and magnitude of each risk. A

[31] Another more recent technique, *vulnerability analysis*, is used to identify receptors facing the greatest losses and damage, the sources of vulnerability, and how it can be ameliorated or eliminated (Turner II et al. 2003a; Turner II et al. 2003b).

performance assessment (PA) is the application of risk assessment techniques to an engineered system and comparison of results to standards (NAS 2005; Rechard 1999).

PRA and PA techniques have provided very useful information to both regulators and the general public; however, both approaches rely on identifying all significant failures; can be costly and time consuming; and are based upon probabilistic information that is often difficult to obtain, sparse, or unavailable. Furthermore, probabilistic techniques are not appropriate for every situation. For example, when screening evaluations based on assumptions known to provide higher that expected risks indicate that the resulting risks will be acceptable or when remediation costs are very low and action can be taken without prior risk assessment (USEPA 1997c). The sophistication of the assessment (including the treatment of uncertainty) should be commensurate with the magnitude and complexity presented by the contaminated site (USN 2001).

The "Risk Triplet" as a General Framework for Risk Assessment

Perhaps the most technically mature of the risk assessment techniques is *probabilistic risk assessment* (PRA). The risk-triplet concept suggested by Kaplan and Garrick (1981) provides a useful conceptual framework for PRA (Garrick 2007). The risk triplet links scenarios describing what can go wrong to consequences via likelihoods providing a degree of consistency between the concepts of “risk” and “risk assessment.”

When quantitative risk assessment techniques are applied to engineered systems and the resulting results are compared to standards, the analysis is denoted a *performance assessment* (NAS 2005; Rechard 1999). The performance assessment may or may not be probabilistic in nature (Rechard 1999) and thus may be seen as a special case of application of the risk-triplet. Performance assessment concepts are important to this research because buried wastes will ultimately either be managed in-place or retrieved for disposal elsewhere—both the original and final sites may require the application of performance assessment techniques.

The *Red Book* description of health risk assessment can also be seen as a special case of the risk-triplet applied to human health concerns involving exposure to toxic chemicals and radiation. The scenarios for chemical exposure risks that were of primary concern in the *Red Book* can be described using three health risk assessment elements (i.e., hazard identification, dose-response assessment, and exposure assessment). Consequences are typically addressed during the risk characterization phase.

The fundamental difference between probabilistic risk assessment and typical health risk assessment approaches is in the manner uncertainties are addressed. Whereas no mandate exists for human health risk assessments to more than address uncertainties qualitatively, inclusion of uncertainty is an integral part of using PRA to assess risks. One important contribution of PRA is that not only the adverse consequences of events are considered—the likelihood of the event actually occurring is an integral part of the risk conceptualization. This concept of probability linking adverse events and consequences appears lost in the health risk assessment field especially in how likely receptors are to being exposed to harmful agents.

Another important contribution from the risk-triplet approach is that no point value from the risk curve represents the risk—the risk curve itself describes the risk for decision-making purposes (Kaplan and Garrick 1981). The situation for human health risk assessment is even more difficult because not only uncertainty but also variations in exposures and differences in effects for varying receptors over time must be considered.

Despite these differences, much can be learned from the PRA and performance assessment fields when performing health risk assessments.

Selected Human Health Risk Assessment Techniques

Specific health risk assessment approaches have been developed since the process was formalized in the *Red Book* (NAS 1983). A few representative approaches will be discussed to illustrate the breadth and depth of the existing approaches and distinguish among the many flavors of *risk assessment*. The approaches that will be discussed are

- Radiation risk assessment (HealthCanada 1998; ITRC 2002; Louvar and Louvar 1997)
- Carcinogen risk assessment (USEPA 1986; 2005)
- Non-carcinogen risk assessment (Louvar and Louvar 1997)

Because many contaminated sites contain both hazardous and radioactive contaminants that can migrate to potential receptors via multiple pathways, aggregate and cumulative exposures to toxic agents and their corresponding effects must be considered. A more recent addition to the health risk assessment field, probabilistic health risk assessment, which extends deterministic health risk assessment techniques based upon point risk estimates with those taken from PRA to provide a more complete picture of the risks posed to receptors will be discussed. Finally, because the ultimate goal is to decide on which remedial actions to take, comparison of the risk estimates obtained from these various techniques (in addition to other non-risk factors) is important to the remedial decision-making process.

Radiation Health Risk Assessment

One of the first health risk assessment techniques was developed to analyze potential human impacts from exposure to a specific type of hazard—ionizing radiation[32]. This technique is well-established and much of it is based on long-term studies of individuals exposed to reasonably well-known (and often large) doses of ionizing radiation (Eckerman et al. 1999; HealthCanada 1998; ITRC 2002). In the United States, there are two basic metrics for expressing the potential impacts from exposure to ionizing radiation: dose and risk (ITRC 2002).

The NRC uses dose to directly express risk whereas the EPA relies upon converting dose or exposure to an upper-bound estimate of risk using dose-to-effect factors. This difference in risk expression has led to confusion in both regulatory and public circles (NRC/EPA 2002), especially in that "risks" obtained from the EPA method may be confused for "true" or expected risks. Because of large uncertainties in the dose-to-effect factors mandated by EPA, any impacts from exposures to radioactive wastes should be expressed as both doses and corresponding risks for transparency.

Carcinogen and Non-carcinogen Health Risk Assessment

Other general assessment approaches are often based on the effects (i.e., carcinogenic or non-carcinogenic) when humans are exposed to toxic and hazardous chemicals.[33] The approach for carcinogens is similar to that taken by the EPA for radiation effects (i.e., dose is converted to risk using dose-to-risk factors denoted *slope*

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[32] It has been long hypothesized that any exposure to ionizing radiation can cause genetic effects (Crump et al. 1976; Friess 1987; Paustenbach 2002). This concept has been adopted as the basis for regulation.

[33] Carcinogenic risk assessment (i.e., concerning carcinogenic effects from chemical exposure) is distinguished from radiation risk assessment for human exposure to ionizing radiation.

factors that are defined by regulatory fiat). A different approach is taken for non-carcinogenic effects to chronic toxic chemical exposure whereby a reference dose is defined that represents a "no effect" dose[34] and then a hazard index is computed as the ratio of the estimated dose to the reference dose (i.e., dose or risk is not quantified).

The practice of quantifying (often bounding) carcinogenic risks but not non-carcinogenic risks may have led to the overemphasis of cancer risks over other types of health risks (Crump 2003; Graham 1995). Uncertainties are considered very differently in the approaches. For carcinogenic effects, uncertainties are considered in the exposure analysis; however, slope factors are considered *fixed* (despite large uncertainties), which can lead to confusing *computed* risk estimates, which should not be confused with "true" or expected risks. On the other hand, the reference dose is computed using uncertainty/variability factors[35] based on expert judgment concerning the quality and quantity of information. Because many DOE buried waste sites contain not only radioactive wastes but also toxic and hazardous chemicals, all such potential impacts on human receptors should be considered. Summaries of the characteristics and strengths and weaknesses of the hazard analysis and health risk assessment techniques described in this chapter are provided in Table 3 through Table 5.

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[34] The reference dose has been established via modification of a "no-observed-adverse-effect-level" (NOAEL) or "lowest-observed-adverse-effect-level" (LOAEL) using uncertainty and modifying factors. The benchmark dose approach appears to be an improved alternative to this procedure for non-cancer risk assessment (Crump et al. 1995).

[35] So-called modifying factors, which were intended to represent scientific uncertainties in the study or database not captured elsewhere (i.e., by uncertainty/variability factors), were phased out in 2004. Refer to http://www.epa.gov/ncea/iris/help_gloss.htm#u (accessed March 14, 2008) for additional information on the five types of uncertainty/variability factors used in the EPA Integrated Risk Information System (IRIS) database (USEPA 2006).

Table 4. Selected Human Health Risk Assessment Techniques —Basic Information

Risk Analysis Technique	Primary Industries and Applications	Hazards Considered	Potential Receptors	Types of Consequences	Time Frames Considered	Risk Expression	Uncertainties Considered	References
Radiation risk assessment	All industries & government but especially in EPA, DOE, & NRC to identify risks associated with exposure to ionizing radiation	Radioactive substances including internal & external exposures	Human (both workers & general public)	Genetic effects, tumors & cancer-related fatalities	Long-term & latent effects of exposure to radiation	Quantitative	Parameter, model, scenario uncertainties, variations should be acknowledged in characterization	(Louvar and Louvar 1997)
Carcinogen risk assessment	All industries & government but especially FDA, EPA, & ATSDR to identify risks associated with exposure to carcinogens	Carcinogenic substances	Human (both workers & general public)	Genetic effects, tumors & cancer-related fatalities	Long-term & latent effects of exposure to toxic chemicals	Quantitative	Parameter, model, scenario uncertainties, variations should be acknowledged in characterization	(USEPA 1986; 2005)
Non-carcinogen risk assessment	All industries & government but especially in EPA to identify risks associated with exposure to toxic chemicals	Toxic substances	Human (both workers & general public)	Illnesses & fatalities	Acute & chronic effects of exposure to toxic chemicals	Semi-quantitative (e.g., hazard indices)	No agreed upon method, but significant uncertainties should be acknowledged	(Louvar and Louvar 1997)

Table 5. Selected Health Risk Assessment Techniques—Strengths and Weaknesses [Modified after (Bahr 1997; Rasche 2001; Rouvroye and Brombacher 1999)]

Risk Analysis Technique	Primary Industries and Applications	Strengths	Weaknesses	References
Radiation risk assessment	All industries and government but especially in EPA & DOE to identify risks associated with exposure to ionizing radiation	Well defined technique with long history of application to various types of contaminated sites	Quantitative estimates of risk often mistaken for estimates of actual risk, bases for slope factors, large uncertainties in models & slope factors, choice of receptor	(Louvar and Louvar 1997)
Carcinogenic risk assessment	All industries and government but especially in FDA & EPA to identify risks associated with exposure to carcinogens	Well defined technique with long history of application to various types of contaminated sites	Quantitative estimates of risk often mistaken for estimates of actual risk, bases for slope factors, large uncertainties in models & slope factors, choice of receptor	(USEPA 1986; 2005)
Non-carcinogenic risk assessment	All industries and government but especially in EPA to identify risks associated with exposure to toxic chemicals	Well defined technique with long history of application to various types of contaminated sites	Bases for reference factors (e.g., RfD), large uncertainties in models & reference factors, choice of receptor	(Louvar and Louvar 1997)

Aggregate and Cumulative Risk Assessment Approaches

As knowledge has increased and better models have been developed, a number of new and more sophisticated human health risk assessment techniques have emerged. For example, aggregate risk assessment is the process of evaluating risks resulting from a single chemical by multiple pathways and exposure routes (USEPA 2001a). Cumulative risk assessment extends the assessment by evaluating the cumulative effects of multiple and simultaneous chemical exposures (via multiple pathways and routes of exposure) (ILSI 1999; USEPA 2003). For any complicated buried waste site, there are likely to be multiple contaminants of potential concern that impact various receptors via multiple pathways simultaneously. The framework and methodology considers cumulative risks in terms of how risks from different sources via different pathways over varying time frames are combined; however, a detailed cumulative analysis including synergistic and antagonistic dose-to-effect analyses is outside the scope of this research.

Comparative Risk Assessment

Comparative risk assessment is the process of comparing estimates of risks to characterize environmental profiles and priorities on site-wide, regional, and/or national level (Andrews et al. 2002; Boutin et al. 1998; Bridges et al. 2004; Hofstetter et al. 2002; Jones and Klein 1999; Morgenstern et al. 2000; USEPA 1990). Many attempts have been to allocate resources and prioritize remedial efforts based on comparisons of estimated risks (Finkel 1994; Finkel and Golding 1994; Habicht 1994; Jones and Klein 1999; USEPA 1987; 1990). Most of these attempts at rank-ordering based on risk have met with both public and scientific disdain or outright protest (Ashford 2002; Commoner 1994; Geisinger 2001; Hornstein 1992; O'Brien 1994; Slovic 2003). It has become generally accepted that risks should be one input (with social values, costs, etc.) to the *risk-informed* (*vice risk-based*) decision-making process (Apostolakis 2004; NAS 2005). The ultimate purpose of the framework developed in this research is to provide the risk information necessary to the risk-informed decision-making process.

Probabilistic Health Risk Assessment

A recent addition to the human health risk assessment landscape is the development of probabilistic techniques for human health risk assessment[36]. By the early 1990s most human health risk assessments were based on using point values intended to represent upper-bound risk estimates (Finley and Paustenbach 1994); these analyses were thus denoted *point* or *deterministic* risk assessments. However, because of concerns of "compounding conservatism" introduced by using "worst-case" and bounding parameter

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[36] Probabilistic human health risk assessment techniques have also been referred to as *stochastic* risk assessments (Batchelor et al. 1998).

values when assessing exposure and risk (Burmaster and Harris 1993; Cullen 1994), risk assessors began in the early 1990s to investigate the well-established PRA techniques initially developed for reactor safety analysis (Keller and Modarres 2005; Rechard 1999) as a way to provide more accurate and meaningful information to risk managers. At a national level, the agencies regulating human health were lagging behind by the mid-1990s; there was no regulatory guidance for performing probabilistic health risk assessments (Finley and Paustenbach 1994)[37]. However, less than a decade later, guidance for introducing probabilistic techniques into human health risk assessment had been provided at both state and federal levels (USEPA 2001c).

A general equation for estimating the health *risk* from an exposure to a given individual (denoted the j[th] individual) can be represented by

$$Risk_j = \underbrace{f\left(\varpi_{1,j}, \varpi_{2,j}, \ldots, \varpi_{m,j}, v_{1,j}, v_{2,j}, \ldots, v_{n,j}\right)}_{exposure} \times \underbrace{v_{\text{toxicity},j}}_{slope\ factor} \qquad [3]$$

where *f*() is a function of *m* known or constant parameters $\varpi_{1,j}$, …, $\varpi_{m,j}$ and *n* uncertain parameters $v_{1,j}$, …, $v_{n,j}$ to estimate exposure, which to obtain an estimate of health risk is multiplied by the toxicity or slope factor, $v_{\text{toxicity},j}$, that is uncertain (and "conservative"[38]) but assumed fixed for regulatory purposes. Historically, despite the "conservatism" built into the toxicity, bounding or worst-case values for the uncertain parameters (e.g., body weight, breathing rate, ingestion rate, exposure frequency, etc.) were also selected (often by regulatory mandate) to intentionally overestimate exposure and health risk for a given individual or population (Anderson and Yuhas 1996; Burmaster 1996; Burmaster and

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[37] At the regional level, the U.S. EPA issued guidance on the use of probabilistic techniques for human health risk assessment as early as 1994 (USEPA 1994; 2001c).

[38] In this context, *conservative* means that the parameters are defined in such a manner as to result in a higher than expected risk per unit dose.

Bloomfield 1996; Burmaster and Harris 1993; Cullen 1994; NAS 1983; Pate-Cornell 2002). This method of estimating risk has other been referred to as *deterministic* risk assessment although *point estimate* risk assessment would appear more descriptive[39].

Because of the use of numerous "conservative" parameters and the non-linear nature of the exposure-risk relationship, the risk results obtained from point estimate risk assessments are often orders of magnitude higher than the 90th- to 95th-percentile values often dictated by regulation (Bogen 1994; Burmaster and Harris 1993; Cullen 1994; Finley and Paustenbach 1994; USEPA 1989). An example of the point estimate approach is illustrated in Figure 2[40] where the central tendency (CTE) and reasonable maximum (RME) exposures are compared to those from a probabilistic exposure assessment. More than 99.9% of the probabilistic results (using 95% percentile parameters) are less than the RME; thus the RME is highly "conservative" in this case.

However, the degree of "conservatism" or uncertainty in point risk estimates cannot be known *a priori*. Although uncertainty and sensitivity analyses were suggested by the EPA as part of risk characterization (USEPA 2001c), these analyses were often qualitative in nature and not integrated into the risk assessment[41]. The end result was that more remedial work was mandated and performed for a contaminated site than might have been warranted to be protective of human health.

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[39] The use of *deterministic* is a bit of a misnomer because the time evolution of the system being modeled cannot be predicted accurately nor is the system non-stochastic in nature. One has merely (and often arbitrarily) decided to simplify the analysis by selecting single values to account for both uncertainty and variability in the input parameters, many of which are stochastic. The use of average or some other percentile value may make the result more deterministic but really instead results in risk estimates of unknown uncertainty and degree of "conservatism."

[40] The probability density functions used for the exposure variables in this example simulation were originally described in the DRAFT version of the RAGS PRA manual, which is not available for citation.

[41] A sensitivity analysis provides little if any additional useful information if input parameters used to estimate exposure are already set to their worst-case or maximum values.

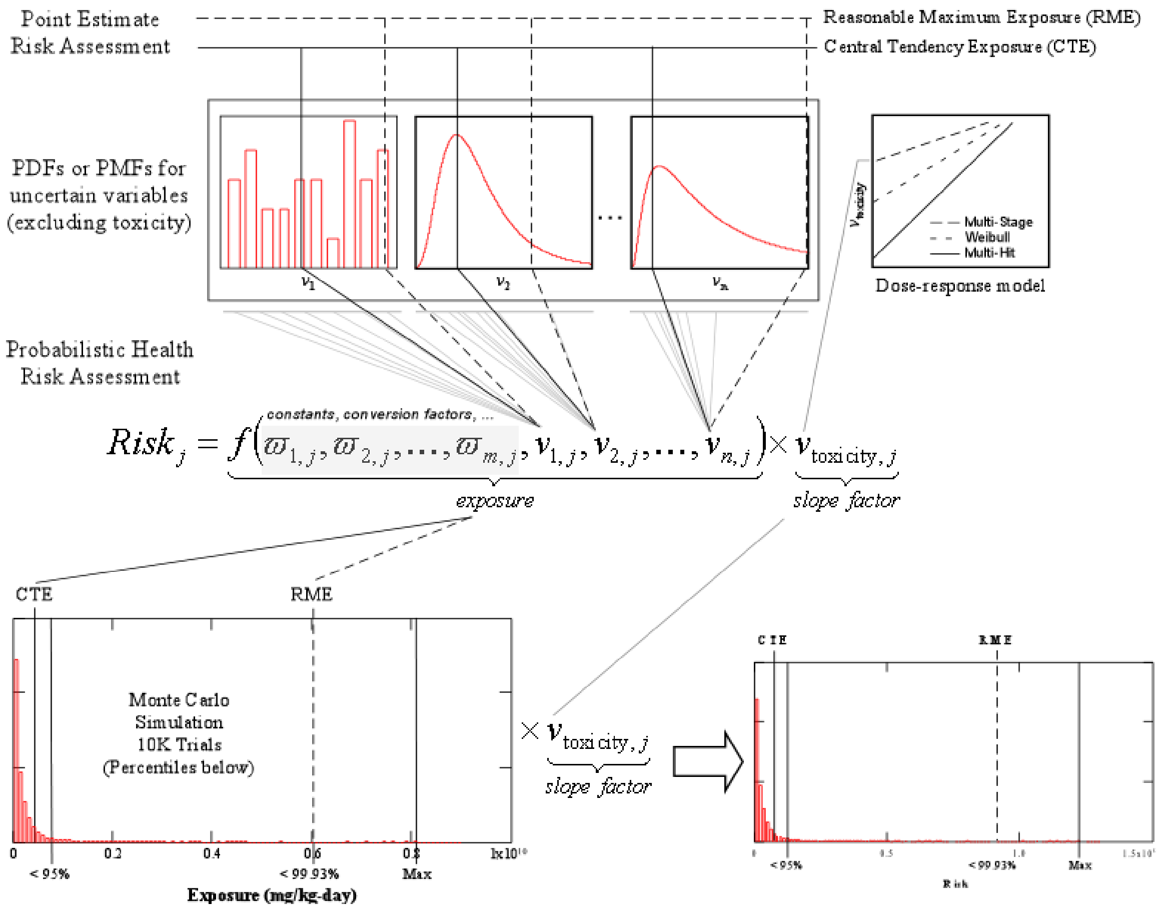

Figure 2. Conceptual models of regulatory point estimate and probabilistic *human health* risk (exposure) analyses for carcinogens (after (USEPA 2001c)). Uncertain exposure variables (v_1, v_2, ...v_n) are described by frequency and probability distributions. A unique exposure estimate is calculated for each set of inputs. Repeated random sampling results in a distribution of exposures, which can be converted to a distribution of risk using the mandated toxicity.

Because of concerns in the point estimate approach, the well-established PRA techniques developed for nuclear reactor safety applications began to be examined in the mid-1990s as a way to provide more comprehensive and useful information to risk managers (Finley and Paustenbach 1994). The most widely used PRA technique is Monte Carlo simulation, which was first used in 1946 to perform uncertainty analyses related to the development of the hydrogen bomb (Rugen and Callahan 1996; USEPA 2001c).

Probabilistic risk assessments rely on the same fundamental expressions as their point estimate counterparts where uncertain input parameters (i.e., $v_{i,j}$ in Equation 3 and Figure 2) are replaced by random samples from appropriate frequency or probability distributions[42]. The number of trials (a.k.a., realizations in PRA parlance) needed can determined by a trial-and-error examination of the stability of the output statistics although more rigorous techniques exist (Haldar and Mahadevan 2000). The distribution of results provides an estimate of the uncertainty in the output and can be used to estimate percentiles for risk management purposes.

There are differences between how probabilistic risk assessment is performed for human health risk assessment and how PRA is performed for reactor safety. One difference is that one of the most significant uncertain parameters in Equation 3, namely toxicity, is intentionally fixed for regulatory purposes (USEPA 2001c). In PRA for nuclear reactor safety, uncertainties may be unintentionally missed or underrepresented; however, significant uncertainties are not purposely omitted.

For health risk assessments, a *probabilistic exposure assessment* is performed and the output distribution is multiplied by a fixed toxicity to generate a distribution of risk values as illustrated in Figure 2[43]. Risk is not assessed probabilistically, which will artificially decrease the uncertainty in the resulting risk distribution and provide values that exceed the "true" or expected percentiles desired for risk management. These

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[42] To be most accurate, uncertain parameters should be separated into those that are uncertain and those that are variable and a two-dimensional Monte Carlo simulation performed (Burmaster 1997; Cullen and Frey 1999; Frey 1992; Frey and Bharvirkar 2002; Frey and Rhodes 1998; Hattis and Burmaster 1994; Hoffman and Hammonds 1994). The discussion will be limited to one of uncertainty for simplicity. An additional simplification is implied here—the uncertain input parameters are independent. If this is not the case, each realization would be computed from a randomly selected *n*-variate from the *joint* uncertainty distribution.

[43] "Risk" values are often calculated directly for each realization instead of the exposure distribution being calculated first and then multiplied by a fixed toxicity; however, mathematically the results are identical and the conclusion drawn remains.

differences do not mean that human health risks should not be assessed probabilistically; however, these issues do illustrate why it lends transparency to risk communication if the exposure and risk estimates are both presented to interested stakeholders.

Another interesting difference between health risk assessment and PRA involves the probability of exposure. For regulatory practices involving potential human exposure to radiation and hazardous chemicals, the probability that the receptor of interest will be exposed to the harmful agent is often assumed to be unity. In the process of evaluating potential exposures and receptors, conditions are evaluated to identify complete pathways and whether exposure is possible. For each complete pathway from source to receptor, models are typically used to estimate potential degrees of exposure based on time spent on-site; however, the likelihoods of exposure are not quantified and instead are, in essence, assumed to be unity. Exposures and risks computed in this manner are likely to be overstated and should not be confused with "true" or expected results[44].

The final difference between traditional and human health PRA methods involves the size of the potentially impacted receptor population. For regulatory purposes, a single, hypothetical receptor (or set of receptors) is often used to represent the exposed population of interest; in other words, the size of the potentially exposed population is often ignored for regulatory purposes. On the other hand, considerable effort is expended to estimate risks to all potential receptors in traditional PRA analyses for reactor safety and radioactive waste transportation analyses. Although this complicates the analysis, the size of the potentially impacted population would seem to be an important consideration

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[44] Attempts have been made to quantify exposure likelihoods for ecological risk assessments (Hope 2000; 2001; USEPA 2001c); however, few if any attempts appear to have been made for human receptors. It has also been indicated that the ecological dose-response relationship may be treated probabilistically for regulatory purposes (USEPA 2001c). Thus it appears that ecological probabilistic risk assessments may often be more faithful to traditional PRA methods than their human health counterparts.

in determining not only what remedial actions to take but also the order in which contaminated sites should be addressed.

Issues have been raised for both point estimate and PRA techniques for health risk assessment; however, these issues should not discourage the use of these techniques when warranted to support remedial action decisions. Table 6 provides a comparison of the important characteristics of these two general methods. Point estimates of human health risk are reasonable for screening purposes when knowledge of uncertainty is not critical, when risk reductions and trade-offs are being compared, and when decisions can be made despite the degree of uncertainty in the risk estimates. When more detailed and accurate risk estimates are warranted (e.g., as a result of the screening process), the probabilistic health assessment may be needed. However, the results from either type of assessment should not be mistaken for the "true" or expected risk results for a given set of conditions[45].

The most important point to remember is that the method for estimating risk, in either instance, must be clearly described including attendant uncertainties and their potential impacts on the remedial decision. Much can be learned from the traditional PRA methods as they have been applied to nuclear reactor safety and other industries. Concepts such as the size of the potentially impacted population matters and all significant uncertainties should be addressed (including those for toxicity and exposure) impact the remedial decision process as do other risk and non-risk factors.

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[45] This is the case when either point estimate or probabilistic techniques are employed to evaluate human health risks for exposures to chemicals and radiation and a mixture of uncertain and fixed slope factors or other default parameters are used (Anderson and Yuhas 1996; Burmaster and Bloomfield 1996; Burmaster and Harris 1993; Cullen 1994; Pate-Cornell 2002) or when uncertainties and variabilities are not treated appropriately (Bogen 1990; Burmaster 1997; Frey 1992).

Table 6. Distinguishing characteristics of point estimate and probabilistic human health risk assessment methods (Adapted from (USEPA 2001c))

Characteristic	Point estimate techniques	Probabilistic techniques
Ease of calculations	Calculations are simple and no sophisticated software is needed.	May require sophisticated software and unfamiliar analysis techniques for both exposure and uncertainty analysis.
Standardization	Regulatory agencies have often defined defaults values and methods.	Little standardization of default input distributions or techniques for generating site-specific input distributions although Monte Carlo analysis is a well established analytical technique.
Level of detail	Most appropriate for *screening analyses*. May allow for regulatory decisions to be made without additional information. Provides no incentive improved characterization of the site or evaluation of improved remedial methods.	Appropriate for detailed analyses including those requiring site-specific information. Should be performed after screening analysis to refine exposure and risk estimates. Provides information on missing information or additional characterization and data needed.
Uncertainty analysis	Often qualitative; however, central tendency and reasonable maximum exposure cases (if performed) provide semi-quantitative idea of uncertainty. Result often mistaken for "the answer" and importance of uncertainty may be unrecognized.	Can provide more comprehensive and meaningful use of available input information and characterization of uncertainties. Communication of uncertainty can either build trust or open the analysis to questions. May convey false sense of accuracy when data are sparse.
Sensitivity analysis	The use of default values provides no means for true sensitivity analysis and thus any such analysis (and its usefulness) is limited at best.	Used to identify the variables, models, and parameters that most highly influence the risk estimates.
Meaning of results	Provides no indication of relationship between point estimate risk and regulatory limit or confidence in point estimate risk.	Unlike true PRA, human health PRA is actually a probabilistic exposure assessment because of use of a fixed toxicity. The resulting risks are likely bounding but should not be mistaken for "true" or expected risks.
Resources	Methods are easy and inexpensive to perform and require little site-specific information.	May require considerable effort and resources to complete the analysis including development of probabilistic inputs and training of both assessors and managers.
Communication	Methods are easy to describe and understand (although many of the underlying assumptions may not be).	Provides much more comprehensive and useful exposure and bounding risk information to managers; however, may be more difficult to communicate results (and easy to obscure important assumptions). Any questions concerning PRA methods may impugn management decisions.
Public perception	Easy to understand and perform although still may omit relevant uncertainties and especially non-risk factors so mistrust still possible if not probable depending upon stakeholder perception.	Performance of these analyses is often mistaken for reason to intentionally misrepresent results or overspend or delay action.

Generally any approach that evaluates adverse impacts to human health from exposure to hazardous conditions could be considered a *human health risk assessment*. However, the concept of human health risk assessment used here is that recommended by the Presidential/Congressional Commission on Risk Assessment and Risk Management (P/CCRARM 1997a) and Human Health Evaluation Manual (HHEM) (USEPA 1989; 1991a; b; 1998b). The HHEM risk assessment framework extends consideration of human health risks during remedial activities to not only chemicals but also typical accident-related hazards found in other industries (USEPA 1991b).

The P/CCRARM suggested that risk assessment be extended beyond its narrow, scientific focus found in the *Red Book* (NAS 1983) to not only characterize the scientific and technical nature of a risk but to also note its "subjective, cultural, and comparative dimensions" and more actively involve stakeholders in the assessment (P/CCRARM 1997a). These concepts are consistent with the U.S. Nuclear Regulatory Commission (NRC) idea that risk is but one input to the decision-making process. Furthermore, there are important lessons to be learned from other risk assessment techniques (e.g., ecological, probabilistic, etc.) that should be incorporated into the human health risk assessment process when appropriate. All these concepts are important elements in making an informed remedial decision.

Selected Risk Assessment and Risk Management Paradigms

Regulatory actions have often been built upon two fundamental building blocks: *risk assessment* and *risk management*. *Risk assessment* involves estimating the effects to receptors from exposures to hazardous materials and conditions to evaluate potential harm. *Risk management* is a process of weighing alternatives and deciding on an appropriate course of action. A certain amount of interaction is recognized between the risk assessment and management processes (even if only during the *risk characterization* stage). Other non-risk factors including social, political, and economic issues often also play pivotal roles in the *risk management* process; therefore, decisions should be *risk-informed*[46] not *risk-based* (Apostolakis 2004; Callan 1998; Hornstein 1992; NAS 2005).

The focus in this research is on risk assessment and management practices used to satisfy regulatory requirements for *human health* concerns, especially for DOE sites contaminated with buried hazardous and radioactive wastes. Selected human health risk assessment and management methodologies are summarized in Table 7.

Human health risk assessment techniques have been extended to ecological receptors and to stressors other than chemicals. Frameworks have been proposed for assessing and managing human health and ecological risks (Jardine et al. 2003; Rasche 2001). Other frameworks have been developed to evaluate risks for financial institutions (Alexander 2005), nuclear reactors (Apostolakis 2004; NRC 2002), pest management (PMRA 2000), food and agriculture (McNab and Alves 2003a; b), product development (Sonnemann et al. 2003; USEPA 2000b), prescription drugs (FDA 1999), contaminated material reuse (Eighmy and Chesner 2001), water resource management (Hämäläinen et al. 2001; Lawrence and Shaw 1999), marine transport (IMO 2002a; b; USCG 1996a; b; c), radioactive waste transport (Gallegos and Channell 1990; Raj et al. 1996), emergency response (DCDEP 2000), and machine development (Anderson 2004). These techniques have been examined for pertinent input to the framework for evaluating the human health risks associated with the disposition of DOE buried waste sites.

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[46] The term "risk-informed approach" is defined for the NRC as follows: "A 'risk-informed' approach to regulatory decision-making represents a philosophy whereby risk insights are considered together with other factors to establish requirements that better focus licensee and regulatory attention on design and operational issues commensurate with their importance to health and safety" (NRC 1998). The use of "risk-informed" in this research is consistent with this definition outside the licensing context.

Table 7. Selected Human Health Risk Assessment and Risk Management Methodologies

Risk Analysis Report	**Description and Key Characteristics**
Risk Assessment in the Federal Government: Managing the Process (NAS 1983) or the *Red Book*	The roots of much of health risk assessment can be traced here. First formalized effort to describe human health risk assessment and management process in a structured way and consolidated earlier efforts at developing a comprehensive framework (Jardine et al. 2003). Suggested that risk assessment and management practices be kept separate. Assumed risk assessment was a scientific, value-free practice.
Guidelines for Carcinogenic Risk Assessment (USEPA 1986)	Set forth principles and procedures designed to establish "consistency and technical quality" in risk assessments and ensure that the risk assessment process is based upon valid scientific information. EPA published the final version of the guidelines in 2005 (USEPA 2005).
Risk Assessment Guidance for Superfund (RAGS) Human Health Evaluation Manual (HHEM) (USEPA 1989; 1991a; b; 1998b)	Goal is "to provide a framework for developing the risk information necessary to assist decision-making at remedial sites" (USEPA 1989). Provides information on baseline risks, remedial goals, and remedial action risks. Appears too broad in scope for efficient application to buried waste disposition.
Science and Judgment in Risk Assessment (NAS 1994b)	Indicated that EPA's human health risk assessment methods were sound although EPA needed to improve and establish more clearly the scientific and policy bases and describe the uncertainties and variabilities associated with health risk estimates. Indicated an iterative approach to risk assessment was needed as well as mixtures of toxic chemicals.
U.S. DOE Worker Health Risk Evaluation Methodology for Assessing Risks Associated with Environmental Restoration and Waste Management (Blaylock et al. 1995)	Can be used to estimate worker health risks over a broad spectrum of activities. Supports methodology used to assess human health risks for DOE Programmatic Environmental Impact Statements. Only generally follows RAGS HHEM (USEPA 1989) guidelines and thus might not be appropriate for RCRA or CERCLA sites.
Understanding Risk: Informing Decisions in a Democratic Society (NAS 1996)	Suggested that risk assessment needed to be an "analytic-deliberative" decision-making process. Making risk assessment results understandable to the lay people involves more than "translating scientific knowledge" (NAS 1996).
Framework for Environmental Risk Management (P/CCRARM 1997a) and *Risk Assessment and Risk Management In Regulatory Decision-Making* (P/CCRARM 1997b)	Considered by some to be the most influential framework (Jardine et al. 2003). Recommended that the traditional, scientific scope of risk assessment be expanded to include considerations of the *subjective*, *cultural*, and *comparative* dimensions of risk and that a fundamental change in paradigm was needed (e.g., include multiple sources and pathways). These ideas were presented previously in two 1994 NAS reports (NAS 1994a; b).
U.S. Navy Human Health Risk Assessment Guidance (USN 2001)	Made tiered approach (suggested by the RAGS HHEM (USEPA 1989)) explicit to incorporate risk management into the decision-making process, minimize the level of effort, and eliminate sites of no concern.

Table 7, continued

Risk Analysis Report	Description and Key Characteristics
Framework for Cumulative Risk Assessment (USEPA 2003)	In response to expert opinion and legislative direction to move beyond single chemical assessments and to focus on the cumulative effects of *chemical* exposures occurring simultaneously. First step in long-term effort to develop cumulative risk assessment guidance and serves to provide information not to establish cumulative risk assessment protocols.
INEEL Environment, Safety, and Health Risk Assessment Program (ESHRAP) (Eide and Wierman 2003)	Can be used to generate "best point-estimate" worker and general public risks from exposures to both radioactive and hazardous chemicals as well as industrial risks from activities associated with cleanup and management activities and typically covers the entire waste management program. No probabilistic or sensitivity analysis features available.

The process of risk assessment cannot be entirely divorced from that of risk management. Even though attempts have been made to maintain as much separation as possible between these processes (NAS 1983; Perhac 1996), it appears unwise to evaluate the risk information without considering the implications of the risk management and decision-making processes (NAS 1994a; P/CCRARM 1997a). One must be aware of the types of information—only one of which is risk—that are needed to make an informed decision. Trade-offs must be made between the types of risk information that must be provided and the social, political, and economic factors that will likely play significant roles in decisions concerning remedial activities.

The different types of risks that may be evaluated over the life-cycle of a major site disposition may have very different bases (e.g., latent cancer fatalities, non-cancer health effects, traumatic injuries and fatalities, etc.); therefore, no rigorous, normative basis can be devised for their comparison (Arrow 1951; Hornstein 1992). For example, some techniques are used to translate risks to a common basis (i.e., monetary value) so that the costs and benefits resulting from proposed remedial actions can be compared in a straightforward manner (Linkov et al. 2004). However, necessary value judgments (e.g., cost of a life lost, years of life lost, etc.) and conversions often obscure important information or negate preferences from decision-makers and other stakeholders. These issues have often led to a false sense of certainty in the results and ultimately a loss of credibility in the rankings generated.

There are times that risk assessment can be simplified to an evaluation of reductions in life-cycle risks[47]; this simplification can provide a more streamlined and efficient approach to risk comparisons. However, it is unlikely that issues related to a lack of a normative basis will be resolved because even the reduced set of risks will likely have different bases. The risks and risk trade-offs corresponding to remedial alternatives must be presented as the raw input to the decision-making process. Any constructed normative analysis of risks must be provided after the risk trade-offs have been presented.

Many health risk assessment approaches have corresponding or integrated risk management frameworks. For example, the P/CCRARM approach integrates risk analysis and management as shown in Figure 3 (P/CCRARM 1997a). This framework was developed to help meet the needs of addressing multiple environmental media and sources of risk in an iterative fashion while involving stakeholders at every stage (P/CCRARM 1997a). Other frameworks focus on different areas of risk management (e.g., stakeholder involvement, transparency, tiered risk management, the decision-making process, etc.) or on specific areas (e.g., food safety, hazardous materials transport, etc.).

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[47] For example, if residual risks for all proposed remedial actions are either negligible or acceptable, then these risks may be ignored in the general analysis.

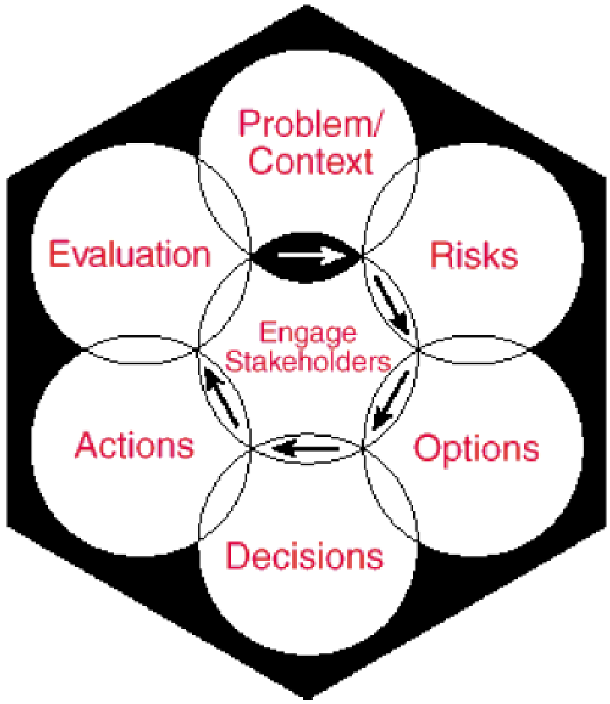

Figure 3. P/CCRARM Framework for Risk Management (P/CCRARM 1997a)

Despite the wealth of health risk assessment and management strategies available in the literature, a number of key resources for risk assessment and management practices can be identified that include the *Red Book* (NAS 1983), Human Health Evaluation Manual (USEPA 1989; 1991a; b; 1998b), and P/CCRARM report (P/CCRARM 1997a). These reports form the foundation of much of the health risk assessment work that is currently being performed although additional frameworks have been developed to systematize assessments of multiple hazards and pathways, formalize tiered assessments, develop frameworks for specific applications, etc. This additional information has been evaluated for pertinent input to the framework and methodology developed in this research for the consistent technical evaluation of the risks associated with buried waste disposition as one input among others to the decision-making process.

Impact of Uncertainty on Health Risk Assessment

Properly addressing uncertainty is of critical importance to communicating health risk assessment results in a transparent fashion[48]. The fact remains that uncertainties and missing information are unavoidable in any site evaluation. Decisions must be made in the face of these uncertainties; uncertainty and need for additional information cannot be allowed to delay necessary remedial actions or permit assessors from generating risk information consistent with preconceived notions. Therefore, to provide transparency, meaningful exposure, risk, and uncertainty information must be provided as well as how uncertainties and missing information might impact the decision-making process.

Two typical ways of classifying uncertainties in health risk assessment are found in the literature (NAS 1994b). One method classifies uncertainties based upon where in the risk assessment process they occur (Bogen 1990; NAS 1994b). A more common approach categorizes uncertainties into abstract, general categories. For example, one set of uncertainty categories is *bias*, *randomness*, and *variability* (NAS 1994b). Another set (i.e., *parameter*, *model*, and *scenario*) was suggested by Linkov and Burmistrov (2003)[49]:

- *Parameter uncertainty*—lack of knowledge in the true value of an input parameter to a model.
- *Model uncertainty*—lack of knowledge about the structure and accuracy of the model used (including impact of simplifying assumptions and mathematical representations).
- *Scenario uncertainty*—lack of information regarding missing or incomplete information needed to adequately define the model; this lack of information is sometimes referred to as modeler uncertainty (Linkov and Burmistrov 2003).

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[48] In this research, the concepts of uncertainty and variability are distinct. Uncertainty denotes a lack of information; whereas, variability describes true heterogeneity.

[49] A similar categorization was provided earlier by Konikow and Bredehoeft (1992).

The first two categories above comprise the preferred taxonomy in *Science and Judgment in Risk Assessment* (NAS 1994b); however, the third category may be critical and can, in some cases, dominate the overall uncertainty in risk estimates[50].

Other taxonomies for classifying uncertainties have been proposed (Cullen and Frey 1999; Morgan et al. 1990; Stirling 2003; USDOE 2000; USEPA 1992b; 1997b; c; Yoe 1996). One element that runs through these taxonomies and risk assessment is the need for *expert judgment* to determine the appropriate parameter values, distributions, models, and scenarios. Expert judgment is valuable in that these persons often have the greatest experience with these types of problems; however, experts' judgments often suffer from the same biases as lay people, especially when forced to rely upon intuition (Kahneman et al. 1982; Slovic 1987; Slovic et al. 1979). Stakeholder input must be included in the process, or there is likely to be a lack of transparency resulting in mistrust of the analysis based upon expert subjectivity or preconceived notions and attitudes.

Uncertainties will be a part of any risk assessment (including those relying on point estimates) and *cannot* be removed entirely from the analysis. However, this does not mean that meaningful estimates and comparisons of risks cannot be made. A better approach is a consistent approach to classifying, estimating, and reducing uncertainties commensurate with their potential impact on the decision-making process.

[50] One study found that the greatest uncertainty resulted from modeler's interpretation of scenarios resulting in differences in predictions of *seven orders of magnitude* (Linkov and Burmistrov 2003).

Improving the Health Risk Assessment Process

Considerable anecdotal evidence and the results of various studies would appear to impugn the ability of the health risk assessment process to provide meaningful inputs to the decision-making and risk management processes. A few concepts that would appear to undermine the value of health risk assessment to decision-making include:

- There are differences in how (and whether) cancer develops in humans versus the animals (e.g., mice, rats, etc.) used to study dose-response relationships (Cunningham 2002; Trosko and Upham 2005). Thus experimental evidence that an agent is an animal carcinogen does not necessarily mean that the same agent is a human carcinogen. However, this begs another interesting issue: Does the lack of experimental evidence that an agent does not produce cancer in a particular animal species mean that it is not a human carcinogen? This notion has led to the development of biologically-based models for studying the dose-response relationship although enthusiasm for such models appears to have waned recently for some (Crump 2003).

- Environmental exposure of humans is typically two to five orders of magnitude less than the lowest dose typically tested in the corresponding animal study (Paustenbach 2002). Because one must extrapolate to very low doses, there is no objective, *a priori* way to determine which of the myriad models available (e.g., Multi-Hit, Multi-Stage, etc.) is "best" or the degree that the extrapolated dose-response relationship may or may not overestimate the "true" or expected risk. A bounding value is often selected to represent the dose-response for the agent at low doses, which some mistake for the "true" or expected risk per unit dose. The fact that carcinogens and non-carcinogens are regulated differently also introduces confusion and may tend to overemphasize the relative importance of carcinogenic risks to the public (Crump 2003).

- Many health risk assessments have suffered from compounding conservatisms when estimating potential human exposure, resulting in risk estimates that may border on being meaningless or, at least, mandating overly expensive cleanup relative to the risks reduced (Bogen 1994; Burmaster and Harris 1993; Cullen 1994; Finkel 1989; Maxim 1989; McClellan 2003; Nichols and Zeckhauser 1988). Although bounding or worst-case parameters were often dictated to risk assessors (based upon regulators' judgments), the use of these parameters results in exposure and risk estimates of unknown "conservatism." A case can be made that mandating bounding parameters makes the results much more difficult to characterize (because the uncertainties in the results often cannot be quantified).

- For exposure assessment, model uncertainty can range over several orders of magnitude and modeler uncertainty can vary by more than four (and perhaps as many as seven) orders of magnitude (Linkov and Burmistrov 2003). It can be demonstrated that modeler uncertainty can be significantly reduced using an analytical-deliberative process and paying particular attention to how scenarios are constructed (Linkov and Burmistrov 2003).

- When one considers the life-cycle of a contaminated site disposition, many different types of risks (e.g., fatalities, injuries, latent cancer effects, etc.) with attendant uncertainties must be evaluated. Different types of risks cannot be compared using a single normative basis (Arrow 1951). There are also many non-human health risk factors (e.g., costs, social values, loss of habitat, etc.) that must be factored into the analysis further complicating the decision-making process.

General acknowledgment (including by many in the scientific community) of the concerns resulting from large uncertainties attendant in risk assessment were thought by some to have portended the “waning days of risk assessment” (Montague 1999) as a meaningful input to the risk management process[51]. However, despite such portents, health risk assessment continues to be a valuable input to the decision-making process for managing risks associated with contaminated sites. This does not mean that the health risk assessment process cannot be improved to increase both its value and acceptance.

Many potential improvements to health risk assessment have been suggested since the *Red Book* was published in 1983. In 1995 Paustenbach (1995a; 1995b) provided a set of lessons learned to help improve the scientific conduct of health risk assessments. Examples included not considering all animal carcinogens as serious human hazards, presenting upper bound and expected dose-response estimates, avoiding compounding conservatism and using probabilistic techniques whenever possible, and placing risk estimates into proper perspective. In 1997 the Presidential/Congressional Commission on Risk Assessment and Risk Management (P/CCRARM) published a report recommending

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[51] The Environmental Research Foundation published an Internet article in 1999 entitled "The Waning Days of Risk Assessment" stating that "[r]isk assessment, it is now clear, promises what it cannot deliver, and so is misleading at best and fraudulent at worst ... Risk assessment is inherently an undemocratic process because most people cannot understand the data, the calculations, or the basis for the risk assessor's judgment" (Montague 1999).

that the traditional, scientific scope of risk assessment be expanded to include considerations of *subjective, cultural, and comparative dimensions* of risk and further suggested that a fundamental change in the risk assessment paradigm was needed that included multiple sources and pathways (Jardine et al. 2003; P/CCRARM 1997a)[52].

There have been numerous suggestions on better methods to handle uncertainties in the risk assessment process (Morgan et al. 1990; Pate-Cornell 1996; Paté-Cornell 1999). The EPA has gone so far as to provide suggestions on the types of probabilistic methods and distributions that can be used to incorporate uncertainties into human health risk assessments (USEPA 1997c; 1999; 2001c). However, despite these improvements to the scientific and communication aspects of risk assessment, it would be difficult to argue that the results from recent health risk assessments are any better accepted (certainly by stakeholders) than they were a decade ago. The results may even be less well accepted because of the increased understanding and sophistication on part of the stakeholders.

Several general problems appear to recur in the public acceptance of health risk assessment results for determining the remedial actions that should be applied to a contaminated site. One is technical in nature in that the environment and the migration of contaminants must often be *modeled* to provide sufficient assurance that remedial action is first needed and then would be effective. Application of this technical process begs two thorny problems. The first problem involves how the environment must be modeled and that fact that *any simplification* (and many will likely be required) may promote distrust in the results especially if the risk assessor already has engendered an air of mistrust on the part of stakeholders. The second issue involves how to compare risk estimates, which

[52] These ideas were presented previously in two 1994 NAS reports (NAS 1994a; b).

may be of different types, not only to other types of risk but to other non-risk factors (social values, cost, etc.) that will ultimately be included in the decision-making process.

The reason that the concepts of consistency, transparency, and trust are so important to health risk assessment is that the model and normative problems cannot be solved. The environment cannot be modeled without using simplifying assumptions and no normative basis can be defined that allows different types of risk and non-risk factors to be compared so that an objective rank-ordering can be generated (Arrow 1951). These factors may help explain why even though great strides have been made in many technical areas needed in risk assessment[53] there still appears to be a basic mistrust of health risk assessment and the results generated. Some of this lack of trust may be due to past practices and misunderstanding, which will take much time and effort to overcome; however, improvements in the risk assessment process itself appear to be needed.

Despite concerns with using risk assessment techniques to evaluate contaminated sites, there are reasons that risk assessments will continue to be performed for the foreseeable future. The first reason is that many Federal Agencies (e.g., EPA, FDA, DOE, etc.) require risk assessments and Federal and State regulations (e.g., CERCLA, RCRA, Toxic Substances Control Act, etc.) often contain provisions that either mandate

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[53] Some of the scientific and technical advances include identifying hazardous agents and modeling their effects in animals and humans, extrapolating human effects from animal studies, understanding fate and transport of agents in the environment and resulting human exposure (which may have seen the greatest strides), and translating assessment results into more understandable forms (Edler and Kitsos 2005; Goldstein 2005; Paustenbach 2000; 2002).

or promote the use of risk assessments[54]. The second, and scientifically more palatable, reason is that risk assessment is the *right* thing to do when evaluating risks posed by contaminated sites and their dispositions.

Some have proposed to supplant risk assessment with some form of precautionary approach (Cross 1996; Montague 1999; Wiener 2002); however, two things would be certain if the precautionary principle is adopted in principle: the impetus for technical innovation will be diminished and, without assuming some degree of risk, there will be no benefit. Instead of surrendering to the notion that no involuntary risk is acceptable, the focus should be shifted to the risk assessment process as a journey of organization and discovery and not on risk results and their attendant uncertainties as absolutes. Risk assessment is a tool that can be used to identify and evaluate relevant risks, and the risk information generated is one set of inputs (along with non-risk factors) to the decision-making process. Employing risk assessment not only identifies relevant risks and uncertainties, perhaps more importantly, it should help focus attention on the critical assumptions made and the significant contributors to risk, which are issues critical to the decision-making process and can help focus additional research.

The suggestions made thus far have focused on understanding the benefits of risk assessment and placing both the assessment and results in their appropriate contexts. Many suggestions have been made to improve the scientific and technical techniques constituting the risk assessment process (Edler and Kitsos 2005; Goldstein 2005;

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[54] In 1980, the U.S. Supreme Court [in *Industrial Union Department, AFL-CIO v. American Petroleum Institute*, 448 U.S. 607 (1980)] overturned the OSHA standard for occupational exposure to benzene requiring that the agency first prove that a chemical poses a "significant risk" before issuing a standard (Jardine et al. 2003; Martonik et al. 1998). Therefore, risk assessment would likely be required for such proof; however, the decision also affirmed the Agency's right to impose conservative assumptions in evaluating carcinogenic hazards to err on the side of over-protection (Adler 2003; USDOL 1997b). Thus the "true" or expected risks once again could be excluded from the regulatory process.

Paustenbach 2000; 2002); however, few if any will seemingly have a significant impact on the issues that have led to mistrust in the process and the results. For example, how will improvements in extrapolating human effects from animal studies at different doses provide confidence in the dose-response relationship when the fundamental issues remain the same? How will replacing the basis for such extrapolation with a biologically-based model do anything but open up a different model to questions concerning assumptions, uncertainty, etc.? The most sophisticated fate and transport models remain simplified representations of the true movement of contaminants through the environment. What does it mean to the stakeholder when the site contaminants represent a "risk" of less than 10^{-6} when the uncertainty may be many orders of magnitude?

Despite suggestions otherwise, a great deal of effort continues to be expended on refining “assumption-laden mathematical estimates of small risks” (P/CCRARM 1997a) and arbitrarily reducing uncertainties in parameters and models, instead of expending the effort required to reduce risks and improve human health and the environment. Improving needed models and reducing uncertainties may help increase acceptance among the scientific community; however, general acceptance of risk assessment as a necessary input to the risk management process requires a much different focus.

To truly improve risk assessment and its acceptance requires focusing efforts on consistency, transparency, and trust and not only on improving technical elements that will likely never be fully understood by broader audiences. There are many qualities of risk assessment that can help lend consistency and transparency and ultimately trust. The risk assessment process should be *tiered* so that the level of detail in the analysis of both risk and uncertainty and the types of simplifying assumptions tolerated is commensurate with the importance, complexity, and stage of the buried waste site disposition. It is also important that risks and risk trade-offs be the primary foci of the assessment to reduce the impact of the large attendant uncertainties to the point possible. The risk assessment process should be *iterative* so that the risk assessment can be updated as new information is obtained, new questions are asked, or regulations are changed that must be addressed. Risk assessment should be thought of as a journey much more than a goal. This journey, to which all interested parties must be invited and early, must address all relevant risks and consider the impacts of uncertainty consistently and transparency so that trust can be engendered once again.

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CHAPTER III

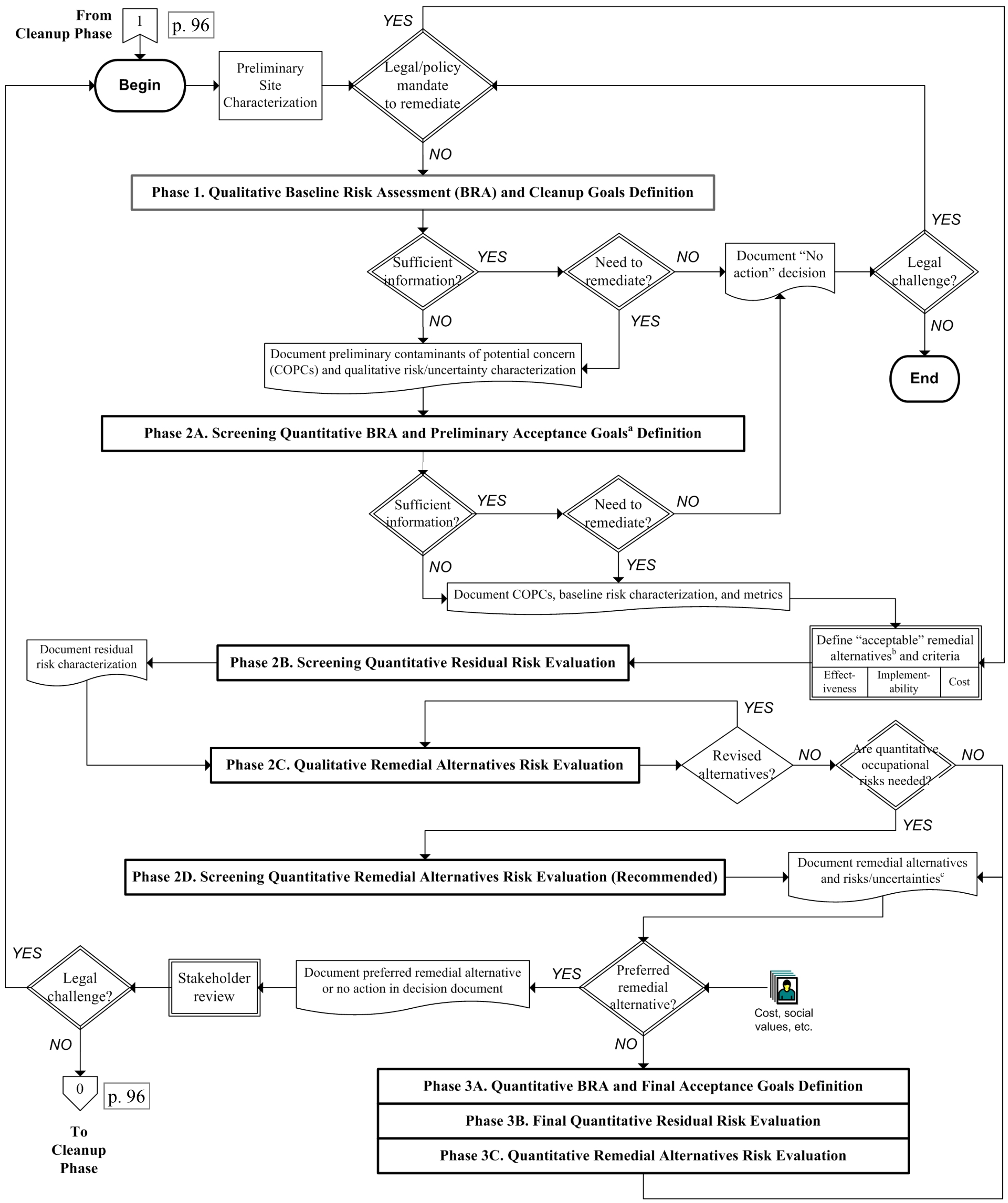
THE LIFE-CYCLE RISK ANALYSIS FRAMEWORK AND METHODOLOGY FOR DEPARTMENT OF ENERGY (DOE) BURIED WASTES

This chapter describes the development of the general life-cycle risk analysis framework and methodology for assessing the risks and risk trade-offs associated with the disposition of buried wastes managed by the U.S. Department of Energy (DOE). The buried wastes in question were generated from historic nuclear material production. The results from the application of this framework can be used as one input, along with other non-risk factors, to a risk-informed decision-making process.

Risk Analysis Framework for Department of Energy (DOE) Buried Wastes

The graphical framework shown in Figure 4 and Figure 5 outlines the general process for the evaluation and comparison of the risks and risk trade-offs involved with either managing buried wastes in-place or retrieving the wastes for treatment and disposal. The symbols used in the framework diagrams are defined in the Framework Symbols section provided in the front matter.

Only the high-level details of the framework are presented in Figure 4 and Figure 5; the framework is presented in greater detail in Figure 6 through Figure 10. The methodology (i.e., instructions, conceptual models, diagrams, metrics, etc.) is defined describing the application of the framework to DOE buried waste sites. The resulting framework and methodology is ultimately applied to two prototype sites to demonstrate the value of the framework and methodology in evaluating buried waste sites.



Notes:

a. For example, preliminary acceptance goals could represent preliminary remediation goals (PRGs) per CERCLA (USEPA 1991a).
b. Using expert judgment, three of the nine CERCLA evaluation criteria are used (40 CFR 300); others will be included if mandated by law.
c. For this study, the two general categories of remedial alternatives are retrieval/treat/dispose and contain in-place. However, no action can also be a viable alternative depending upon the results of the various assessments.

Figure 4. Framework for Assessing the Life-Cycle Risks Associated with Disposition of Buried Wastes (Overall Framework). Symbols are defined in the front matter.

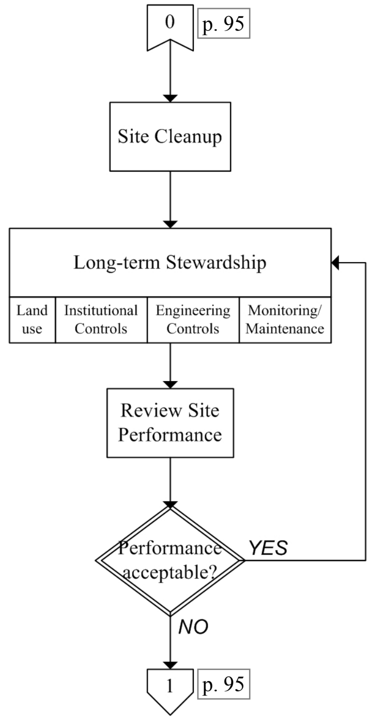


Figure 5. Framework for Assessing the Life-Cycle Risks Associated with Disposition of Buried Wastes (Cleanup Phase). Symbols are defined in the front matter.

Risk Analysis Methodology for Department of Energy (DOE) Buried Wastes

The methodology applies to the general framework illustrated in Figure 4 and Figure 5 (and the detailed versions shown in Figure 6 through Figure 10). The framework and methodology apply to contaminated DOE buried waste sites. Many of these sites are managed under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) or Resource Conservation and Recovery Act (RCRA) or both[55]. However, the framework and methodology are designed to be generic in nature and should be applicable to buried waste sites administered under other laws or regulations.

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[55] The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) or "Superfund" addresses uncontrolled releases of hazardous substances from abandoned or non-operating facilities or those that present an immediate threat to human health and the environment. The primary focus of the Resource Conservation and Recovery Act (RCRA) is on operating facilities.

Phase 0: Preanalysis Activities

Phase 0, which is denoted by a dashed box (beginning with "Site Identification") in Figure 6, describes those activities undertaken before any technical evaluation of the buried waste site is required. Before any remedial action is considered, the site must be identified for possible action. Site identification can take place in several ways. For example, sites are typically identified for CERCLA action from information supplied by states or waste handlers although citizens can also petition the U.S. Environmental Protection Agency (EPA) to investigate a site. Many DOE facilities have been placed on the National Priorities List (NPL), a list of national priorities for known or threatened releases of hazardous contaminants that is intended to guide the EPA in determining those sites that warrant investigation[56].

Once a suspect buried waste site is identified, a preliminary site characterization is performed to identify what is known about site conditions and contaminants and hazards. If there is no immediate, legal mandate for site remediation[57], Phase 1 of the analysis framework calls for a *qualitative assessment* of existing site conditions based upon existing information, conservative assumptions, and expert judgment.

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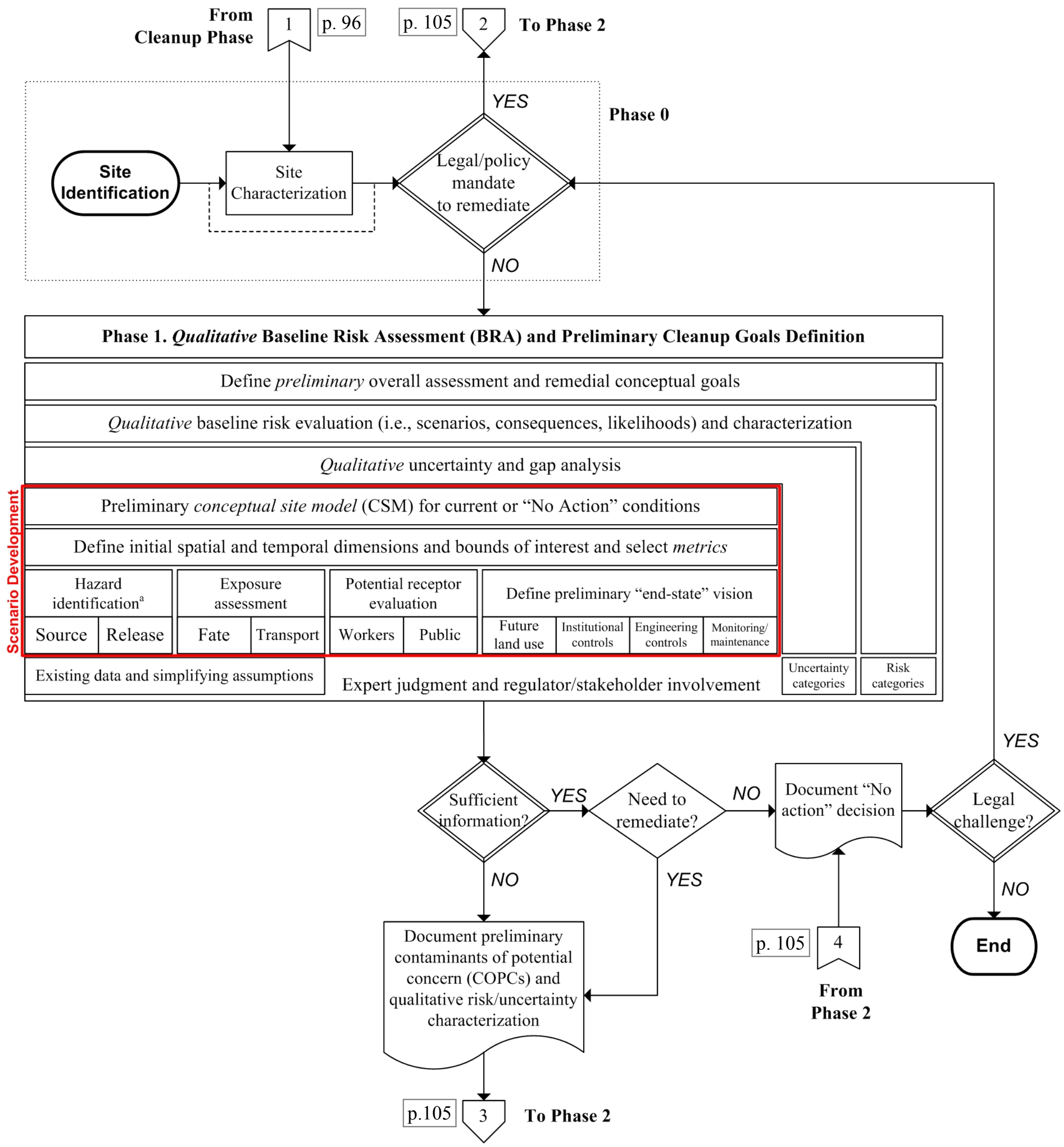
[56] More information can be accessed on both CERCLA (a.k.a., "Superfund") and the National Priorities List (NPL) at http://www.epa.gov/superfund/ (accessed March 6, 2008).

[57] As illustrated in Figure 6, there may be a legal mandate for remedial action for a site before any formal characterization is performed. However, if this is the case, it is assumed that some initial site characterization will be completed. If the site poses an immediate threat to human health and/or the environment, then early remedial actions can be taken to mitigate immediate site hazards.



Notes:
Each phase builds from the lower to higher steps.
a. Toxicity implicit in hazard identification based on the assumed veracity of EPA (toxicity) information.

Figure 6. Risk Assessment Framework for DOE Buried Wastes—Detailed Phase 1. Symbols are defined in the front matter.

Phase 1: Qualitative Baseline Risk Assessment and Cleanup Goals Definition

As illustrated in Figure 6, Phase 1 of the framework calls for a *qualitative assessment* of existing site conditions and potential hazards using existing information, conservative assumptions, and expert judgment to determine:

- if sufficient information exists to make a remedial decision *and*
- if remedial actions are required based upon the qualitative assessment.

The intent of this initial evaluation phase is intended to be more organizational than technical in nature.

The risk analysis framework is *tiered* so that the level of analysis is commensurate with the magnitude of the suspected impacts of contaminants and complexity of the buried waste site and likely remedial decisions and actions. The framework is *iterative* both *explicitly* and *implicitly*. As indicated in Figure 4, the assessment process may be explicitly restarted or revised based upon new information or regulatory decisions. Furthermore, as the assessment progresses, the information used to estimate risks is updated to include more accurate models and site-specific data.

The basic building block of the *qualitative baseline risk assessment* (Phase 1) is illustrated in Figure 7. The foundation for the building block is the set of four elements suggested previously (NAS 1983) supplemented with information concerning future land use and the relevant potential receptors. The analysis in the block builds from the bottom up in terms of the information needed to complete each step. This information is updated during subsequent steps in the assessment process as new information is obtained and/or needed to perform more detailed assessments.

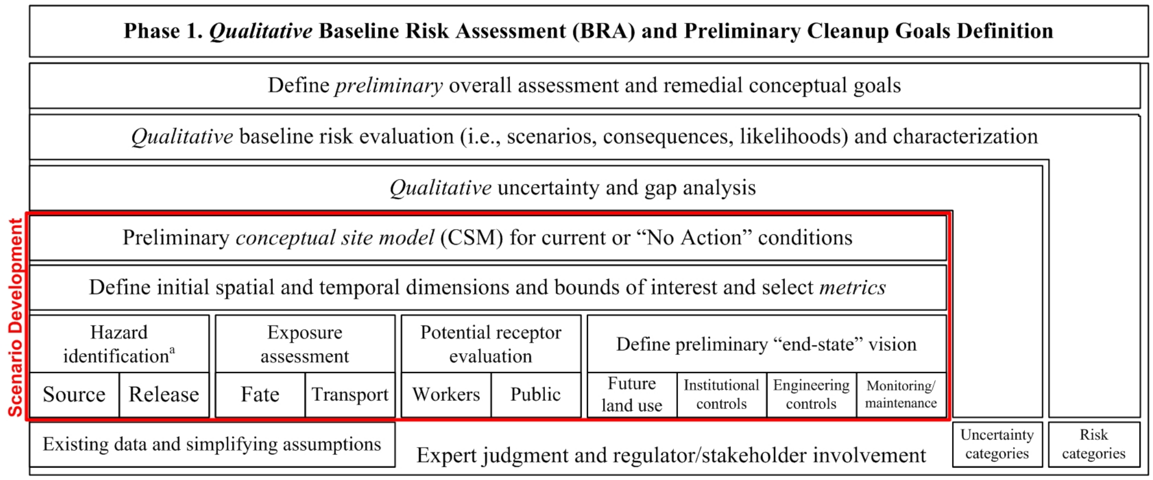


Figure 7. Building Block for Phase 1: *Qualitative* Baseline Risk Assessment

Conceptual Site Model Development

As illustrated in Figure 7, the first critical step in Phase 1 is development of a conceptual site model (CSM) from available information[58]. Conceptual site models graphically illustrate the relationships between contaminant sources and potential receptors via transport pathways and exposure routes (ASTM 1995; USDOE 2003). The CSM thus ties together essential risk concepts and aids in identifying failure scenarios. The CSM is a mature technique to provide this critical information for a contaminated site in a transparent manner[59]. The example generic CSM (including the narrative) for exposure hazards corresponding to baseline conditions for the conceptual burial model defined in Chapter V is presented in Figure 8.

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[58] Other pictorial representations of site conditions, contaminant movement, etc. may be needed to convey information to the stakeholder; however, the conceptual site model, which links contaminant sources to receptors, is seen as the *minimum* graphical representation required for transparency.

[59] Often, in human health or ecological risk assessments, the mere fact that there is a *possible* (not always probable) path of contaminants from source to potential receptor is enough to obviate the remaining risk-triplet questions. The purpose of this risk analysis framework is to consider all aspects of risk.

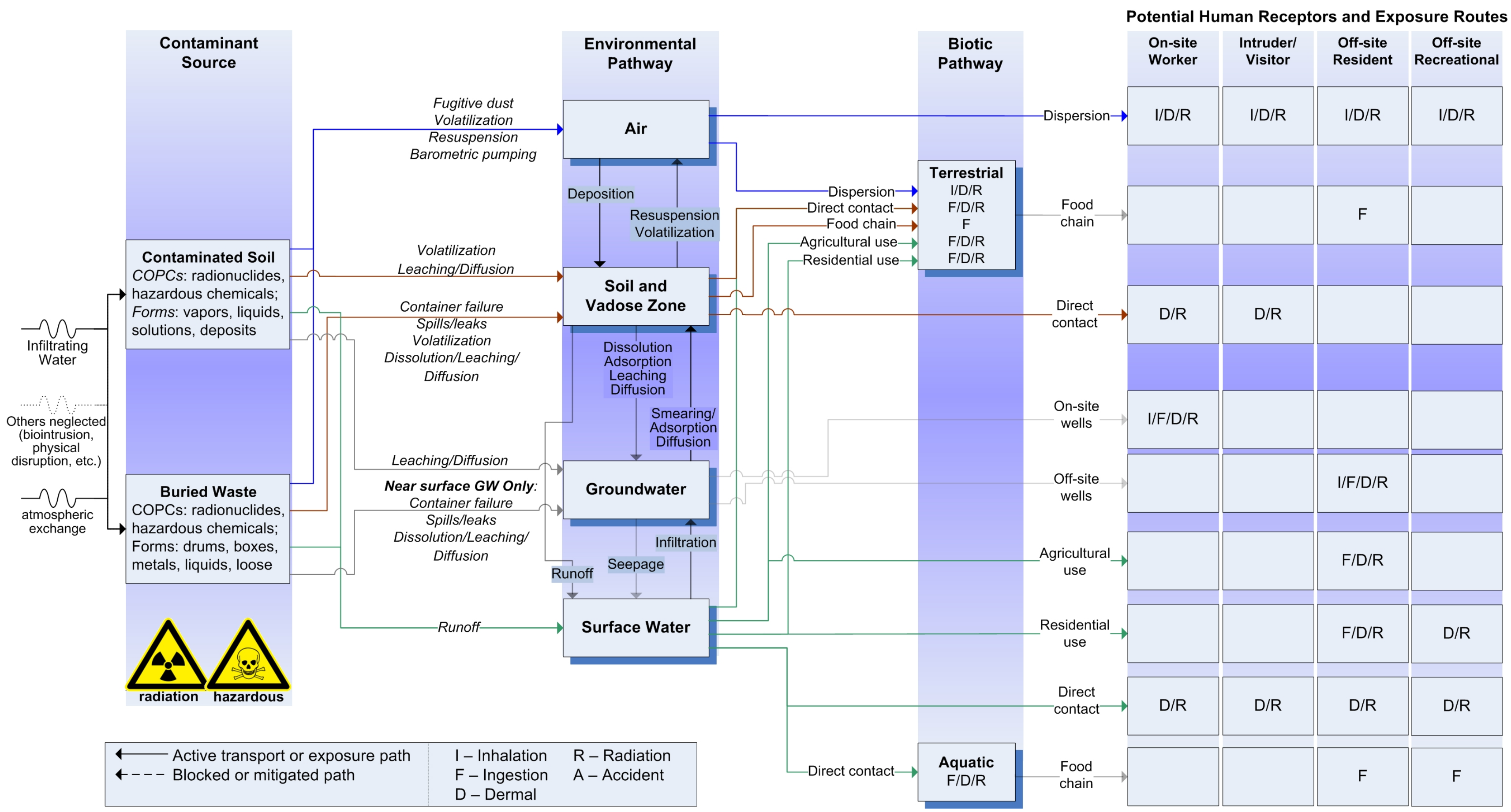


Figure 8. Simplified, generic baseline conceptual site model (CSM) representing *risks from exposures* to chemicals and radionuclides **before any remedial actions have been undertaken**.

Narrative for Figure 8: Simplified, Generic Conceptual Site Model (CSM) for Baseline Exposure Risks

A brief description of the high-risk chemicals (e.g., volatile organic compounds, nitrates, etc.) and radionuclides (e.g., short-lived fission products, long-lived transuranic (TRU) elements, etc.) is provided in this paragraph. Specific examples might be that volatile organic compounds (e.g., carbon tetrachloride, tetrachloroethylene, etc.) pose the most imminent risk to human health and have been detected in the environment near the burial site above their maximum contaminant levels (MCLs). Mobile long-lived fission and activation products might be the next most immediate concern based upon information concerning inventory, waste forms, migration pathways, etc.

Any barriers or steps taken to mitigate impacts for the CSM in Figure 8 are provided next. For baseline conditions, no barriers or other actions may have been taken to date. However, for complicated DOE sites, early actions may have already been taken to mitigate highest-profile risks. Example barriers and mitigation actions might include:

1. A vapor extraction system is being used to mitigate VOC migration to the aquifer.
2. The site has restricted access to prevent intrusion by the public, and the site is surrounded by a security fence.
3. An extensive groundwater-monitoring program is in place at the site. Drinking water wells used to supply potable water to the work force are located outside known contaminated areas and are routinely monitored for water quality.
4. Natural attenuation of volatile organic compounds, nitrates, short-lived fission products (e.g., Cs-137, Sr-90, etc.) will prevent some of them from reaching the aquifer.
5. An extensive surface water management system, including dikes and drainage channels, has been installed at the site to minimize the potential for flooding and releases by way of surface water.

Other pertinent information is supplied here. For example, if there is already extensive contamination in the environment from the buried waste site, then a pathway may exist from the burial site to a receptor. In this case, the only potential “barriers” that exist are decay, dispersion, and dilution for the radionuclides and dispersion and dilution for the inorganic constituents and heavy metals.

An essential part of CSM development (and Phase 1) is consideration of the spatial and temporal dimensions of interest; potential receptors (e.g., workers, general public, etc.); how or if the contaminants can migrate to receptors; how receptors might be impacted by exposure to contaminants; and when specific receptors might be impacted. While the baseline CSM in Figure 8 follows recommended guidelines (ASTM 1995; USDOE 2003), this diagram is unique in that it indicates the temporal nature of the health exposure risks posed by the site. The darker shading of the transport pathways in Figure 8 corresponds to the more immediate the impact of the potential exposure. Depending on the types and extent of information available, additional diagrams should be developed when needed to illustrate contaminant sources, subsurface stratigraphy, contaminant movement, etc. (Meyer and Gee 1999).

Qualitative Uncertainty and Gap Analyses

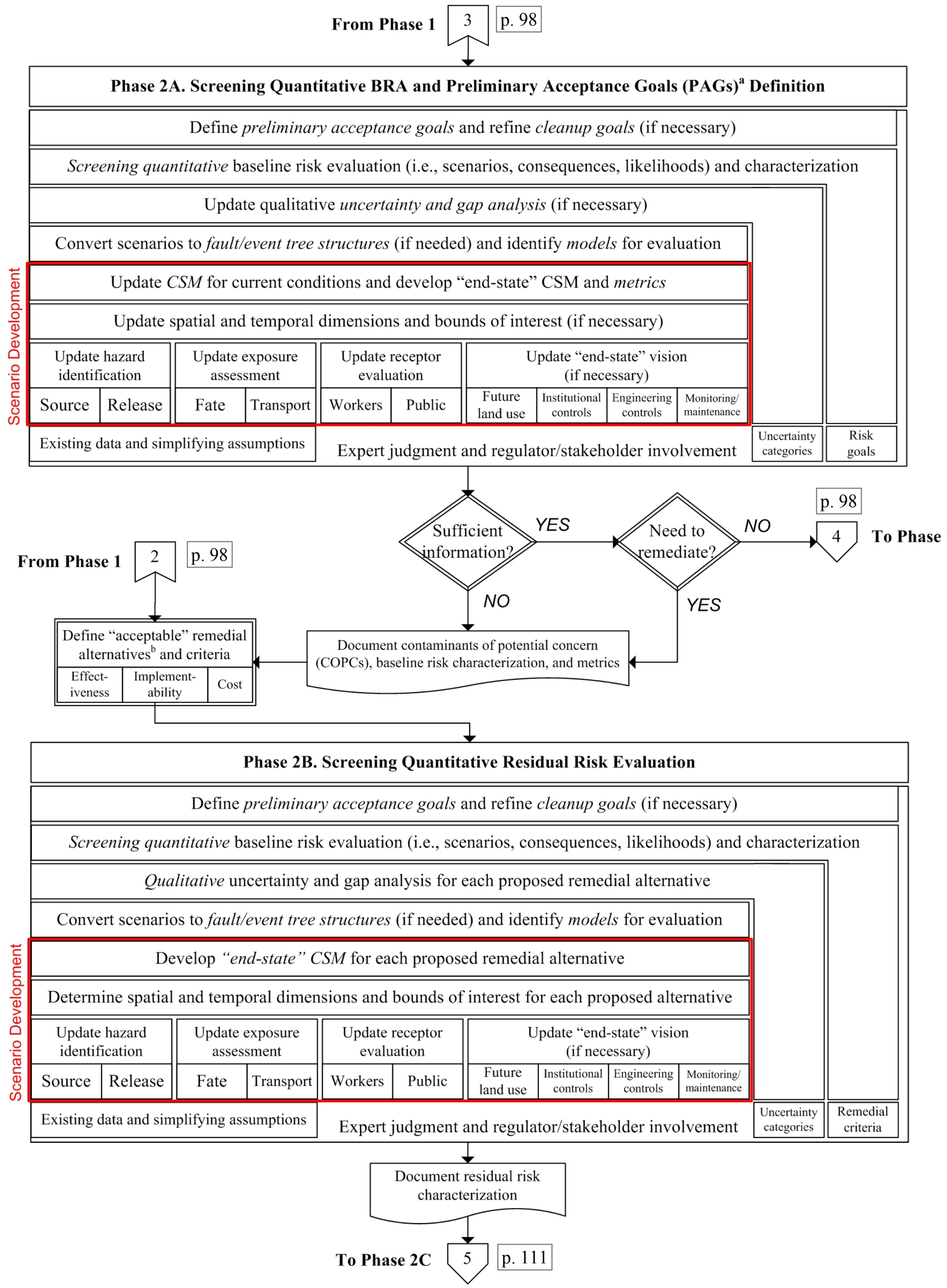
A qualitative analysis is performed using existing information to identify critical uncertainties and gaps in information that might impact the ability to make a remedial decision. A qualitative baseline risk evaluation is performed using available information to assess potential site hazards using the CSM. Site hazards and risks are placed in the context of the risk-triplet (Kaplan and Garrick 1981) by defining scenarios (i.e., "what can go wrong" in terms of, for example, contaminant release, migration, and exposure) and consequences of exposure linked by the likelihood of exposure.

The site conditions considered to be hazardous (and perhaps leading to the site investigation) would be a good place to begin. Based on the *qualitative* results of Phase 1, *preliminary cleanup goals* for both the assessment and potential remedial actions are defined in accordance with stakeholder input to direct future research and work. These goals are high-level concepts representing desired outcomes of remedial actions. Examples include CERCLA evaluation criteria (e.g., protect human health and the environment, short-term effectiveness, volume reduction, etc.), *and are not* synonymous with specific target contaminant levels (e.g., preliminary remediation goals (USEPA 1991), soil screening levels (USEPA 1996), etc.) used to evaluate whether or not remedial actions are successful.

If sufficient information exists to determine that no remedial action is required, this "no action" decision is documented for public review and comment. However, a successful legal or regulatory challenge to the "no action" decision could restart the assessment. If remedial action cannot be ruled out, then an initial set of *contaminants of potential concern* (COPCs) is defined with corresponding transport pathways, exposure routes, and receptors (as used to define conceptual site models.)

Phase 2: Screening Quantitative Baseline and Remedial Alternative Risk Analysis

If remedial action for a site is not ruled out, then the purpose of Phase 2 (as illustrated in Figure 9 and Figure 10) is the *screening quantitative* baseline risk assessment (BRA) and evaluation of remedial alternatives. In general, the *qualitative* assessment from Phase 1 is used as the basis for the screening quantitative BRA. Site-specific data should be incorporated into the analysis when available; however, for *screening* purposes, more general data can be used if there is sufficient basis that the resulting risks will not be substantially underestimated. The initial step in Phase 2A (as shown in Figure 9) is to update the CSM from Phase 1 that forms the basis of the *screening quantitative* BRA.



Notes:
Each phase builds from the bottom to the top.
a. For example, preliminary acceptance goals could be preliminary remediation goals per CERCLA.
b. Three of the nine CERCLA evaluation criteria are used (40 CFR 300); others will be included if mandated by law.

Figure 9. Risk Assessment Framework—Detailed Phases 2A and 2B. Symbols are defined in the front matter.

Phase 2A: Screening Quantitative Baseline Risk Assessment

The steps in Phase 2A are *quantitative* analogues to those in Phase 1.The screening quantitative baseline risk assessment (BRA) provides estimated risks to receptors and uncertainties assuming no remedial actions are taken. The types of risk and uncertainty information developed at this early stage of the assessment should be commensurate with the importance of the remedial decision. For example, the Risk Assessment Guidance for Superfund (RAGS) (USEPA 1989) suggests only bounding risks be estimated[60]. However, best estimate and bounding risks should be provided at a minimum to provide an idea of the ranges of risks and uncertainties involved. No single estimate can adequately communicate the risk (Kaplan and Garrick 1981).

Phase 2A: Uncertainty Treatment and Value Judgments

The discussion of uncertainties in risk estimates can take a number of forms based on the importance of the remedial decision and complexity of the needed analysis. At a minimum, a *qualitative* evaluation of uncertainties in the parameters, models, scenarios, etc. used in the analysis and the likely impacts of these uncertainties on the risk estimates must be provided. Risk estimates without an analysis of uncertainties are of very limited practical usefulness. A probabilistic risk analysis (e.g., *Monte Carlo* simulation (Haldar and Mahadevan 2000; Ross 1990; Rugen and Callahan 1996; USEPA 1994; 1997)) may provide useful insights into the uncertainties and their potential impacts. However, a

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[60] "In the past, exposures generally were estimated for an average and an upper-bound exposure case, instead of a single exposure case (for both current and future land use) as recommended [in the RAGS]. The advantage of the two-case approach is that the resulting range of exposures provides some measure of the uncertainty surrounding these estimates. The disadvantage of this approach is that the upper-bound estimate of exposure may be above the range of possible exposures, whereas the average estimate is lower than exposures potentially experienced by much of the population" (USEPA 1989).

probabilistic analysis should be initiated with care because of the time and resources required and the fact that the information (e.g., parameters, probability distributions, etc.) needed to perform the analysis may be lacking or in question. The result might be diminished transparency and acceptance of the results (Goldstein 1995).

However, remedial decisions are made using risk assessments or other evaluations that involve value judgments. Value judgments are not restricted to selecting probability distributions for Monte Carlo analysis (even these may often be the most obvious). Value judgments may be hidden in regulatory language and default parameters mandated for risk analysis. For example, regulatory mandates concerning where, when and how long receptors are exposed to contaminants are issues where judgment is involved for deterministic or probabilistic analysis. Default parameters for exposure scenarios often represent bounding (e.g., 95th-percentile) values from a default set of data to provide a degree of "conservatism" in the analysis. However, selection of a bounding value may be arbitrary (even if usual and customary) and the default data may not apply to the population being evaluated (or, in fact, any members of the potentially impacted group).

The most important issue is one of communication and not the particulars of the specific type of risk analysis. At a minimum, explicit declaration of the value judgments and assumptions made and description of significant uncertainties and their likely impacts on the remedial decision are provided. A probabilistic analysis can provide very useful information when warranted by the importance of the decision. It is prudent that the risk assessor's toolbox include capabilities for (USEPA 2001)

- *sensitivity analyses* to determine those elements in the risk analysis that have the most influence on risk estimates, and

- *probabilistic analyses* to estimate the uncertainties in the risk estimates based upon those elements in the analysis found to likely have the largest influences.

The information gathering and validation exercises can be restricted to those elements[61] that are likely to have the most impact on the risk estimates.

Phase 2A: Screening Quantitative BRA Results and Preliminary Acceptance Goals

Quantitative estimates of risks to impacted receptors expected from potential exposures to site contaminants are the primary results from Phase 2A. If there are contaminants to which receptors might be exposed that exceed regulatory or legal limits (either based on model predictions or analytical analysis), then *preliminary acceptance goals* need to be established for site cleanup as indicated in Figure 9. These goals are related to but distinct from *cleanup goals*, the high-level concepts defined in Phase 1. A*cceptance goals* represent specific contaminant levels (e.g., preliminary remediation goals (USEPA 1991), soil screening levels (USEPA 1996), etc.) or agreed upon metrics that correspond to the *cleanup goals* (e.g., protective human health and the environment, etc.). The COPC list and preliminary acceptance goals are critical results of Phase 2A.

Phase 2B: Remedial Alternatives and Residual Risks

A set of *acceptable* remedial alternatives for managing risks at the buried waste site are defined using expert judgment and analysis incorporating stakeholder input. One method of selecting alternatives employs three of the nine CERCLA evaluation criteria (i.e., effectiveness, implementability, and cost) (CFR 1994) to screen out unacceptable

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[61] An example of how to assess the sensitivity of risk results to various input parameters is provided in EPA RAGS, Vol. III - Part A entitled "Process for Conducting Probabilistic Risk Assessment" (USEPA 2001).

remedial alternatives (Zitnik et al. 2002)[62]. Other criteria and alternatives are evaluated if required by law or agreement with regulators and stakeholders. The final set of remedial alternatives may be developed over several iterations of expert analysis and regulator and stakeholder interaction (Holdren et al. 2007).

After acceptable remedial alternatives have been selected, *residual risks* to the general public for each proposed alternative are estimated in Phase 2B in much the same way (and to a commensurate level of detail) as baseline risks were estimated in Phase 2A. Residual risks are those remaining after the remedial alternative has been applied to the site to provide a protective final state. The protective state for the contaminated site corresponds to the acceptance goals defined in Phase 2A.

However, it is possible that none of the proposed remedial alternatives can be used to clean up the site to a protective state based on the analysis in Phase 2B. Several potential courses of action are possible at this point:

- Select new "acceptable" remedial alternatives and repeat the Phase 2B residual risk evaluation or
- Perform a more detailed and accurate risk analysis (i.e., continue to Phase 3 of the risk analysis framework as indicated in Figure 4).

The *cleanup goals* defined in Phase 1 may also have to be updated based on new information.

Phase 2C: Qualitative Risk Analysis for Proposed Remedial Alternatives

The next step, Phase 2C as detailed in Figure 10, is the first that involves evaluating risks associated with the possible implementation of proposed remedial

[62] The sufficiency of this approach is not examined; however, the general method for defining acceptable alternatives is evaluated using the risk screening tool developed as part of this research.

alternatives. Two general types of risk (i.e., exposure and standard industrial) are initially evaluated *qualitatively.* That is, not only risks associated with exposures to hazardous chemicals and radiation must be considered, those accident or *standard industrial* risks (e.g., slips, trips, falls, etc.) that might be experienced by workers while performing remedial actions must also be considered in the evaluation process.

However, work at a DOE site involves some degree of accident risk that varies by the nature of the work being done. The effort to characterize standard industrial risks for remedial alternatives should be focused on the *additional* risks that would be experienced by a worker performing remedial actions.

Phase 2C: Task Lists and Management Flow Diagrams

The first step in evaluating the risks for a remedial alternative is the identification of the major process steps and component tasks comprising the alternative. This information can be readily encapsulated in *task lists* and *management flow diagrams* that outline the steps required to execute the remedial alternative. Task lists can be generated based on available information including Work Breakdown Structures (WBS) used to structure projects into modular and understandable elements for project management purposes. A management flow diagram for a remedial alternative consists of the general process steps that must be completed—and the order in which they are undertaken—to provide a protective final state for the buried wastes. The manner in which these diagrams are to appear will not be prescribed here; however, examples of acceptable diagrams are provided in Chapter IV, Brown et al. (2005), and Switzer et al. (2005).

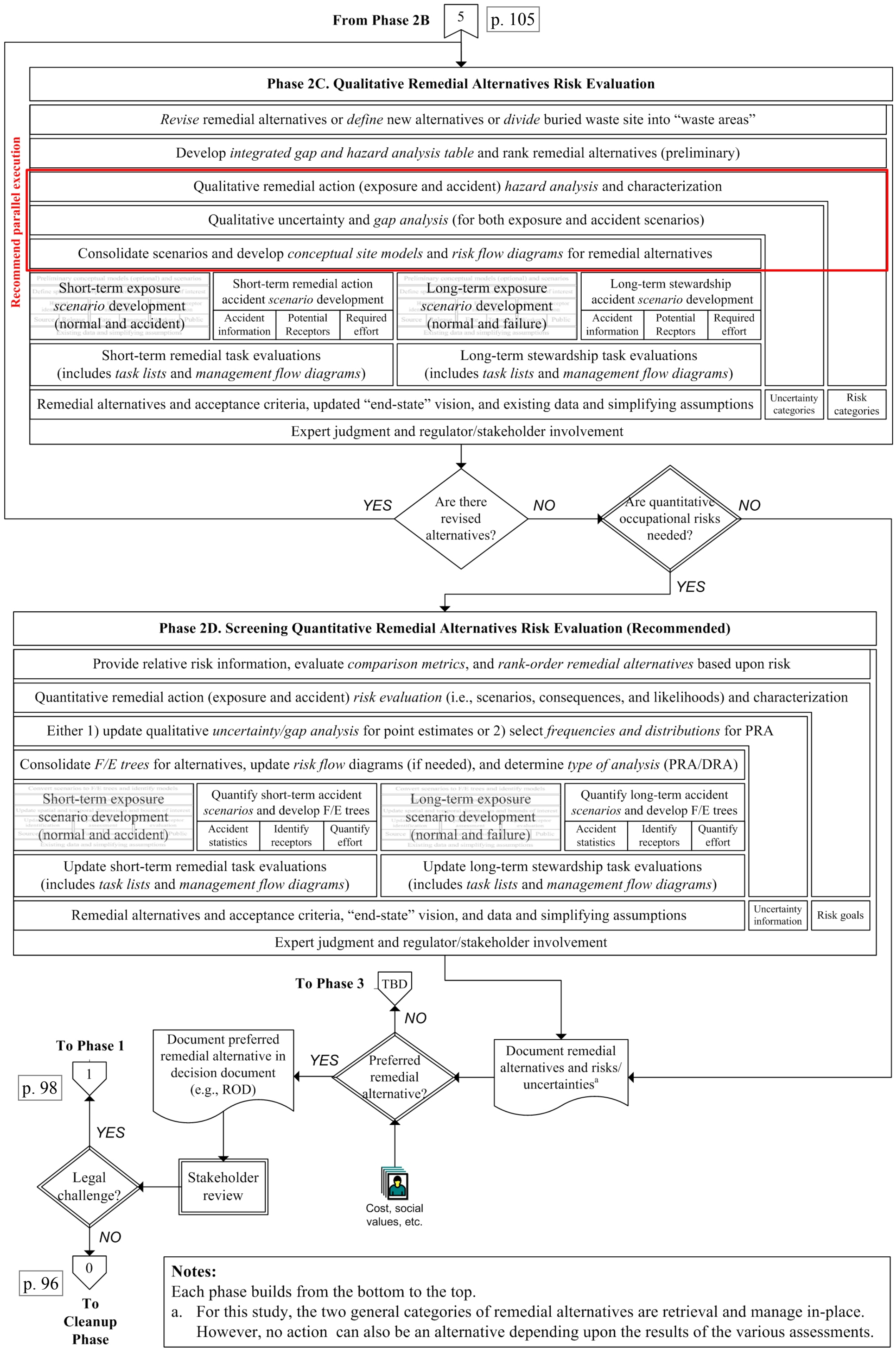

Figure 10. Risk Assessment Framework—Detailed Phases 2C and 2D. Symbols are defined in the front matter.

Phase 2C: Hazard and Gap Analyses for Remedial Alternatives

For each task, a *hazard analysis* is performed to identify, at a minimum, the *frequency*, *elements of risk*, *potential receptor*, *basis for characterizing risk*, and *contribution to overall risk*. An *uncertainty and information gap analysis* describing, at a minimum, the *key knowledge barriers*, *missing information*, and *uncertainties* is also performed. The exact representations of the hazard and uncertainty results are not prescribed here although tabulated results should be considered. Examples are available in Chapter IV, Appendix A and B, Brown et al. (2005), and Switzer et al. (2005). These *qualitative evaluations* form the foundation of the risk and uncertainty information required to begin an informed selection of remedial actions for a contaminated site.

Phase 2C: Comparison Metrics for Qualitative Risk Estimates

Risks and risk trade-offs among various proposed remedial alternatives are the primary information that will result from application of the framework and methodology. Many different types of risks (e.g., injury, fatality, latent cancer incidence, etc.) and risk trade-offs will be output by any substantive risk analysis for a complex buried waste site. Thus not only is it problematic to compare risk assessment results with non-risk factors (e.g., cost, social values, etc.) (Arrow 1951), it is also problematic to compare risks with different bases for proposed remedial alternatives. Guidelines for defining metrics are provided and a set of example metrics are defined for use in this research.

The metrics for evaluating qualitative risk results defined by Brown et al. (2005) are reproduced in Exhibit 1. A consistent set of definitions like those in Exhibit 1 allow reviewers to "mean the same thing" when generic terms such as "*low*" or "*high*" are used to describe risk. Risks are classified using expert judgment based on the risk-triplet

(Kaplan and Garrick 1981) to categorize the likelihood of the event occurring and the consequences of the event.

Although it is understood that the definitions in Exhibit 1 are one of many possible and there will not likely be unanimous agreement on any set of definitions, a common basis must be used for evaluating remedial alternatives. The categories used in this research are subject to revision as further knowledge is obtained or stakeholders included; however, any such set of categories must be defined, consistent, and visible to be of use in evaluating remedial alternatives.

In this research, a *probable* event is defined as something very likely to occur during task execution and a *possible* event is defined as something with a reasonable expectation of occurring. A *severe* consequence is defined as the loss of the ability to satisfy applicable and relevant design and performance criteria and protect human health and the environment. A *critical* consequence is defined as significantly degraded performance and ability to protect human health and the environment. These definitions are expanded in Exhibit 1. Combinations of *probable*/*critical*, *possible*/*severe* and *probable*/*severe* results provide sufficient likelihood of occurrence and impact to deem the corresponding hazards *high-risk*.

Exhibit 1. Definitions Used in Hazard Analysis (from Brown et al. (2005))

Task Frequency

Frequent: Occurs very often (e.g., more than once per quarter for long-duration tasks) or continuously.

Anticipated: Occurs several times (e.g., on the average of once per year) over the project lifetime or occurs infrequently but with long duration.

Occasional: Occurs sporadically or at a well-defined time (e.g., start-up or closure) or has a remote possibility of occurrence.

Unlikely: One can reasonably assume that this will not occur, but its occurrence is not impossible.

How likely is it? (Event Probability)

Probable: Very likely to occur (e.g., more than 50 times out of 100) during task execution.

Possible: Expected to occur (e.g., between 1 time out of 100 and 50 times out of 100) during task execution.

Unlikely: One can reasonably assume that this hazard will not transpire (e.g., less than one chance out of 100), but its occurrence is not impossible.

Consequence Severity[63]

Severe: Loss of ability to satisfy applicable and relevant design and performance criteria and protect human health (both worker and general public) and the environment (both on- and off-site). Likely to result in death or permanent disability including that from latent cancer effects to a large group of people (e.g., greater than 25 and greater than 5, respectively). Loss of major or safety-critical system or equipment. Major property or facility damage (e.g., greater than $1 million). Severe environmental damage (e.g., significant loss of protected or endangered species habitat). Severe security failure (e.g., loss of material with potential "dirty bomb" applicability)[64].

Critical: Significantly degraded performance versus applicable and relevant design and performance criteria and the ability to protect human health (both worker and the general public) and the environment (both on- and off-site). Likely to result in traumatic injury, illness, and/or disability requiring medical treatment to a moderate-sized group of people (e.g., 10 to 25 and 2 to 5 for injuries and deaths, respectively). Significantly degraded performance of major or safety-critical system or equipment. Significant property damage (of less than $1 million) requiring repairs and replacement and/or environmental damage requiring treatment. Breach of security (e.g., potential loss of control over material with potential "dirty bomb" applicability)[64].

Marginal: Some degraded performance versus applicable and relevant design and/or performance criteria or reduced ability to protect human health (both worker and the general public) as well as the environment (both on- and off-site). Minor damage to equipment, facilities, property, or environment that does not require immediate action. Injury or illness likely to result and will be limited to a small group of people (e.g., less than 10 and less than 2 for injuries and deaths, respectively). Minimal breach of or threat to security[64].

[63] Direct injuries and deaths are taken into account; psychological damage, economic loss, and stigma are not considered.

[64] It is recognized that this report primarily concerns human health; however, those tasks that involve risks to facilities and property, the environment, and site security will also be noted where appropriate.

One purpose of the risk assessment in this framework is to estimate the overall health risk associated with proposed alternatives for subsequent comparison and rank-ordering as input to the decision-making process. For the initial screening evaluation in this phase, the overall contribution to risk for a remedial task is selected using the risk-assessment matrix[65] in Table 8 based on expert judgment and the categories in Exhibit 1 (Brown et al. 2005). Table 8 provides an example of one such matrix although others can be defined based on alternative perspectives and desires.

Table 8. Example Risk-Assessment Matrix from Brown et al. (2005)

		How likely is it? (Event Probability)		
		Probable	**Possible**	**Unlikely**
Severity	**Severe**	High	Significant	Low
	Critical	Significant	Significant	Low
	Marginal	Low	Low	Low

For each task comprising a process step, a corresponding overall contribution to risk can be defined using the event probability and consequence severity and the matrix in Table 8. A method is needed to "roll up" the risk results for the tasks and process steps comprising a remedial alternative into a single *metric* representing the overall risk for the alternative (Brown et al. 2005). Because the risk results (i.e., *high*, *significant*, and *low* from Exhibit 1) describing the tasks comprising process steps are categorical variables, there is no simple, mathematical expression that can be derived to "roll up" the risk results. Instead the *criteria* in Exhibit 2 can be used to roll-up the risk information into a

[65] The primary reference for the hazard categorization is *Review of the Army's Technical Guides on Assessing and Managing Chemical Hazards to Deployed Personnel* (NAS 2004).

single overall-contribution-to-risk metric. The metrics indicating the overall contribution to risk for all tasks comprising a remedial alternative are determined and entered into the summary table for the remedial alternative considered.

Exhibit 2. Criteria for "Rolling-up" Risk Results (from Brown et al. (2005))

The following *criteria* can be used to roll-up the risk results into a single metric:

1. If a process step has at least one hazard that is considered *high* risk, then that process step is considered *high* risk in terms of its contribution to the overall risk.

   There may be a subsequent attempt to rank-order the high risk hazards; however, this will be by its very nature subjective because of the many assumptions already made. For example, one rank-ordering might place the potential for human health effects first (based upon numbers of people impacted, death versus injury, immediate versus latent, off-site versus on-site, etc.) followed by ecological risk, then national security and finally property damage. A different group may have different priorities. If there is not at least a majority agreement, then the individual rank-ordering will be given with a description of the drivers for their choices.

2. If a process step has only hazards that are considered *low* risk, then the contribution to overall risk from that process step is also *low* risk. This is akin to what should be done when considering cumulative radiological dose estimates.

3. If a process step has hazards that are considered as *significant* to overall risk, then the minimum risk contribution must also be *significant*. There is a *high* contribution to overall risk from a process step if ten (10) hazards in a process step are deemed *significant*. This is based upon the fact that the best information that we are likely to find for our analyses is on an order of magnitude. For reasons similar to those in Criterion #2 above, the number of low-risk hazards does not factor into this assessment.

Phase 2C: Comparison Metrics for Qualitative Uncertainty and Gap Results

Uncertainties and gaps in the information used to evaluate risks for a contaminated site are necessary additional dimensions to the analysis. The uncertainties and gaps in information for the tasks comprising a remedial alternative are identified as described in this methodology. A classification scheme analogous to that for risks is used to describe both the importance and extent of the uncertainty or missing information.

Exhibit 3. Definitions Used in Uncertainty and Gap Analysis (from Brown et al. (2005))

How Important (is the Gap)?

Critical: Lack of this piece of knowledge is sufficient to provide a high degree of uncertainty in the ability to assess the threat to human health (both worker and the general public), the environment (both on-site and off-site), and/or security; i.e., result in a critical or severe hazard (as defined in Exhibit 1).

Important: Possession of this knowledge is important to the ability to assess the threat to human health (both worker and the general public), the environment (both on-site and off-site), and/or security. Other information must be lacking to the ability to assess the threat to human health and the environment.

Inconsequential: This knowledge may have localized significance to non-safety-related activities (including routine maintenance, repair, etc.).

How large is the Gap? (Magnitude of the Gap or Level of Knowledge)

Large: Little is known or can be reasonably inferred concerning this piece of information (from other sources of information).

Intermediate: Incomplete information is available concerning this piece of information or can only be inferred from other data not necessarily directly related to the missing piece of information.

Small: Complete or nearly complete information is available concerning this piece of information or an adequate, well-known analogue can be established.

The example classification scheme developed for this research is provided in Exhibit 3 (Brown et al. 2005). It is unlikely that there will be unanimity in accepting any set of definitions; however, any alternative must be consistent, defensible, and visible to both the assessor and stakeholders. From previous experience, it is more important that the most significant gaps in information be summarized than "rolling-up" the various uncertainties and information gaps into a single metric for each remedial alternative (Brown et al. 2005; Switzer et al. 2005). Ultimately, definitions and metrics should be based on combined *risk* and *uncertainty* information for the tasks comprising proposed remedial alternatives.

Phase 2C: Risk Flow Diagrams and Integrated Summary Tables

Additional information is crucial to lending transparency and understandability to the hazard and uncertainty analysis information as it relates to selecting remedial actions buried waste sites[66]. The first is the development of *risk flow diagrams.* These diagrams, based on the results of the hazard analyses, help form the foundation for life-cycle risk assessment and comparison by indicating both the sequence of and relative health risk from the remedial action steps[67].

The exact specifications of the risk flow diagrams are not prescribed here, and two versions have been used. The first version indicated the sequence of and potential interactions among only those activities with potential to pose significant human health risks and incorporate conceptual site models describing the hazards associated with the remedial alternative (Brown et al. 2005; Switzer et al. 2005). A revised version of the risk flow diagram was developed for this research based on the management flow diagram where each process step is shaded to indicate the health risk posed; examples are provided in Chapter IV. As illustrated in Figure 10, risk flow diagrams should be developed in tandem with the hazard and gap analyses to reduce duplication of effort.

Phase 2C: Conceptual Site Models for Remedial Alternatives

The baseline conceptual site model (CSM) developed during Phase 1 (e.g., Figure 8) may be revised, if necessary, based on new information. Conceptual site models are

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[66] The information described in this Chapter was developed for the Idaho Site high-level waste (HLW) calcined bin sets and Subsurface Disposal Area (SDA) (Brown et al. 2005; Switzer et al. 2005) and was presented to the Idaho Site Citizen's Advisory Board (CAB) in July 2005. The CAB endorsed the reports and strongly recommended to the DOE that the provisions of the reports be followed. The CAB recommendations (#123 and #124) are available at http://www.cresp.org/ (accessed March 14, 2008).

[67] Schedule risks can also be described but should be illustrated in separate diagrams.

also developed for both the exposure *and standard industrial* hazards posed by remedial alternatives. The integrated CSM in Figure 11 is unique to this research and provides critical exposure and accident risk information for remedial alternatives in a manner consistent[68] with existing CSMs typically defined for only baseline and desired "end-state" conditions (ASTM 1995; USDOE 2003). The new integrated CSM not only links contaminant sources to potential human receptors to illustrate exposure risks but also the additional accident hazards (e.g., traumatic injury, explosion, fire, criticality, etc.) to those receptors potentially at risk.

A post-closure CSM is also developed for each remedial alternative. A generic CSM is provided in Figure 12 for the likely final state of the buried waste site analogous to the baseline CSM in Figure 8. The CSM in Figure 12 describes the minimum set of barriers, institutional controls, etc. that will likely be needed to leave a buried waste site in a protective state. The barriers indicated (i.e., surface barrier and land-use restrictions) would likely be supplemented by additional engineering and institutional controls. Figure 8 , Figure 11, and Figure 12 provide generic CSMs that can be used to develop CSMs for specific sites that describe risks from the current through post-closure stages as illustrated in Chapter IV.

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[68] The appropriate narrative would also be added as illustrated for the baseline conceptual site model (CSM) in Figure 8.

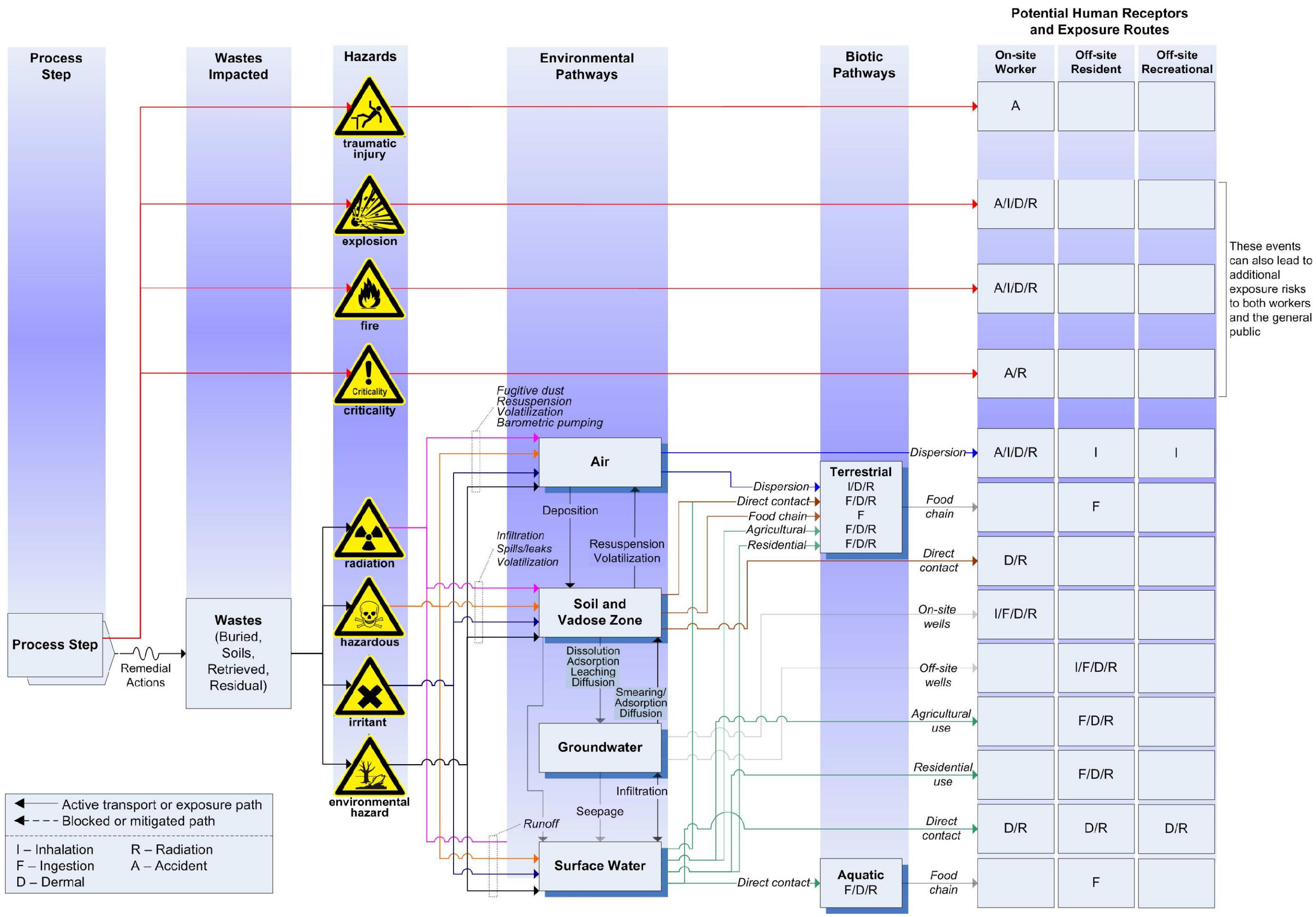

Figure 11. Integrated conceptual site model (CSM) describing exposure and standard industrial risks **during remedial activities** modeled on those expected for the SDA. The hazards will vary based on the buried waste site.

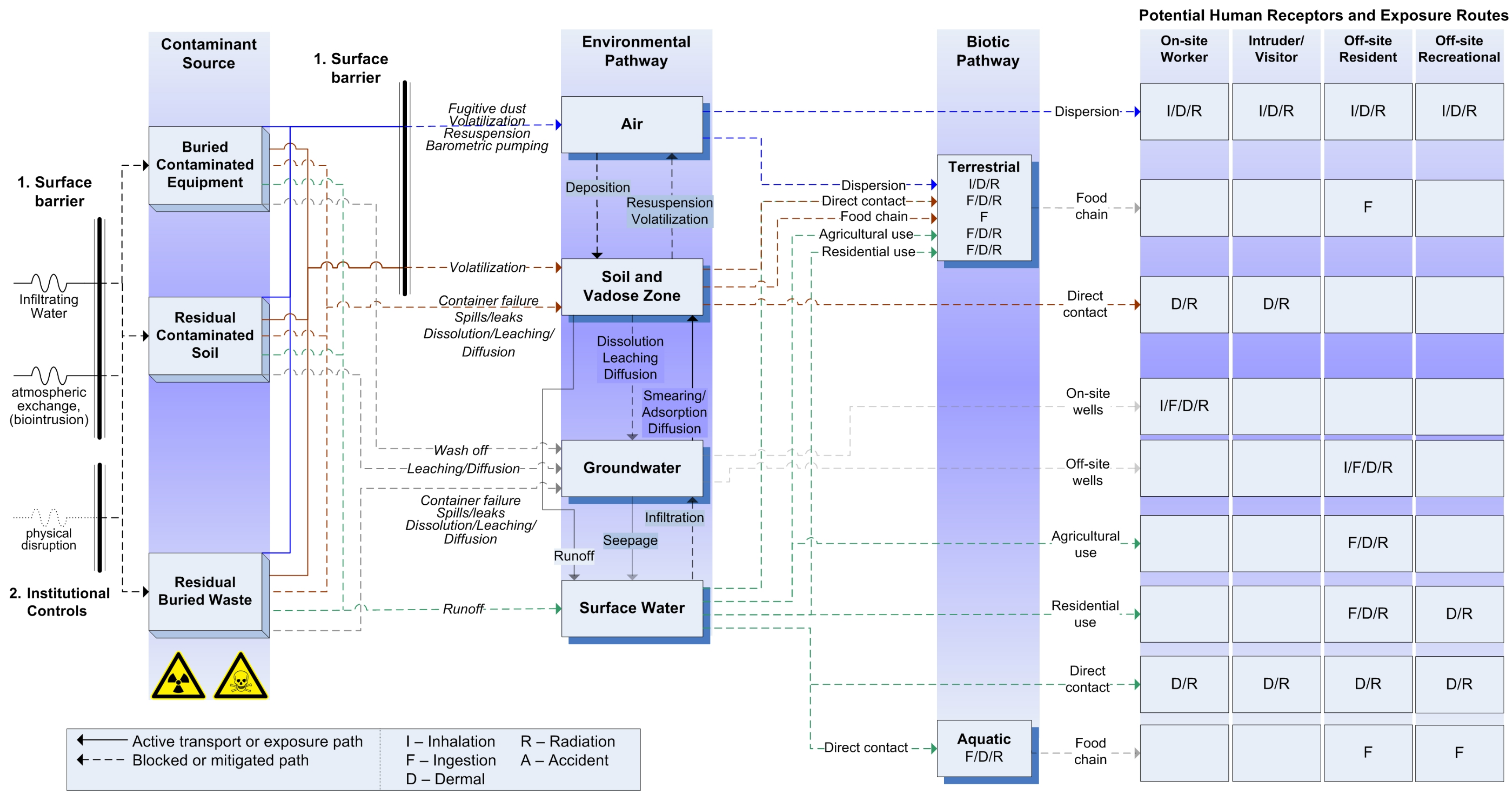

Figure 12. Simplified, generic conceptual site model (CSM) representing the minimum-protective (post-closure) residual risks **after remedial activities have been completed**.

Narrative for Figure 12: Simplified, Generic Conceptual Site Model (CSM) for Post-Closure Conditions

A brief description of any residual high-risk residual chemicals (e.g., volatile organic compounds, nitrates, etc.) and radionuclides (e.g., short-lived fission products, long-lived transuranic (TRU) elements, etc.) should be provided here.

The barriers or steps taken to mitigate impacts for the CSM in Figure 12 are provided here:

1. A surface barrier (e.g., evapotranspiration, RCRA Subtitle 'C', etc.) has been installed on the site to prevent infiltration of water and to control biotic intrusion into the waste areas.
2. Land-use controls are in effect to prevent intrusion into site areas.

Other pertinent information is supplied here. For example, if there is already extensive contamination in the environment from the buried waste site, then a pathway may exist from the burial site to a receptor. In this case, the only potential "barriers" that exist are decay, dispersion, and dilution for the radionuclides and dispersion and dilution for the inorganic constituents and heavy metals.

Phase 2C: Hazard and Uncertainty Analyses for Remedial Alternatives

Significant *hazards and corresponding health risks* from potential exposures to hazardous and radioactive contaminants to human receptors resulting from the proposed remedial activities are identified in Phase 2. However, *standard industrial hazards and corresponding risks* are also identified for proposed remedial alternatives as well as *significant uncertainties and gaps in knowledge* and their potential impacts on the risk estimates. Examination of *standard industrial risks*, often neglected in many risk assessments[69], is very important to life-cycle considerations because these risks (e.g., from slips, falls, traffic accidents, etc.) may dominate the risks posed by potential remedial alternatives (Applegate and Wesloh 1998; Gerrard 2002) and may dominate the overall risks of dispositioning buried wastes.

An *integrated hazard and gap summary* is developed for the remedial alternatives illustrating the most important potential risks and uncertainties including information gaps. A *breakdown of the risks* associated with the proposed remedial alternative as they relate to types of risks (e.g., chemical exposure, radiation, traumatic injury, etc.) and potential general public and workplace receptors must be provided for comparison purposes. The *comparison metrics*, which will be defined as part of this research, should be evaluated for each remedial alternative for subsequent comparisons.

Phase 2C: Life-cycle Considerations

An important distinction between the framework developed in this research and others is the explicit inclusion of risks associated with the ultimate disposition of any

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[69] This omission may be less of neglecting these risks and more assuming the standard industrial risk are subsumed in those related to OSHA guidelines.

wastes that are retrieved (constituting a risk transfer) and long-term stewardship activities (e.g., monitoring, maintenance, etc.). Often only the potential exposure risks to the general public from site contaminants are factored into the assessment and remedial action decisions. However, worker risks from not only exposure to contaminants but also accidents may dominate the likely risks, especially if retrieval activities are employed. Furthermore, risks from transporting wastes elsewhere and their ultimate disposal may be significant from a long-term perspective.

There are two ways of approaching the evaluation of the risks associated with long-term actions. One is to explicitly evaluate the risks from all long-term actions for all remedial alternatives into analysis. On the other hand, because even “no action” alternatives require some form of institutional and engineering controls and long-term monitoring and maintenance, it is proposed that the analysis be simplified by examining differences in the necessary controls and long-term actions dictated by the remedial alternatives. That is, mention should be made of the institutional and engineering controls and long-term monitoring and maintenance required for an alternative (including “no action”); however, *analysis of the risks and uncertainties* associated with *institutional and engineering controls* and *long-term monitoring and maintenance* activities can be restricted to *differences* between remedial and “no action” alternatives to simplify the analysis and reduce the resources necessary to perform the analysis.

If wastes are to be retrieved from the buried waste site for disposal elsewhere, there would be potential risks associated with the final disposal site. For an engineered geologic repository, an analysis of the features, events, and processes (FEP) that might impact performance would be employed (Swift et al. 1999). The wastes retrieved from

DOE buried sites would not likely be disposed in a geologic repository because they are unlikely to be high-level in nature; however, the risk analysis for the final disposal of retrieved waste can leverage off the FEP evaluation process or existing FEP information.

Phase 2: Final Considerations and Decision Making

It is possible, if not likely, that the comprehensive *qualitative* evaluation of exposure and occupation risks for proposed remedial alternatives in Phase 2C will suggest that some alternatives are indeed not worthy of continued evaluation or should be revised (with stakeholder input). One obvious revision is the subdivision of the buried waste site into *waste areas* based on wastes forms, risk drivers, retrievability, etc. that can be managed individually (although whose risks would be evaluated in an integrated fashion). The subdivision might cause some remedial alternatives to be omitted from further consideration, the hazards and corresponding risks associated with revised alternatives to be evaluated, and/or different remedial actions to be considered for the *waste areas*. The risk analysis may have to revert back to the beginning for each *waste area*; however, the risks from all such waste areas must be considered simultaneously to assure that the remedial decision has the appropriate risk basis.

Rare cases can be envisioned that only a single remedial alternative will either be *dominant* (e.g., all risks are lower than for any other alternative) while at the same time being agreeable to regulators and stakeholders. For this rare case, a *quantitative evaluation of remedial risks and uncertainties* would be unnecessary and the analysis would proceed to closure as illustrated in Figure 5. However, it is unlikely that any such obvious remedial alternative will be evident from the *qualitative analysis* of proposed remedial alternatives.

Phase 2D: Screening Quantitative Analysis of Remedial Risks

In the majority of circumstances, a *screening quantitative analysis of remedial risks* will be needed to provide the comprehensive risk information needed to make an informed remedial decision. The quantitative screening remedial risk evaluation in Phase 2D (as illustrated in Figure 10) will follow the basic framework defined in Phase 2C for the qualitative evaluation of remedial risks. Site-specific information will be utilized wherever and whenever possible to estimate human health risks and corresponding uncertainties for both exposure and standard industrial hazards. The *comparison metrics* that were defined based on qualitative information in Phase 2C are reevaluated based on the quantitative results from Phase 2D.

Phase 2D: Comparison Metrics for Quantitative Risk Estimates

The classification scheme used to evaluate event likelihood and consequence severity for the tasks comprising remedial alternatives using qualitative risk results is provided in Exhibit 1 and Exhibit 2. This classification scheme must be modified for application to the quantitative risks and uncertainties generated in Phase 2D.

The original Food and Drug Administration (FDA) *de minimus* risk of $1x10^{-8}$ in 1961 corresponded to one person in the entire U.S. population at the time (Graham 1995). This *de minimus* risk was later changed to $1x10^{-6}$ and was adopted by many agencies including the EPA. A similar notion is adopted in this research where the entire population that might be impacted is used. For example, the impacted population is likely to vary between a few and the over 8,500 workers at the Idaho Site. A population value of 100 workers is selected as a reasonable basis for worker risk classification. For off-site activities (i.e., shipment of the wastes to WIPP), a population of 1,000 is selected. The resulting definitions provided in Exhibit 4 allow the risk consequences to be classified using the same categories used in Exhibit 1 and Exhibit 3.

The event probability classifications of *probable* (> 50% chance), *possible* (between 1 and 50% chance), and *unlikely* (< 1% chance) are retained from the classification in Exhibit 3 used to classify risks based on expert opinion. These categories allow use of the same risk-assessment matrix defined in Table 8 (Brown et al. 2005).

However, *risk* information is one of many inputs to a transparent and informed decision-making process. Other factors (e.g., cost, social values, ecological factors, etc.) must also be factored into the remedial alternative decision. All such factors should be considered to determine which remedial alternative is selected for action. If a decision (including the "no action" decision) can be made based upon the screening quantitative results, then it is documented for review and comment. Otherwise, a much more detailed and accurate risk analysis (i.e., Phase 3) is warranted as illustrated in Figure 4.

Exhibit 4. Definitions Used in Quantitative Risk Analysis

The *risk classification* proceeds as follows where *N* is the population:

- If fatality risk is greater than 1 in *N* workers, the risk is classified as *severe*.
- If injury risk is greater than 1 in *N* workers, the risk is classified as *critical*.
- Otherwise, the risk is classified as *marginal*.

For on-site activities, the population is assumed to be 100 workers. For off-site activities (e.g., shipment of the wastes to WIPP), a population of 1,000 was assumed.

Phase 3: Detailed Quantitative Baseline and Remedial Action Risk Analysis

When no remedial alternative can be agreed upon for a contaminated buried waste site based upon the qualitative and quantitative risk analyses, then a more detailed and accurate analysis of the risks, risk trade-offs, and uncertainties of the proposed (or revised) alternatives is necessary. In general, this analysis phase follows that outlined for Phase 2 of the analysis framework illustrated in Figure 9 and Figure 10.

The most critical aspects of Phase 3 is to first reexamine the goals and criteria on which the remedial action decision was based to see if a decision *could* be made in light of the requirement and limitations on current knowledge as well as what can be gained by additional site characterization. Available information including the models used and the data needed for analyzing the risks and uncertainties for disposition of the site wastes is evaluated in terms of those parameters that most likely drive the risks for baseline conditions and remedial actions.

Additional site characterization and more accurate models or suites of models may be required to provide the increased accuracy needed by decision makers. Additional characterization efforts will likely be required to provide the site-specific information needed to reduce uncertainty in the parameters to provide more accurate predictions. The assessment of uncertainty may have to be separated into a more rigorous evaluation of the lack of knowledge (often denoted "uncertainty" in human health risk assessments) versus true heterogeneity in the system and receptor descriptions (often denoted "variability"). Often two-dimensional Monte Carlo simulations can provide a better representation of the uncertainties in the system if sufficient information is available to separate these elements (Burmaster 1997; Burmaster and Bloomfield 1996). The consideration of the likelihood of exposure based upon future land-use and receptor studies using more formal techniques including the use of fault and event trees may be warranted. Early remedial actions, that would be effective from timing, cost, and effectiveness perspectives, may be required to address any obvious high-risk hazards at any stage in the evaluation process.

Summary of the Methodology for Framework Implementation

The framework for the life-cycle assessment of the risks associated with the disposition of buried wastes and the methodology for applying the framework were described in the previous sections. The methodology is comprised of the instructions needed to assess risks and the information (e.g., risks, metrics, diagrams, etc.) needed to communicate the life-cycle risk information for decision-making purposes. Furthermore, a rational approach is needed for managing uncertainties and missing information in such a way that meaningful risks and risk trade-offs can be compared. Such an approach is developed as part of this research.

Some or all of the following types of information are required during the life-cycle risk analysis of a buried waste site and its potential disposition:

- Preliminary *site evaluation* including available information on contaminants, releases, modes of transport, hazards, site characteristics, potential on- and off-site human receptors, etc.
- *Baseline conceptual site model* linking contaminant sources to receptors.
- *Baseline human health risk assessment* resulting in lists of *contaminants of potential concern* and other potential hazards to human health.
- *Cleanup goals*[70] and comparison *metrics*.

[70] *Cleanup goals* are high-level concepts representing the desired outcomes of remedial actions. Examples include CERCLA evaluation criteria (e.g., protect human health and the environment, short-term effectiveness, etc.). Conceptual goals should not be confused with *acceptance goals* representing target, post-remedial contaminant of concern levels (e.g., preliminary remediation goals (USEPA 1991), soil screening levels (USEPA 1996), etc.) or other *metrics* that correspond to the *cleanup goals*.

- Acceptable *remedial alternatives* (based, for example, on a screening analysis using effectiveness, implementability, and cost per the National Oil and Hazardous Substances Pollution Contingency Plan, 40 CFR 300 (1994)) and corresponding *residual risk* estimates. For each remedial alternative, the following information is needed:

    - *Task list* and corresponding *management flow diagram* outlining the steps required to execute the remedial alternative.

    - *Conceptual site models* relating the natures of the hazards and risks during and after remedial activities to potentially impacted receptors.

    - *Hazard analysis* identifying (for each process step) the task frequency, elements of risk, potentially impacted population, basis for characterizing the risk, and contribution of the remedial task to overall risk.

    - *Risk flow diagram* indicating the sequence of remedial and stewardship activities with potential to pose significant human health risks. *Conceptual site models* for the remedial actions and final protective states should also be developed.

    - *Gap analysis* describing the key knowledge barriers, missing information, variabilities, and uncertainties involved in assessing risks for the remedial alternative.

    - *Integrated hazard and gap analysis* summarizing the most important potential risks and information gaps for the remedial alternative. *Comparison metrics* should also be evaluated.

- *Life-cycle risk breakdown* indicating the life-cycle risks for proposed remedial alternatives as they relate to types of risks and potential receptors. Indications of the uncertainties associated with the risks are included.

The above components of the risk analysis, many of which would be initially developed during the *qualitative phases* of the analysis, help focus the assessment during subsequent phases and provide a basis for comparison of potential remedial alternatives.

Additional Metrics for Remedial Alternative Comparison

Metrics were developed for the risk and uncertainty results from both qualitative and quantitative analyses. Standard quantitative "metrics" (e.g., latent cancer incidence, mortality, etc.) for exposures to hazardous chemicals and radiation were used. The initial consideration is to evaluate the possibility of reducing the number of quantitative metrics that must be compared without significant loss of information.

For example, because the effective dose and the mortality and morbidity risks corresponding to radiation exposures are all activity-driven, there is a good chance of that risk and dose estimates might be highly enough correlated over time to focus on one or the other for comparison purposes (ISCORS 2002). Under some circumstances, risks from radiation exposures might be compared to carcinogenic risks or those from standard industrial accidents with appropriate caveats. Such comparisons illustrate the benefit of performing the quantitative assessment and differences in results for different types of risks should be evaluated further to determine the reason for the difference and to add transparency to the decision-making process.

The development of a rigorous, normative basis for life-cycle risk comparison is not possible (Arrow 1951); however, *meaningful* comparisons of life-cycle risk results using well-conceived metrics can be made in the appropriate context. The intent of risk assessment process should not be to characterize all risks fully and attempt to eliminate all uncertainty; the intent should be to characterize risks and uncertainties to the point that a decision can be made (when taking risk and non-risk factors into account). The initial set of metrics proposed in this research will help in this regard.

Approach to Managing Uncertainties and Missing Information

The uncertainties in the risk results are necessary additional dimensions to provide context and transparency to the analysis. Thus the management of uncertainties is critical to providing meaningful life-cycle risk estimates for proposed remedial alternatives. Furthermore, any transparent evaluation of the risks associated with remedial actions requires the explicit declaration of the value judgments and assumptions made by the risk assessor as well as the significant sources of uncertainty and the likely impact of the uncertainties on the risk estimates. To prevent confusion, *from this point forward* uncertainty will denote a lack of information; whereas, variability will be used to describe true heterogeneity.

Three primary ideas form the foundation for management of uncertainty, missing information, and variability in the framework and methodology. The first idea is that the intent of the risk assessment process should not be to eliminate all uncertainties from the analysis or fully characterize all variability. Uncertainty and variability will remain part of any risk assessment (including deterministic one) and *cannot* be removed from the analysis nor ignored. One must live with both. These facts do not mean that meaningful estimates of risks and comparisons of these risks cannot be made.

A better approach to uncertainty management than an ill-fated attempt to arbitrarily reduce all uncertainties or fully characterize all variability is a consistent approach to classifying, estimating, and reducing uncertainties based upon their potential impact on risk estimates and their comparisons. The level of effort expended in identifying, quantifying, and reducing uncertainties and missing information should be commensurate with the potential impact of better information on the ability to compare resulting life-cycle risk estimates for potential remedial alternatives.

The second important idea to uncertainty management is that both uncertainty and variability will be encountered during the risk analysis process and distinctions between these may have to be taken into account. A clear distinction is needed to convey the relative importance of lack of knowledge versus heterogeneity when describing risk results to interested parties. If for example, the decision is made to confound uncertainty and variability for certain parameters while performing a screening risk analysis, this fact should be communicated. Several techniques have been developed for managing uncertainty and variability when using models to estimate exposure and risk (Bogen 1990; Burmaster 1997; Cullen and Frey 1999; Frey 1992; Hertwich et al. 1999; Hoffman and Hammonds 1994; Rai and Kreski 1998).

In probabilistic human health risk analyses, a two-dimensional Monte Carlo simulation can be employed where variability is treated by selecting "individuals" from an appropriate *frequency* distribution and then for each "individual" random variates are selected from an appropriate *probability* distribution and propagating these through the model to obtain output values. The resulting distribution of output values then appropriately reflects the impacts of both variability and uncertainty (within the limits of defining appropriate distributions). The tails of the output distribution (and likely the percentile values of interest) will not be reliable if uncertainty is not treated distinctly from variability.

Although it is understood that not every risk assessment requires the sophistication of a two-dimensional Monte Carlo simulation or a probabilistic human health risk analysis, the distinction between variability and uncertainty is important to maintain. The approach to managing uncertainty and variability in the proposed framework and methodology maintains this important distinction. For a screening risk analysis, it is appropriate to select characteristic (and not worst-case) individuals based upon exposure scenarios to capture the variability in human exposure, dose, and risk.

The third and perhaps most important idea pertaining to the management of uncertainty is "not to diminish the role of uncertainty, but rather to properly and fully reflect it in [the] information that decision makers will be asked to consider" (NAS 2005). That is, one purpose of the framework and methodology developed in this research is to assure that the uncertainties and missing information important to the comparison of life-cycle risks for remedial alternatives are identified and described in a transparent and forthright manner. This transparency is needed to assure that stakeholders are as comfortable as possible with the bases for the risk estimates. The sophistication in dealing with uncertainties and missing information should be commensurate with the importance and complexity of the problem and analysis. For example, in the tiered risk analysis proposed in this proposal, the requirements of the uncertainty and variability analyses and information requirements will change from the screening phase (i.e., qualitative descriptions) to the detailed analysis phase (possibly quantitative descriptions employing two-dimensional Monte Carlo simulation techniques).

Screening Quantitative Health Risk Assessment

The exposure and standard industrial risks associated with the buried wastes and those remedial actions that may be required to disposition buried wastes must be considered in the risk assessment process to provide the information needed to make a fair and equitable decision. Numerous methodologies and software tools are available that consider various aspects of the risk assessment process in varying degrees of detail; however, none integrate all the aspects needed to address the life-cycle risks posed by the disposition of DOE buried wastes. In fact, few methodologies incorporate both exposure *and* standard industrial risks. For example, the DOE Health Risk Evaluation

Methodology (Blaylock et al. 1995) can be used to assess both exposure and standard industrial risks *to workers* performing environmental restoration and waste management activities at DOE sites but does not address concomitant risks to the general public.

One screening risk analysis tool that integrates both exposure and accident risks associated with the disposition of wastes is the Environment, Safety, and Health Risk Assessment Program (ESHRAP) (Eide and Nitschke 2002; Eide et al. 2002; Eide and Wierman 2003). ESHRAP is a software tool that can be used to estimate worker and general public risks from exposures to radioactive and hazardous chemicals and accident risks from activities associated with cleanup and management activities and typically covers the entire waste management program (Eide and Wierman 2003). The *point estimate* risk values generated are meant to be “best estimate” rather than bounding or conservative. Although an excellent initial screening tool, ESHRAP contains neither sensitivity nor probabilistic analysis facilities providing no ability to assess uncertainties nor defend risk estimates to stakeholders. Something more is needed.

An adequate screening risk analysis tool is needed that can provide defensible life-cycle risk estimates for both baseline and remedial options as input to the risk-informed decision-making process. The screening tool should be based on as-simple-as-possible models, simplifying assumptions (intended to produce higher than expected risk predictions), default parameters, and readily available site information. To estimate baseline and residual risks, the models in the screening risk tool should estimate

- infiltration into the burial site,
- contaminant releases (i.e., surface wash, dissolution, and diffusion) (Anderson and Becker 2006; Sullivan 2006) and transport fluxes (i.e., advection, diffusion, resuspension, barometric pumping, plant- and animal-induced transport, etc.) out of the burial site,

- contaminant fluxes through possible exposure media (e.g., air, vadose zone, saturated zone, surface water, etc.),
- potential human exposure (or "intake") to contaminants, and
- intake-to-dose or intake-to-risk conversion[71].

During potential remedial activities, there is also likely a greater potential for exposure to both hazardous and radioactive contaminants and standard industrial accidents; these effects must also evaluated using the same tool.

Because no screening risk tool was identified that provides an adequate *quantitative assessment* of the life-cycle risks and uncertainties associated with the disposition of buried wastes, a proof-of-concept tool was developed in the GoldSim simulation software (GTG 2005a; b; c). Because of the complexities of actual buried waste sites, a conceptual buried waste model is defined in Chapter V. Simplified models and conservative assumptions formed the basis of the proof-of-concept screening risk analysis tool developed in GoldSim as described in Chapter VI. Because the screening risk tool is based upon simplified models and conservative assumptions, screening and probabilistic analyses are very efficient for such a complex system.

Prototypic Site Selection

The framework and methodology developed in this research is applied to two prototypic sites in Chapter IV to illustrate the effectiveness and flexibility of the approach to promote consistency in planning for the disposition of buried waste across the DOE

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[71] For any complicated buried waste site, there are likely to be multiple contaminants of concern that impact potential receptors via multiple pathways. The framework and methodology will consider these cumulative risks in terms of how risks from different sources via different pathways over varying time frames will be combined; however, a detailed cumulative analysis including synergistic and antagonistic dose-to-effect analyses are outside the scope of this research.

Complex. An examination of existing literature and previous experience indicates that the Oak Ridge Reservation (ORR) Bear Creek Valley Burial Grounds (BCBG) and the Idaho Site Subsurface Disposal Area (SDA) are excellent prototype sites. These sites bracket the types of contaminants, hazards, and conditions that are expected from the various DOE sites where the decision between managing the waste in-place and retrieving the wastes for disposal elsewhere will be made.

Remedial investigations (under CERCLA) have been completed for both prototype sites and baseline human health risks have been estimated for both (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002; SAIC 1996a; b). Remedial risks for proposed remedial actions have also been estimated for the SDA (Schofield 2002; Zitnik et al. 2002). The results from application of the proposed framework will be compared to those obtained from the corresponding baseline human health risk assessments and any available remedial evaluations for verification purposes.

Conclusions

A general life-cycle risk analysis framework and methodology have been developed for assessing the risks and risk trade-offs associated with the disposition of buried wastes generated from past DOE nuclear material production. The conceptual framework outlines graphically the general process for the rational and transparent comparison of the risks and risk trade-offs involved with either managing buried wastes in-place or retrieving the wastes for treatment and disposal elsewhere. The methodology (including instructions, conceptual models, diagrams, metrics, etc.) is defined describing the process of applying the framework to DOE buried waste sites. The resulting framework and methodology is applied to two prototype sites in the following chapters to demonstrate the value of the approach in evaluating the risks for DOE buried waste sites.

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CHAPTER IV

APPLICATION OF THE LIFE-CYCLE RISK ANALYSIS FRAMEWORK TO TWO DEPARTMENT OF ENERGY (DOE) BURIED WASTE SITES

The framework defined in Chapter III was applied to two prototypic sites to illustrate the effectiveness and flexibility of the approach to promote consistency and transparency in planning for the disposition of buried waste sites across the Department of Energy (DOE) Complex. An examination of literature and previous experience indicated that the Oak Ridge Reservation (ORR) Bear Creek Valley Burial Grounds (BCBG) and Idaho Site Subsurface Disposal Area (SDA) were excellent prototype sites for framework evaluation. These sites bracket the types of contaminants, hazards, and conditions expected from the DOE sites where a decision must be made between either managing wastes in-place or retrieving the wastes for treatment and disposal elsewhere.

Remedial investigations (performed under the Comprehensive Environmental Response, Compensation, and Liability Act or CERCLA) including *baseline* human health risk assessments have been completed for both the SDA (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) and the BCBG (SAIC 1993; 1996e). Furthermore, short-term risks for proposed remedial actions were estimated for the SDA as part of the CERCLA feasibility study (Schofield 2002; Zitnik et al. 2002). The results from application of the framework will be compared to those obtained from the corresponding baseline health risk assessments and available remedial evaluations for verification purposes.

Prototype Site Descriptions

The prototype sites selected for framework evaluation are the Idaho Site Subsurface Disposal Area (SDA) and the Oak Ridge National Laboratory (ORNL) Bear Creek Burial Grounds (BCBG) in Tennessee. These sites bracket the types of contaminants, hazards, and conditions that are expected from relevant DOE sites. Application of the framework to these sites will demonstrate the effectiveness and flexibility of the approach to risk analysis for remedial decisions concerning managing buried wastes in-place or retrieving these wastes for disposal elsewhere. The primary distinguishing features between these sites are:

- *Climate*: The SDA climate is arid to semi-arid and that for the BCBG is humid.
- *Vadose zone*: The SDA vadose zone is very deep; whereas, a substantial portion of the of the BCBG is periodically inundated.
- *Temporal nature of risks*: The SDA risks tend to be long-term primarily via the groundwater pathway and the BCBG risks tend to be short-term via primarily the surface water pathway.
- *Nature of buried wastes*: The SDA contains volatile organic compounds (VOCs), fission products, and TRU wastes; whereas, the BCBG contains VOCs, uranium-contaminated wastes, pyrophoric uranium fines, chips, and cuttings, and unstable materials including reactive, pyrophoric, and explosive chemicals.

A brief description is provided in this chapter for each prototype site. The appropriate remedial investigation reports (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002; SAIC 1996a; b; c; d; e; f) should be referred to for more detailed information concerning the prototype sites.

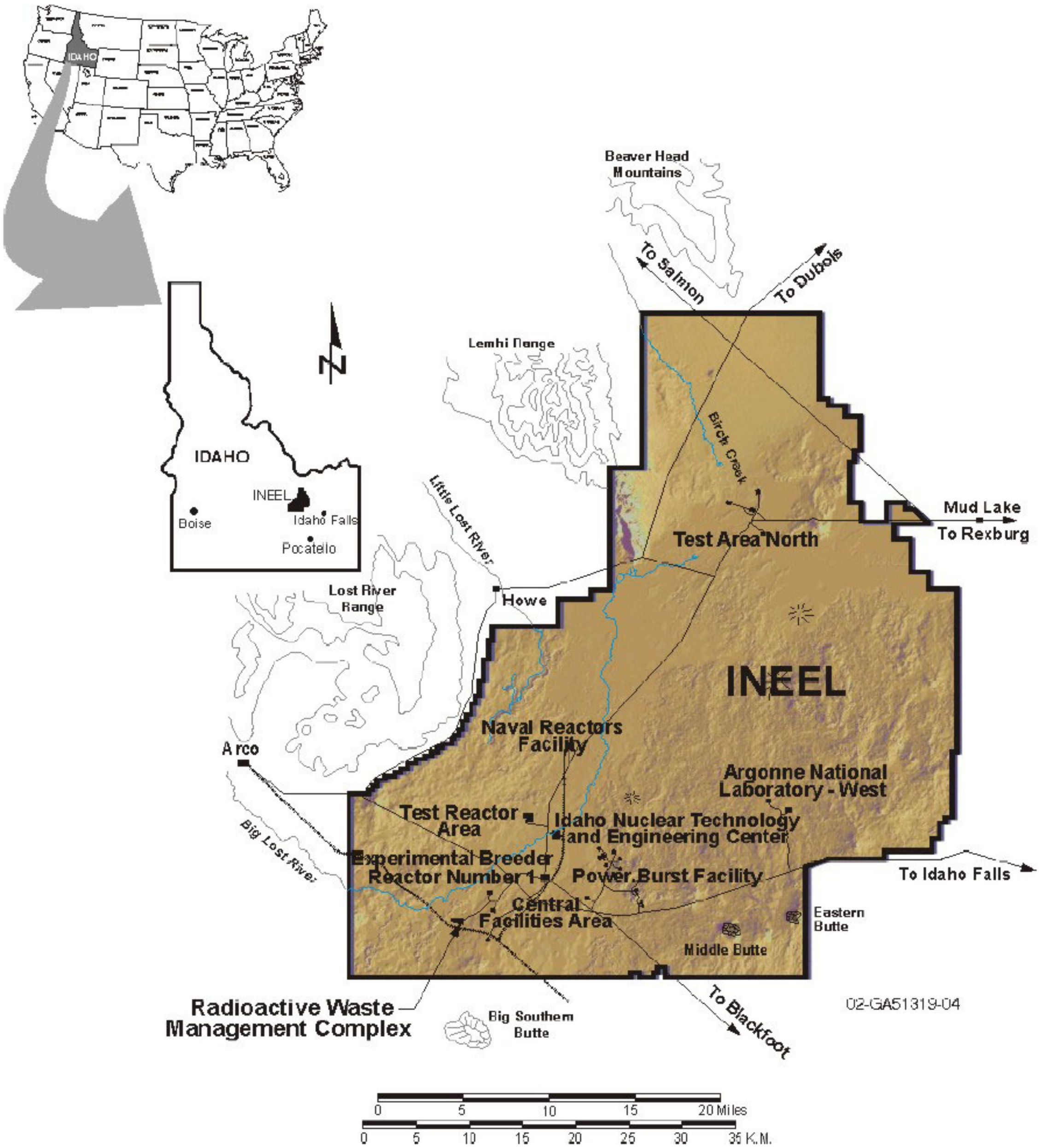

Figure 13. Idaho Site map showing locations of the Radioactive Waste Management Complex (RWMC) of which the Subsurface Disposal Area (SDA) is part and other major facilities (Nitschke et al. 2004)

Idaho Site Subsurface Disposal Area (SDA)

The Idaho Site Subsurface Disposal Area (SDA) comprises a 97-acre area (or approximately that of 88 adjoining football fields) in the Radioactive Waste Management

Complex (RWMC) as illustrated in Figure 13 and Figure 14. Transuranic (TRU) wastes, received from the Rocky Flats Plant near Denver, Colorado, were buried in the SDA before 1970 and stored retrievably (i.e., aboveground) in the RWMC after that. Other wastes (including small amounts of TRU-contaminated materials and large amounts of fission products and organic solvents) were buried in the SDA that were either received from other DOE sites or generated at the Idaho Site.

The wastes originally buried in the SDA were diverse in their sizes and forms. Wastes were buried in drums, garbage cans, and wooden and cardboard boxes; however, large pieces of contaminated equipment, loose material, and debris were also buried in the pits and trenches at the SDA. Highly radioactive materials were buried in shielded casks to reduce potential radiation exposure. Low-level, but high-activity liquid wastes containing fission products were disposed of in augur holes in the SDA.

The wastes buried in the SDA were unique both in their magnitude and diversity. The wastes were diverse in the variety of contaminants (i.e., radioactive and hazardous) and how the contaminants were intermixed. The contaminants differed dramatically in degree of toxicity, how they were released into the subsurface environment, how they move through the environment, and how they would potentially impact both human health and the ecology. The contaminants also vary in how long they remain toxic. The radionuclides will decay over time but at different rates, the organic constituents may be biodegraded, and the hazardous metals will retain their intrinsic toxicity; all contaminants will be subject to attenuation to varying extents by adsorption, dispersion and dilution processes.

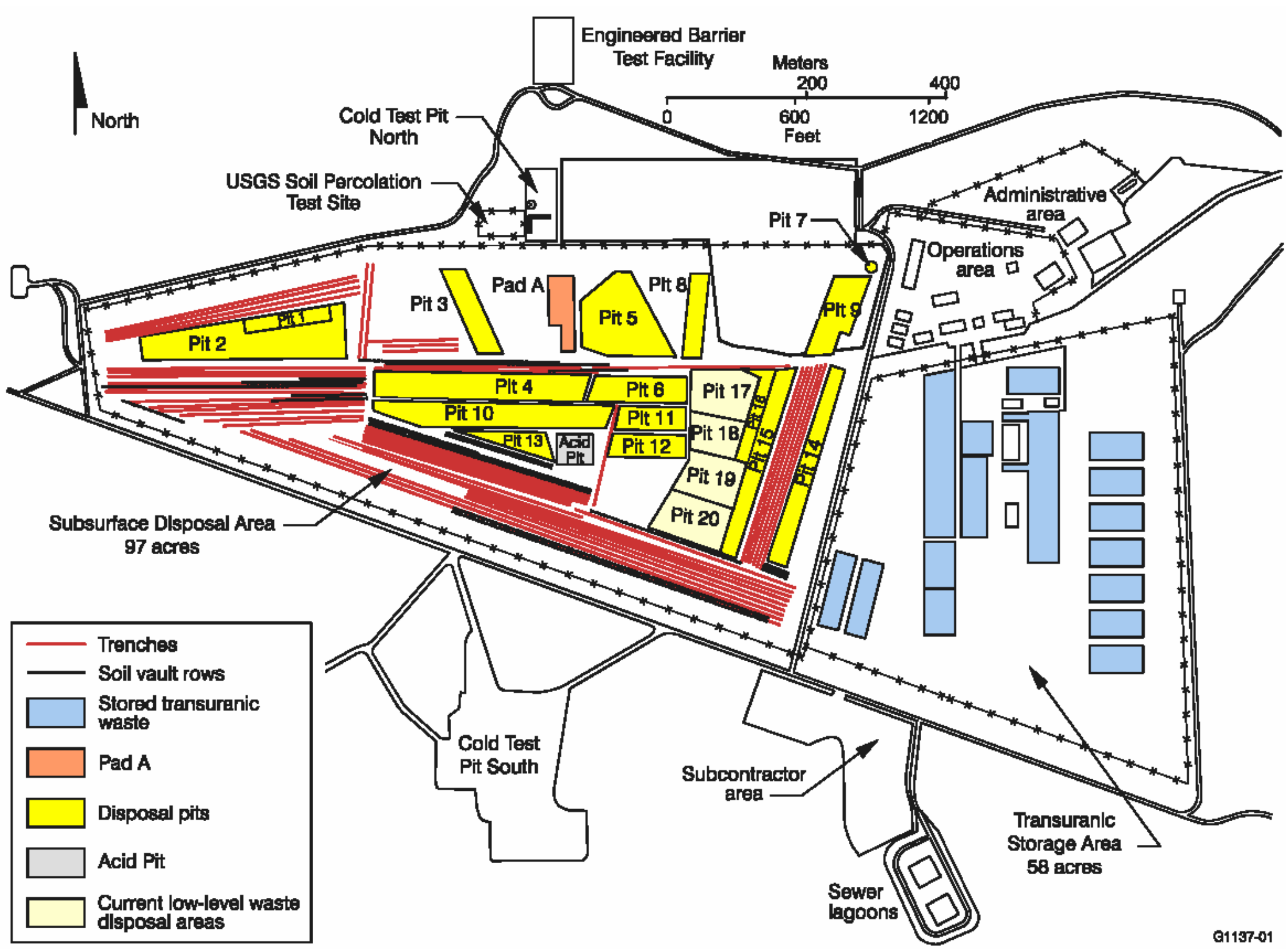

Figure 14. Map of the Subsurface Disposal Area (SDA) within the Idaho Site Radioactive Waste Management Complex (Nitschke et al. 2004).

Some examples will bring the unique nature of the SDA buried wastes into sharper relief. While estimates vary, there was likely more than a metric ton (i.e., 1000 kg) of plutonium buried throughout the SDA; this mass of plutonium, if concentrated, is equivalent to that of a late-model Volkswagen Beetle. Liquid wastes containing high activity fission products (e.g., strontium, cesium, etc.) were injected into simple augur holes. Solvents and waste sludges in containers were also buried in the SDA. Given the diversity of wastes, on-going and future remedial activities for SDA, performed under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), will likely prove to be, if not unique, then very challenging.

The current DOE baseline plan is that the volatile organic contaminants (VOCs) of concern will be removed using vapor extraction and that targeted areas containing buried TRU waste originally from the Rocky Flats Plant[72] will either be i) excavated and the retrieved TRU waste shipped to the Waste Isolation Pilot Plant (WIPP) by no later than 2018 or ii) treated to immobilize the buried TRU waste in-place. Installation of a surface barrier is envisioned for any remedial alternative involving the SDA and will cover all disposal areas of the SDA, not only those currently containing TRU waste.

The wastes as originally buried in the SDA appear reasonably well characterized as to location, type, and subsurface matrices; however, the corresponding information for the contaminants of potential concern (COPC) in the environment surrounding the SDA is not as well characterized, which is understandable considering the nature and amounts

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[72] The SDA pits and trenches that TRU-contaminated wastes were buried include Pits 1-6, Pits 9-12, and Trenches 1-10. Pad A may also contain such wastes. It is known that volatile organic compounds have migrated from the original buried waste sites to the Snake River Aquifer and TRU elements may also have migrated (although indications have been sporadic). A subsurface probing program is on-going that should reduce uncertainty in understanding the extent to which the contaminants have migrated and what must be done if the retrieve, treat, and dispose remedial alternative is selected.

of wastes buried and subsurface. The periodic infiltration of water through the waste areas has resulted in contaminant migration into the subsurface. Major previous flooding events also redistributed some wastes and contaminants.

Waste zone monitoring indicates that VOCs, plutonium isotopes, Am-241, and uranium isotopes have migrated at least some distance from the original burial sites (Holdren et al. 2006). Additional monitoring indicates that, in addition to the VOCs, the following radionuclides have migrated into the vadose zone beneath the SDA: Tc-99, Am-241, Pu-239, Pu-240, Sr-90, and Pu-238. Volatile organic compounds and nitrates have migrated to the sole-source Snake River Plain Aquifer (SRPA) underlying the SDA.

The COPCs detected in the SRPA include carbon tetrachloride, trichloroethylene, uranium isotopes, and Cs-137. The only contaminant whose concentration in the SRPA exceeds its Maximum Contaminant Level (MCL)[73] is carbon tetrachloride (Holdren et al. 2006). Other aquifer contaminants (that are not deemed COPCs) originating from the SDA include tritium, sulfate, chloride, chromium, and toluene. Contaminants including C-14, nitrates, Pu-238, Am-241, Pu-239, Pu-240, tetrachloroethylene, and methylene chloride are intermittently detected in the SRPA. There is an on-going integrated probing project in the SDA to identify the extent of contamination (Miller 2003; Salomon 2004).

To support the Idaho Site CERCLA remedial process, site personnel performed risk assessments for baseline SDA conditions (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) and a short-term risk evaluation for proposed remedial actions (Schofield 2002). For example, the predicted baseline risks for the SDA are provided in

[73] The Maximum Contaminant Level (MCL) is the highest permissible level of the contaminant in drinking water for it to be considered suitable for human consumption. This contaminant level is enforceable by the U.S. EPA. More information is available at http://www.epa.gov/waterscience/criteria/drinking/ (accessed March 13, 2008).

Table 9. These assessments provide guidance on the risks posed by the SDA wastes and the contaminants that will continue migrating from their original burial locations. However, these assessments have not fully considered the range of populations potentially at risk, time frames of specific risks, or land use remaining under government control in perpetuity (Brown et al. 2005). Consideration of these issues is important to determine who is at risk and when under the range of future land use scenarios.

One challenge shared by all stakeholders is deciding how useful are the extant risk characterizations and the magnitude, importance, and impact of uncertainties on the risk results and, ultimately, on the remedial decision. This dissertation and the risk evaluations that are made do not exist in a vacuum. This research seeks to build on, complement, and help structure existing knowledge and continuing remedial and information-gathering efforts at the buried waste sites across the DOE Complex in a way to make the remedial action decisions more consistent and transparent to stakeholders.

"Early remedial" actions to address specific areas of concern based upon mobility and toxicity are being undertaken at the SDA. Beryllium blocks, which became radioactive after being used as reflectors in Idaho Site test reactors, were buried in pits, trenches, and soil vault rows. These blocks have been grouted in-place to immobilize C-14; however, this action does not preclude future retrieval if deemed necessary. There is an on-going removal of organic contamination in the vadose zone using vacuum vapor extraction. Between 1996 and the middle of 2005, almost 92,000 kg of volatile organic compounds (VOCs) were extracted from the SDA using vapor vacuum extraction units and destroyed. The results from applying the risk analysis framework will be examined to determine if these "early actions" are supported by the results.

Table 9. Baseline Risks for SDA Human Health Contaminants of Potential Concern (Information reproduced from Table E-1 in Holdren et al. (2002))

Contaminant	Note	Peak Risk[a]	Year	Peak HI[a]	Year	Primary 1,000-Year Exposure Pathway
Ac-227	1,3	3E-06	3010[b]	NA[c]	NA	Groundwater ingestion
Am-241		**3E-05**	2953	NA	NA	Soil ingestion, inhalation, external exposure, and crop ingestion
Am-243		4E-08	3010[b]	NA	NA	External exposure
C-14	1,4	**6E-04**	2278	NA	NA	Groundwater ingestion
Cl-36		6E-06	2110	NA	NA	Groundwater ingestion
Cs-137		5E-06	2110	NA	NA	External exposure
I-129	1,3	**6E-05**	2110	NA	NA	Groundwater ingestion
Nb-94	1,3	**8E-05**	3010[b]	NA	NA	External exposure (groundwater ingestion)
Np-237	1,4	**4E-04**	3010[b]	NA	NA	Groundwater ingestion
Pa-231		3E-06	3010[b]	NA	NA	Groundwater ingestion
Pb-210		5E-07	3010[b]	NA	NA	Soil and crop ingestion
Pu-238	2	**1E-09**	2286	NA	NA	Soil and crop ingestion
Pu-239	2	**2E-06**	3010[b]	NA	NA	Soil and crop ingestion
Pu-240	2	**2E-06**	3010[b]	NA	NA	Soil and crop ingestion
Ra-226		3E-06	3010[b]	NA	NA	Soil and crop ingestion
Sr-90	1,4	**1E-04**	2110	NA	NA	Crop ingestion
Tc-99	1,4	**4E-04**	2110	NA	NA	Groundwater ingestion and crop ingestion
Th-229		4E-07	3010[b]	NA	NA	Groundwater ingestion
Th-230		7E-07	3010[b]	NA	NA	Groundwater ingestion
Th-232		1E-09	3010[b]	NA	NA	Groundwater ingestion
U-233	1,3	**3E-05**	3010[b]	NA	NA	Groundwater ingestion
U-234	1,4	**2E-03**	3010[b]	NA	NA	Groundwater ingestion
U-235	1,4	**1E-04**	2662	NA	NA	Groundwater ingestion
U-236	1,4	**1E-04**	3010[b]	NA	NA	Groundwater ingestion
U-238	1,4	**3E-03**	3010[b]	NA	NA	Groundwater ingestion
Carbon tetrachloride	1,5	**2E-03**[d]	2105	**5E+01**[d]	2105	Inhalation and groundwater ingestion
Methylene chloride	1,3	**2E-05**[d]	2185	1E-01[d]	2185	Groundwater ingestion
Nitrates	1,6	NA	NA	**1E+00**	2120	Groundwater ingestion
Tetrachloroethylene	1,6	NA	NA	**1E+00**[d]	2137	Groundwater ingestion and dermal exposure to contaminated water

Notes: For toxicological risk, the peak hazard index is given, and for carcinogenic probability, the peak risk is given.

1. Green = the contaminant is identified as a human health contaminant of concern based on carcinogenic risk greater than 1E-05 or a hazard index greater than or equal to 1 contributing to a cumulative hazard index greater than 2.
2. Brown = plutonium isotopes are classified as special case contaminants of concern to acknowledge uncertainties about plutonium mobility in the environment and to reassure stakeholders that risk management decisions for the SDA will be fully protective.
3. Blue = carcinogen risk between 1E-05 and 1E-04.
4. Red = carcinogen risk greater than 1E-04.
5. Pink = toxicological (noncarcinogen) hazard index greater than or equal to 1.

a. The peak risk and hazard index (HI) are the maximum carcinogenic risk and noncarcinogenic hazard index computed during the 1,000-year simulation period.
b. The peak groundwater concentration does not occur before the end of the 1,000-year simulation period. Groundwater ingestion risks and hazard indices were simulated for the peak concentration occurring within 10,000 years and are not presented in this table.
c. NA = not applicable.
d. The risk estimates were produced by scaling the results from the *Interim Risk Assessment* (Becker et al. 1998) based on inventory updates.

Oak Ridge Bear Creek Burial Grounds (BCBG)

The Bear Creek Burial Grounds (BCBG) are located within the Beak Creek Valley, an area mostly contained in the Department of Energy (DOE) Oak Ridge Reservation (as shown in Figure 15) approximately 20 miles northwest of Knoxville, Tennessee. The valley is over 10 miles long and runs from the eastern end of the Oak Ridge Y-12 Plant to the Clinch River. As illustrated in Figure 16, there are multiple individual waste units within the valley containing various types of hazardous and radioactive wastes derived primarily from Y-12 Plant operations. Groundwater has been contaminated throughout at least the eastern 3 miles of the valley, including commingled plumes from various contaminant sources (SAIC 1996a).

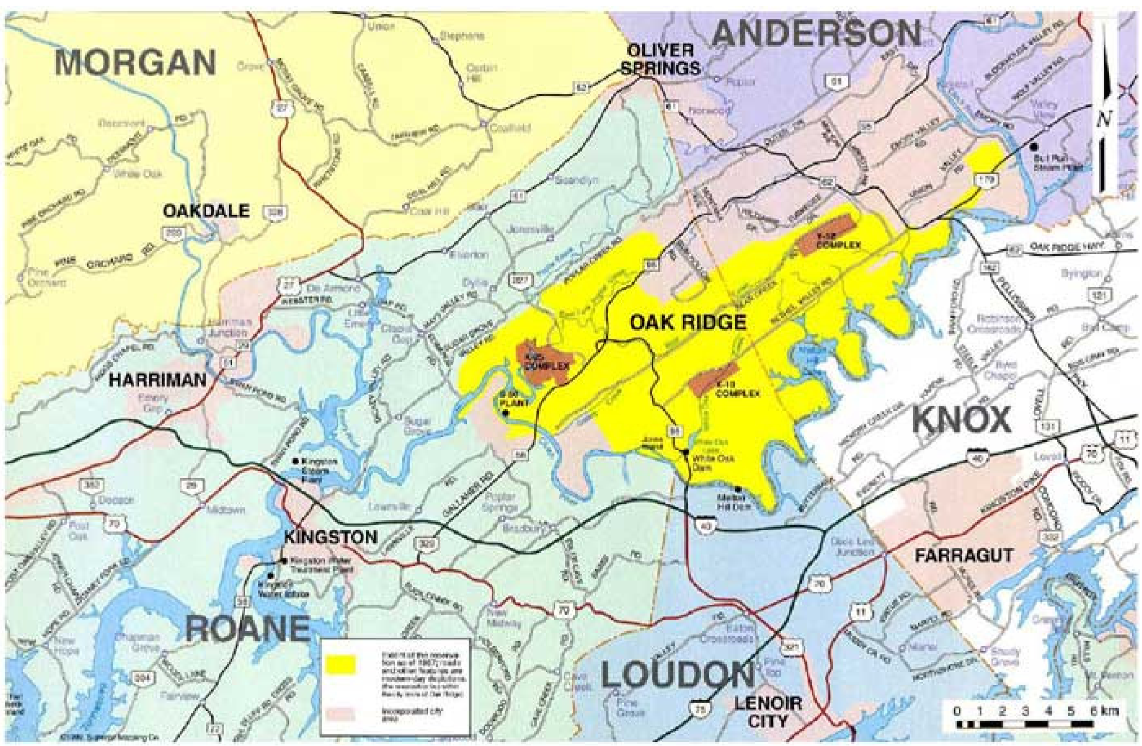

Figure 15. Location of the Oak Ridge Reservation (From ATSDR (2006))

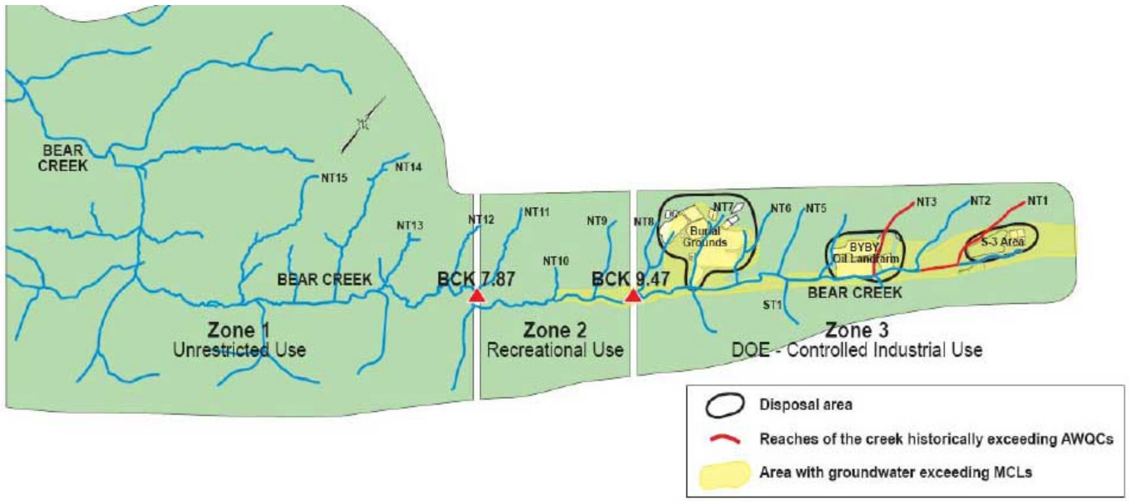

Figure 16. Bear Creek Valley Land-Use Zones. The Bear Creek Burial Grounds (BCBG) are located in Zone 3. (From ATSDR (2006))

Hazardous and radioactive wastes produced during operation of the Y-12 were disposed of at various sites in the Bear Creek Valley. Large volumes of solid hazardous and radioactive wastes (particularly contaminated with uranium) were buried in trenches located at the BCBG. Hazardous liquids are known to have been disposed of at various locations including the BCBG. Soils, groundwater, and surface water at each of these sites including the BCBG are known to be contaminated.

At the BCBG, solid and liquid wastes were disposed of in a series of unlined trenches (SAIC 1996a). Uranium-contaminated wastes including pyrophoric metallic uranium fines, chips, and cuttings dominate the material disposed in the BCBG with a total estimated uranium metal mass of 18.6x10^6 kg (40x10^6 lb). Other unstable materials including reactive and explosive materials were also buried. Liquid waste disposal resulted in volatile organic compound (VOC) contamination in groundwater that may have reached depths of 200 m (600 ft).

Primary contaminants detected in the environment surrounding the Bear Creek Burial Grounds have included (SAIC 1996a):

- *Groundwater*: boron; tetrachloroethylene (PCE); trichloroethene (TCE); 1,1,1-trichloroethane (1,1,1-TCA); and 1,2-dichloroethylene (1,2-DCE);
- *Surface water*: beryllium; 1,1,2-TCA; 1,1-DCE; 1,2-dichloroethane (1,2-DCA); vinyl chloride; 1,1-DCA; 1,1-TCA; chloroethane; and uranium isotopes;
- *Soils surrounding waste areas:* arsenic, vanadium, polychlorinated biphenyls (PCBs), acetone, and toluene; and
- *Wastes*: uranium, beryllium, PCBs, TCE, and PCE.

Organic contamination of environmental media tends to be more widespread than inorganic and radionuclide contamination. Other wastes not found in the environment but buried in the site that would impact waste retrieval and handling actions include pyrophoric uranium and other unstable materials (i.e., reactive and explosive chemicals).

Contaminant concentrations in groundwater near the BCBG exceed Applicable or Relevant and Appropriate Requirements (ARARs) for inorganic and organic chemicals in drinking water. As of 1996, groundwater and surface water contamination had been dominated by volatile organic contaminants (VOCs) with three major groundwater plumes identified (SAIC 1996a):

- The Burial Ground A (BG-A) plume is dominated by PCE, TCE, and 1,2-DCE, and includes DNAPL.
- The Walk-in Pits (WIP) plume is dominated by PCE.
- The plume at the North Tributaries 8 (NT-8) catchment is dominated by 1,2-DCE with lesser concentrations of vinyl chloride.

Radiological contamination has been virtually absent from groundwater wells in the vicinity of the BCBG; however, uranium has been consistently detected in surface water in the area. Leachate collection in the North Tributaries (NT) catchments (as shown in Figure 16) reduced the concentration of radiological and other contaminants in the surface water. Radiological contamination present in the NT-8 floodplain soils may be related to past disposal of contaminated sediments derived from other BCBG areas.

Unlike contamination found near the Idaho Site SDA (where the vadose zone is deep and effects are delayed), the impacts of BCBG contaminants have a much more immediate impact on their surrounding environment. Impacts are primarily felt via the surface water pathway versus those for the SDA that are predicted to impact primarily groundwater resources over very long periods of time[74] (Holdren et al. 2006). This is not to suggest that both sites do not contain very long-lived contaminants—they both do—it is merely that the temporal aspects of risk are very different for the sites.

Only one SDA contaminant (i.e., carbon tetrachloride) has exceeded its MCL and most projections of unacceptable SDA risks are based on models of how contaminants will be released from the SDA and migrate through the vadose and saturated zones (Anderson and Becker 2006; Holdren et al. 2006). On the other hand, *measured* concentrations for numerous BCBG contaminants (e.g., nitrate, uranium, technetium, trichloroethene (TCE) and tetrachloroethene (PCE)) in the Bear Creek Valley often exceed their MCLs in both surface and groundwater samples (SAIC 1996a). Peak risks associated with various contaminants originating in the BCBG based on either measured or predicted maximum concentrations are provided in Table 10. These results highlight the temporal differences for evaluating risks for the two very different prototype sites.

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[74] As might be expected from a complicated site like the SDA, there are notable exceptions, namely volatile organic compounds, which have reached the Snake River Plain Aquifer "only decades" after being buried.

Table 10. Baseline Risks for BCBG Residential Contaminants of Potential Concern (Compiled from Table 5.4 in SAIC (1996a))

Contaminant	Note	Peak Risk[a]	Peak Hazard Index[a]	Primary Exposure Pathway(s)
Co-60	1,3	**3E-04**	NA[b]	Soil external
Cs-137		4E-06	NA	Groundwater ingestion
K-40	1,3	**3E-04**	NA	Groundwater and surface water ingestion
Th-228		6E-06	NA	Groundwater and surface water ingestion
Th-232	1,2	**2E-05**	NA	Soil ingestion
Tl-208	1,3	**1E-03**	NA	Soil external
U-233/234	1,3	**9E-03**	NA	Soil and surface water ingestion, inhalation, and external
U-235	1,3	**3E-02**	NA	Soil external and ingestion
U-238	1,3	**2E-01**	NA	Soil external and soil, surface water, and groundwater ingestion and soil inhalation
1,1,2-Trichloroethane		8E-06	NA	Groundwater and surface water inhalation and ingestion
1,1-Dichloroethene	1,3,4	**4E-02**	**2E+00**	Groundwater and surface water inhalation, ingestion, and dermal contact
1,1-Dichloroethane	1,3	**1E-04**	NA	Groundwater and surface water inhalation and ingestion
4,4'-DDT		3E-06	NA	Soil ingestion
Arsenic	1,2	**9E-05**	NA	Soil ingestion and dermal contact
Benzene	1,3	**7E-04**	NA	Groundwater inhalation and ingestion; surface water inhalation; soil inhalation, ingestion, and dermal contact
Benzidine	1,3	**1E-02**	NA	Soil ingestion and dermal contact
Beryllium	1,3,4	**2E-02**	**2E+00**	Groundwater, surface water, and soil ingestion and dermal contact
Chloroform	1,3	**2E-04**	NA	Groundwater inhalation and ingestion; surface water inhalation
Methylene Chloride		2.5E-06	NA	Groundwater inhalation and ingestion
PCBs	1,3	**3E-02**	NA	Soil ingestion and dermal contact
Tetrachloroethene	1,3,4	**2E-03**	**9E+00**	Groundwater and surface water inhalation, ingestion, and dermal contact
Total Uranium	1,4	NA	**1E+02**	Soil ingestion and dermal contact
***trans*-l,2-Dichloroethene**	1,4	NA	**6E+00**	Groundwater ingestion
Trichloroethene	1,3	**6E-04**	NA	Groundwater, surface water, and soil inhalation, ingestion, and dermal contact
Vinyl Chloride	1,3	**6E-03**	NA	Groundwater and surface water ingestion, inhalation, and dermal contact

Notes: For toxicological risk, the peak hazard index is given, and for carcinogenic probability, the peak risk is given. No corresponding peak risk times are provided in SAIC (1996a) although many of the results are based on measurements of environmental media from the site. The following categories are based upon those in found in Holdren et al. (2002).

1. Green = the contaminant is identified as a human health contaminant of concern based on carcinogenic risk greater than 1E-05 or a hazard index greater than or equal to 1.
2. Blue = carcinogen risk between 1E-05 and 1E-04.
3. Red = carcinogen risk greater than 1E-04.
4. Pink = toxicological (noncarcinogen) hazard index greater than or equal to 1.

a. The peak risk and hazard index (HI) are the maximum carcinogenic risk and noncarcinogenic hazard index either measured in the environment or predicted during the simulation period.

b. NA = not applicable.

Subsurface Disposal Area (SDA) Risk Analysis

The risk analysis framework described in Chapter III was applied to the Idaho Site Subsurface Disposal Area (SDA). This site was selected for evaluation based on past experience working with DOE personnel and selected Vanderbilt University faculty and Consortium for Risk Evaluation with Stakeholder Participation (CRESP) members. A detailed, qualitative risk analysis was completed for the SDA as requested by the DOE Office of Environmental Management (DOE-EM) (Brown et al. 2005).

According to the risk analysis framework and methodology defined in Chapter III, the first step in evaluating life-cycle risks for a selected site is to complete an initial characterization of site conditions. Fortunately for the interests of this research, the SDA is being studied for remedial action under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)[75]. Remedial investigations (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) and feasibility studies (Holdren et al. 2007; Schofield 2002; Zitnik et al. 2002) have been completed for the site. The information available in the Idaho Site CERCLA Administrative Record and that provided by Idaho Site personnel were used as the basis of the SDA risk evaluation.

The next step in assessing risks posed by a contaminated site is to develop a baseline conceptual site model (CSM) linking sources of contamination to potentially impacted receptors (ASTM 1995; USDOE 2003). The CSM provides an excellent framework for evaluating and communicating the risks posed by contaminated sites. Many CSMs were developed to describe contaminated sites as part of the DOE Risk-Based End State (RBES) initiative (CRESP-II 2003; USDOE 2003) with many elements

[75] The Information Repository and Administrative Record is available at http://ar.inel.gov/ (accessed March 13, 2008) to provide public access to information concerning the Idaho Site Environmental Cleanup Program performed in accordance with CERCLA and the Federal Facility Agreement and Consent Order.

common to each. Because of these commonalities, a generic baseline CSM was defined in Figure 9 in Chapter III for potential human health impacts from buried waste sites.

SDA: Conceptual Site Model (CSM) Development

The conceptual site model (CSM) in Figure 17 describing baseline SDA conditions and potential exposure risks to the general public and workers was developed using Idaho Site information (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2007; Holdren et al. 2002; USDOE-ID 2004b)[76]. These reports were used as the basis for hazard identification, exposure assessment, and receptor evaluation. Because the SDA has been in operation since 1954 and has contaminated the sole source aquifer beneath it, "early" remedial actions have been taken to treat problem areas of the site. A series of vapor vacuum extraction and treatment units have been in operation since 1996 (USDOE-ID 2004b). Despite having removed and destroyed almost 92,000 kg of VOCs, these units have operated less effectively than originally designed (Holdren et al. 2006). Areas were grouted around buried beryllium blocks that became radioactive after use as reflectors in test reactors; the grouting will help immobilize C-14.

The Idaho Site has restricted access to areas within the site to prevent intrusion by the public and has a security fence around the SDA. Drinking water wells supplying potable water to the site have been located outside the SDA and are monitored to assure water quality (USDOE-ID 2004b). An extensive surface water management system was constructed in response to major SDA flooding events in 1962, 1969, and 1982 (Holdren et al. 2006).

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[76] Conceptual site models have been developed by Idaho Site personnel (Holdren et al. 2006; USDOE-ID 2004b); however, these CSMs are not graphical and do not provide the necessary narrative description.

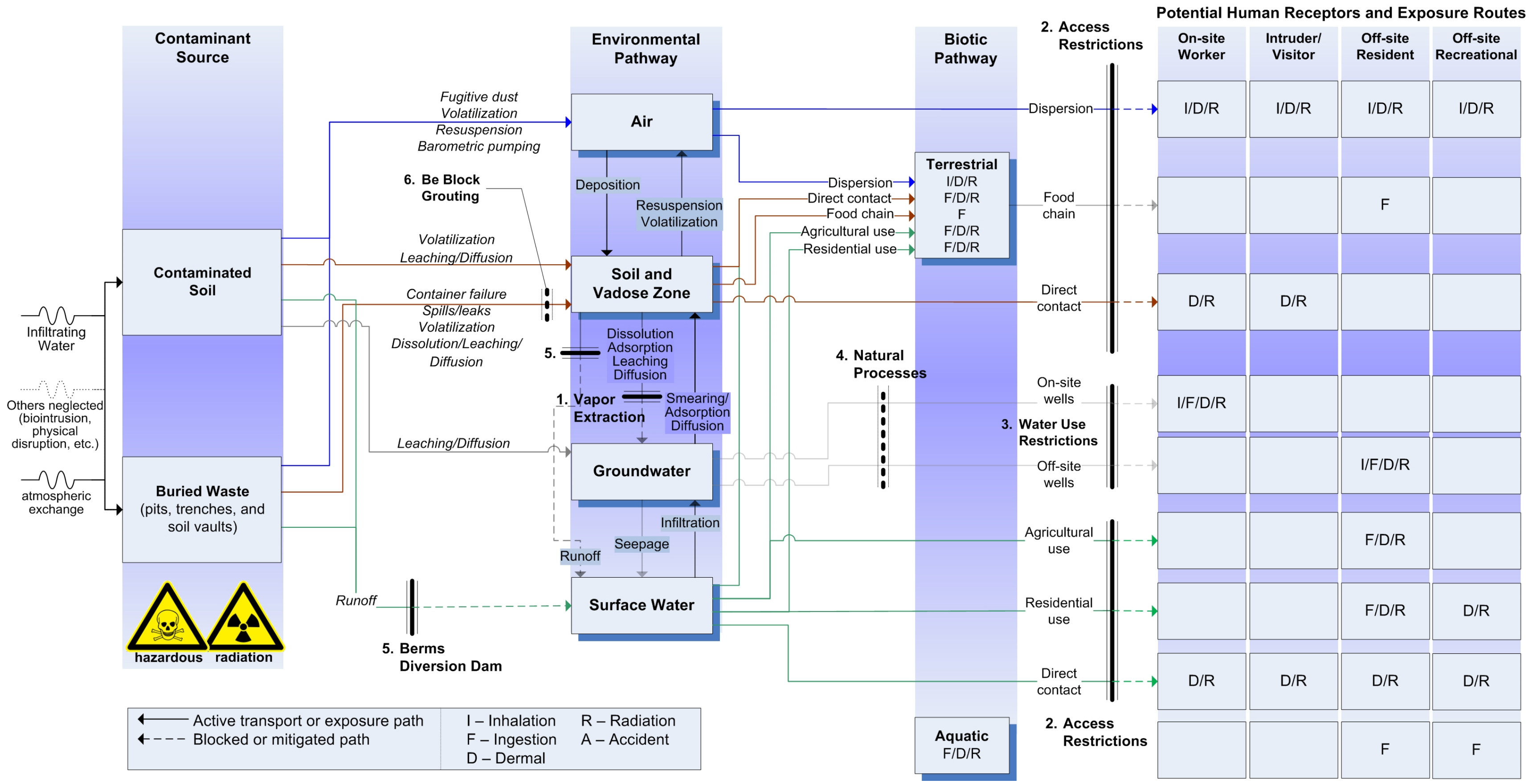

Figure 17. Baseline conceptual site model (CSM) for exposures to chemicals and radionuclides from the Subsurface Disposal Area (SDA) **before any *additional* remedial actions are undertaken**.

Narrative for Figure 17: Baseline Conceptual Site Model (CSM) for the Idaho Site Buried Waste Area

Volatile organic compounds (i.e., carbon tetrachloride and tetrachloroethylene) and nitrates pose the most imminent risks to human health (Becker et al. 1998; Holdren et al. 2006; Holdren and Broomfield 2004; USDOE-ID 2004b). Carbon tetrachloride has been detected in the Snake River Plain Aquifer (SRPA) above the maximum contaminant level (MCL) and volatiles are being extracted from the vadose zone to reduce said risk. Mobile long-lived fission and activation products are the next most immediate concern (USDOE-ID 2004b). Note that there is no aquatic biotic pathway or near surface water pathway.

The following barriers (or steps taken to mitigate impacts) are shown in Figure 17:

1. A vapor extraction system extending into the vadose zone is being used to mitigate VOC migration to the aquifer (USDOE-ID 2004b). Multiple vapor vacuum extraction with treatment units were installed within the SDA and brought into operation in 1996. Data from monitoring well vapor samples are being used to assess the effectiveness of the remedy and to optimize VOC removal.
2. The Idaho Site has restricted access to prevent intrusion by the public, and the SDA is surrounded by a security fence (USDOE-ID 2004b).
3. An extensive groundwater-monitoring program is in place at RWMC. Drinking water wells used to supply potable water to the work force are located outside of the SDA and are routinely monitored for water quality (USDOE-ID 2004b).
4. Natural attenuation of VOCs, nitrates, short-lived radionuclides (e.g., Cs-137, Sr-90, etc.), and natural processes resulting in relative immobility for some long-lived radionuclides may prevent them from reaching the Snake River via the Snake River plain Aquifer. This barrier is dotted to represent the uncertainty in relying upon natural processes.
5. An extensive surface water management system, including dikes and drainage channels, has been implemented at the SDA to minimize the potential for flooding and releases by way of surface water (USDOE-ID 2004a).
6. The beryllium blocks have been grouted to reduce the potential for C-14 migration from the site. Other radionuclides are not impacted and thus the barrier is dotted.

For very long-lived and reasonably mobile radionuclides (as well as mobile nitrates and heavy metals) that are already in the vadose zone, a pathway exists from the SDA to the Snake River via the vadose zone and groundwater. The only potential “barriers” that exist are decay, dispersion, and dilution for the radionuclides and dispersion and dilution for the nitrates and heavy metals.

General Relationship between CSM and Scenario Development

Development of the baseline CSM for the SDA entails the scenario development for the site. In essence, the scenario for potential exposure and thus risk to receptors are the solid lines in Figure 17 representing complete transport pathways from contaminant sources to receptors. *Gross* indications of the temporal aspects and likelihood of the exposure risks are provided by how the pathways are represented in the CSM. The lighter the pathway shading the longer the likely time to impact and a dotted line means there is little or no likelihood of human exposure via the pathway.

A *gross* indication of the consequences of a complete transport pathway is conveyed by the CSM in that a complete pathway indicates a possible exposure of a receptor to contaminants although the actual impact (e.g., latent cancer incidence, irritation, etc.) is not provided. Although the explicit consequence is not provided, a reasonable rule-of-thumb is that potential exposure (via a complete transport pathway) is an undesired consequence. This rule-of-thumb drives the desired "end-state" for remedial actions that include blocked pathways from source to receptor.

SDA: Qualitative Baseline Uncertainty and Gap Analysis

The primary uncertainties that impact the ability to determine the risks to potentially impacted receptors include the inventories and geospatial distributions of the radioactive and hazardous contaminants of potential concern (COPCs). Without a source of contamination and release to and no pathway for migration through the environment and exposure to a receptor, there is no driver for risk and no need for remedial action. Results from environmental monitoring provide a snapshot, albeit an uncertain snapshot, of the extent of contaminant migration (in the locations sampled).

The initial set of contaminants of concern (COCs) identified for the SDA was defined based on a screening analysis (Becker et al. 1998) and was reevaluated to assure the correct COCs are investigated. One contaminant, 1,4-dioxane, was added to the most recent remedial investigation (Holdren et al. 2006). Thus the COCs that must be investigated for a site may vary over time based on new information or models.

Analyses of contaminated environmental media may prompt remedial action; however, these results are not sufficient to evaluate future risks to all receptors. Future risks must be *predicted* (or bounded) using models and current data. Uncertainties in input data and parameters and the models (from simplification and assumption) used to predict future exposures are essential to understanding baseline risks. An explicit declaration of the value judgments and simplifying assumptions made by the risk assessor is needed as well as the likely impact of significant uncertainties on the risk estimates.

The uncertainty information for baseline conditions was taken from Idaho Site remedial investigation reports (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002). Examples of uncertainties used in the analysis included inventory estimates, water infiltration rate, and subsurface stratigraphy including the sedimentary interbed regions that impact contaminant migration). The most profound impact (i.e., several orders of magnitude) resulted from whether or not the sedimentary interbeds were assumed to adsorb plutonium isotopes. Other parameters and assumptions made in modeling the migration of contaminants to potential receptors will impact risk results. However, the sensitivities of parameters and models provided in the Idaho Site reports were based on one-at-a-time type analyses; no integrated analysis of sensitivities exists.

SDA: Qualitative Baseline Risk Evaluation

Extensive studies have been completed to evaluate the potential risks associated with wastes originally buried in the SDA (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002). Peak predicted baseline risks for SDA contaminants are provided in Table 9 (Holdren et al. 2002). Without additional information[77], the baseline peak risks for the SDA were used to evaluate the risk drivers for SDA buried wastes.

Volatile organic compounds (i.e., carbon tetrachloride, methylene chloride, and tetrachloroethylene) and total nitrates, most of which originated at the DOE Rocky Flats Plant (RFP), posed the most *imminent* risk to human health (Holdren et al. 2006). Carbon tetrachloride has been detected in the Snake River Plain Aquifer. Volatile organic compounds (VOCs) are being extracted from the vadose zone using vapor vacuum extraction (VVE) to reduce the contamination source and corresponding risks. Extraction will continue, and if risks are not sufficiently mitigated, VOCs will continue to pose the most imminent risk to human health. Vapor vacuum extraction does not address the risks from nitrates, some of which are thought to have migrated to the aquifer from the SDA[78].

Mobile, long-lived fission and activation products posed the next most immediate concern to human health under baseline conditions. Most of the mobile fission and activation products were generated by Idaho Site reactor operations (Holdren et al. 2002). For example, those contaminants that had peak risks greater than 10^{-6} in the 100-year period following institutional control (i.e., ending in 2110) included (where peak risk is provided in parentheses):

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[77] Additional information is available from the screening quantitative risk evaluation performed in Chapter VII; however, for this stage of the analysis, this quantitative analysis is assumed to not exist.

[78] The high background levels of nitrates in the aquifer make these determinations difficult; however, the increasing trend in nitrate in monitoring wells in the vicinity of the SDA is likely attributable to the SDA.

Tc-99 ($4x10^{-4}$) > Sr-90 ($1x10^{-4}$) > I-129 ($6x10^{-5}$) > Cl-36 ($6x10^{-6}$) > Cs-137 ($5x10^{-6}$)

Large uncertainties have been noted for the model parameters used to estimate risks for C-14, I-129, and Tc-99 (Holdren et al. 2002). These contaminants have been detected sporadically in the subsurface and additional work was completed to better model their transport through the Idaho Site environment (Holdren et al. 2006).

The majority of the contaminants (i.e., carbon tetrachloride, methylene chloride, tetrachloroethylene, and nitrates) that pose the most *imminent* threat to human health originated at the Rocky Flats Plant. The next most immediate concern is posed by mobile, long-lived fission and activation products primarily generated during Idaho Site reactor operations. The beryllium block grouting project will help to reduce the release of C-14 but will not limit the release or transport of other mobile, long-lived fission and activation products because these contaminants have different sources. Long-term risks result primarily from uranium and long-lived actinides. Thus the immediacy, magnitude, and source of risks must be taken into account to determine the best remedial path forward.

SDA: Preliminary Overall Assessment and Cleanup Goals

The overall cleanup goals for the Idaho Site Subsurface Disposal Area (SDA) were simple to define. The SDA is being cleaned up under the auspices of CERCLA, which specifies nine evaluation criteria[79]. These nine criteria were defined as the preliminary cleanup goals for the SDA. The other critical outcome of Phase 1 was whether or not there was sufficient information to require remedial action. An affirmative

[79] The nine CERCLA evaluation criteria are overall protection of human health and the environment, compliance with Applicable or Relevant and Appropriate Requirements (ARARs), long-term effectiveness and permanence, reduction of toxicity, mobility, or volume through treatment, short-term effectiveness, implementability, cost, state acceptance, and community acceptance (CFR 1994).

answer was obvious based on the remedial investigation results for the SDA. The contaminants of potential concern (COPCs) for the SDA were the same as those being studied in the most recent remedial investigation report (Holdren et al. 2006).

SDA Screening Quantitative Baseline and Residual Risk Evaluations

To determine the extent to which the SDA must be remedied, quantitative estimates of risk were used to identify contaminants of concern and define acceptance goals to assure that cleanup would be completed to the extent needed to satisfy cleanup goals. *Acceptance goals* represent specific contaminant levels (e.g., preliminary remediation goals (USEPA 1991), soil screening levels (USEPA 1996), etc.) or other agreed upon metrics corresponding to the high-level cleanup goals. The next step in the framework, Phase 2A: *Screening Quantitative BRA and Preliminary Acceptance Goals (PAGs) Definitions* (i.e., Figure 9 in Chapter III) outlines the process to provide the baseline risk estimates needed to identify contaminants and define acceptance goals.

The remedial investigation for the SDA (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) provided quantitative baseline risk estimates. The 1998 *Interim Risk Assessment* (IRA) could be characterized as a screening quantitative baseline risk assessment (BRA) (Becker et al. 1998). The *Ancillary Basis for Risk Assessment* (Holdren et al. 2002) and *Remedial Investigation and Baseline Risk Assessment for Operable Unit (OU) 7-13/14* (Holdren et al. 2006) could be classified as detailed (i.e., Phase 3) risk assessments as described in Chapter III. No additional updates to conceptual site models, scenarios, uncertainty analysis, or cleanup goals were considered necessary.

The final step in the screening quantitative BRA was to define quantitative assessment goals corresponding to the cleanup goals defined in Phase 1. Because the

SDA is being evaluated under CERCLA, the acceptance goals are those corresponding to the U.S. Environmental Protection Agency (EPA) 10^{-4} to 10^{-6} risk criteria for carcinogens or a Hazard Quotient (HQ) of unity for noncarcinogens. However, because cleanup must also comply with Applicable or Relevant and Appropriate Requirements (ARARs), acceptance goals were either the above levels or the appropriate contaminant of concern levels (e.g., preliminary remediation goals (USEPA 1991), soil screening levels (USEPA 1996), state-mandated maximum concentration levels, etc.).

After quantitative estimates of the baseline risks were generated, acceptable remedial alternatives must be proposed. The method employed by Idaho Site personnel for the SDA uses three of the nine CERCLA evaluation criteria (CFR 1994) (i.e., effectiveness, implementability, and cost) to screen unacceptable remedial alternatives (Zitnik et al. 2002). Remedial alternatives that were representative of those defined in the SDA feasibility study were used in this research and are described in the next section.

The residual risk levels obtained from Phase 2B: *Screening Quantitative Residual Risk Evaluation* (Figure 9, Chapter III) corresponded to the most restrictive of either the EPA risk levels or those corresponding to ARARs. No other information (e.g., conceptual site models, uncertainties, scenarios, cleanup goals, etc.) developed to this point of the risk analysis required updating. The next phase (i.e., Phase 2C in Figure 10, Chapter III) of the risk analysis involved determining the life-cycle risks associated with proposed remedial alternatives as input to the informed decision-making process.

General Remedial Alternatives Considered for Review

Both the Idaho Site Subsurface Disposal Area (SDA) and the Oak Ridge Reservation Bear Creek Burial Grounds (BCBG) are deemed to pose unacceptable human health risks (Holdren et al. 2006; SAIC 1996a; e). A feasibility study report was recently completed for the SDA (Holdren et al. 2007); however, no preferred remedial alternative was specified. A feasibility study has not yet been completed for the BCBG although early actions have been completed including the installation of RCRA Subtitle 'C' caps over areas including some in the BCBG (SAIC 1996a). Neither site has a "preferred remedial alternative" (Holdren et al. 2007; USDOE-ORO 2004). Because of the lack of information concerning BCBG remedial actions, the alternatives evaluated for the SDA were considered for potential application to both the SDA and BCBG.

Previous Remedial Actions Considered for the SDA

Liquid wastes contaminated with fissions products were poured into augur holes; whereas, wastes contaminated with transuranic elements and hazardous organics often contained in drums and boxes were buried in other areas. The types and temporal nature of the risks posed vary significantly from one part of the SDA to another. Because the SDA wastes pose unacceptable risks (Holdren et al. 2006), remedial actions are necessary to reduce risks from hazardous and radioactive contaminants to acceptable levels. The following alternatives were considered in the *Preliminary Evaluation of Remedial Alternatives* (Zitnik et al. 2002), the initial major step in the SDA feasibility study:

- A "No Action" Alternative (relying upon monitoring; this alternative is required by EPA as a basis for comparison of remedial alternatives),
- A Limited Action Alternative (including access controls, a surface barrier, and land use restrictions),
- Two Containment Alternatives (relying upon either surface or subsurface barriers to prevent access to waste and to control future contaminant migration),
- Two *In Situ* Treatment Alternatives (using either *in situ* grouting or *in situ* vitrification to treat and stabilize wastes and contaminated soil in place), and

- A Retrieval, Treatment, and Disposal (RTD) Alternative (where wastes are retrieved and the waste and contaminated soil are segregated and treated for either on-or off-site disposal depending on the nature of the waste).

The SDA alternatives originally found to satisfy the three CERCLA screening criteria (i.e., effectiveness, implementability, and cost) were

- **Surface Barrier**. This alternative includes *in situ* grouting (ISG) and *in situ* thermal desorption (ISTD) pretreatment[80] in areas with high organic concentrations, Pad A reconfiguration[81], installation of a long-term multi-layer cover, land-use restrictions, and long-term monitoring and maintenance.

- ***In Situ* Grouting**. This alternative employs ISG to treat and stabilize waste and contaminated soil, Pad A retrieval and *ex situ* treatment/disposal, ISTD pretreatment, long-term multi-layer cover installation, land-use restrictions, and long-term monitoring and maintenance.

- ***In Situ* Vitrification**. This alternative uses *in situ* vitrification (ISV) to treat and stabilize waste and contaminated soil, Pad A reconfiguration, ISG and ISTD pretreatment, installation of a long-term multi-layer cover, land-use restrictions, and long-term monitoring and maintenance.

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[80] *In situ* thermal desorption (ISTD) pretreatment is a remedial process where heat and vacuum are applied to contaminated soil (Abbott 2003). In the SDA, those pits considered for ISTD contain organic and nitrate sludges, combustible solids, and graphite wastes contaminated with plutonium. ISTD may be employed in areas with high organic concentrations to remove nonradioactive contaminants. Two major processes comprise ISTD: (1) underground thermal desorption and (2) subsequent off-gas treatment (Abbott 2003).

[81] Pad A was constructed in 1972 and received wastes until 1978 including many from test retrieval actions in SDA Pits 11 and 12 (Holdren et al. 2006; Zitnik et al. 2002). Pad A consists of an asphalt pad built in an area unsuitable for subsurface disposal because of near-surface outcroppings of basalt. Wastes in Pad A primarily contain nitrates and transuranic (TRU) radioisotopes with concentrations less than 10 nCi/g and radiation levels less than 200 mR/hr at the surface (Holdren et al. 2006; Zitnik et al. 2002). Any remedial alternative considered for the SDA includes emplacement of a surface barrier; therefore, Pad A must be reconfigured before any surface barrier can be constructed. However, to simplify the analysis and modeling and because there was potential for waste migration from these pits, Pad A wastes are "returned" to Pits 11 and 12 in the screening risk tool and reconfiguration of Pad A is not considered further.

- **Retrieval, Treatment, and Disposal**. This alternative employs retrieval and *ex situ* treatment,[82] disposal of TRU wastes at the Waste Isolation Pilot Plant (WIPP), on-site disposal of low-level waste and treated mixed low-level waste material, ISG and ISTD pretreatment, long-term multi-layer cover installation, land-use restrictions, and long-term monitoring and maintenance.

Short-term risks for these alternatives were previously evaluated (Schofield 2002).

The above remedial alternatives appear to provide adequate stabilization and containment of the contaminants buried in the SDA (Holdren et al. 2007; Schofield 2002; Zitnik et al. 2002). Furthermore, the alternatives appear reasonable from what is known of the SDA, the wastes buried there, and available remedial technologies. The SDA feasibility study provides no *new* alternatives; however, some steps are more fully defined and more refined remedial techniques are added (Holdren et al. 2007).

The primary difference between previous alternatives and those in the feasibility study is that the retrieve, treat, and dispose (RTD) alternative was separated into three alternatives based on the extent of excavation (Holdren et al. 2007). Originally, a "partial RTD" alternative (including partial excavation of Pit 4) was intended to represent what was possible (Holdren and Broomfield 2004; Sentieri 2004). However, a *targeted* retrieval alternative based on the SDA areas posing highest risks was suggested by Brown et al. (2005). The inclusion of two new targeted retrieval actions in the SDA feasibility study is an important step in making a risk-informed remedial decision.

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[82] Five types of technological options were originally considered for *ex situ* treatment of wastes retrieved from the SDA: 1) physical, 2) chemical, 3) thermal, 4) electrokinetic, and 5) biological. Of these, three (i.e., physical, chemical, and thermal treatment) options were retained for further consideration (Zitnik et al. 2002). The final decision on the specific treatment technology will be made in conjunction with DOE Idaho, EPA, Idaho Department of Environmental Quality, and other stakeholders. Because of experience with the Advanced Mixed Waste Treatment Facility (AMWTF), it is assumed that *ex situ* treatment will consist of compaction (Zitnik et al. 2002).

General Remedial Alternatives Considered in this Research

The remedial alternatives proposed for the SDA and evaluated in this research fall into two broad categories:

1. *Manage In-Place (MIP)*: Remedial approaches involve *managing buried wastes where they are* to assure they do not migrate and find pathways to receptors, and

2. *Retrieve, Treat, and Dispose (RTD)*: Actions involve *excavation and retrieval of wastes for segregation, treatment, packaging, and ultimate disposal* either on-site or off-site so that the contaminants will be contained.

Options such as the specific type of engineered surface barrier to be used and the need to employ thermal pretreatment appear to be less important in terms of life-cycle risks (and other factors that lead to a risk-informed decision) than whether or not the site will have to be excavated and wastes retrieved, segregated, treated, and packaged for on-site disposal or, in the case of TRU wastes, sent off-site to the WIPP for long-term disposal.

Table 11 summarizes the SDA remedial alternatives evaluated in this research. There is one substantial difference between the alternatives listed in Table 11 and those previously evaluated in earlier research (Brown et al. 2005). That is, the *in situ* vitrification (ISV) remedial alternative is no longer considered a viable option for the SDA[83] and was removed from consideration in this research. No additional remedial alternatives are evaluated in this report.

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[83] Although originally identified as a potential remedial option (Schofield 2002; Zitnik et al. 2002), *in situ* vitrification was removed from consideration for the SDA (Holdren and Broomfield 2004). Although no formal justification was provided provided by Holdren and Broomfield (2004) for this exclusion, the omission is likely due to safety considerations resulting from testing at PNNL, Oak Ridge, and other sites. This alternative is not considered in the current SDA feasibility study (Holdren et al. 2007) and was thus dropped from consideration in this research.

Table 11. Possible Subsurface Disposal Area (SDA) Disposition Alternatives[a]

	Stabilization	Surface Barrier	*In Situ* Treatment	*Ex Situ* Treatment	On-Site Disposal	Off-Site Disposal
1. Manage-in-Place (MIP) Alternative (3 options)						
1A. No Action option[b]						
1B. Surface barrier with *in situ* grouting (ISG) for stabilization	√	√			√	
1C. *In Situ* Grouting (ISG) for contaminant immobilization and soil stabilization with surface barrier	√	√	√		√	
2. Retrieve/Treat/Dispose (RTD) Alternative (2 options)						
2A. Targeted Rocky Flats Plant (RFP) transuranic (TRU) waste retrieval with surface barrier[c]		√	√	√	√	√
2B. Maximum RFP TRU waste retrieval with surface barrier[c]		√	√	√	√	√

a. Although identified as a potential options (Schofield 2002; Zitnik et al. 2002), *in situ* vitrification and *in situ* thermal desorption have been removed from consideration for the SDA because they have been deemed unimplementable (Holdren et al. 2007). These options have also been omitted in this research.

b. This option represents baseline conditions and is required under CERCLA for comparison purposes. Although no further actions would be taken to reduce contaminant mobility, toxicity, or volume, long-term monitoring, maintenance, and institutional controls would be instituted.

c. The retrieval alternatives may hinge on future legal decisions concerning to what extent the transuranic (TRU) wastes originating from the Rocky Flats Plant (RFP) must be removed from the SDA. The options presented concern targeted retrieval of RFP TRU wastes versus maximum retrieval of all buried RFP TRU wastes. The wastes retrieved would be segregated, treated, packaged, and TRU wastes transported to the Waste Isolation Pilot Plant (WIPP) for disposal (Schofield 2002). The non-TRU wastes retrieved would be disposed of on-site. It is also possible that high-level waste (HLW), spent nuclear fuel (SNF), or wastes analogous to HLW and/or SNF were buried. If HLW, SNF, or analogous materials are unearthed during retrieval, then additional segregation and storage tasks (prior to disposal in a geologic repository) would be needed.

For the MIP alternative, buried wastes would be managed using *in situ* techniques without excavation and waste retrieval. All options include long-term monitoring, maintenance, and institutional controls and installation of a surface barrier (except for the "No Action" alternative). Three options were examined under this alternative:

- *"No Action"* (although long-term monitoring, maintenance, and institutional controls would be instituted);
- *Surface Barrier Installation* with *in situ* grouting as needed for subsurface geotechnical stabilization but not contaminant immobilization; and

- *In Situ Grouting* for both subsurface geotechnical stabilization and contaminant immobilization with surface barrier installation.

For the RTD alternatives, site personnel would excavate the buried waste area and retrieve, segregate, treat, and package wastes for disposal either on-site or elsewhere. All options included long-term monitoring, maintenance, and institutional controls and installation of a surface barrier. The extent of excavation and retrieval may ultimately be based on risks, costs, and legal decisions; however, two options were examined in this research:

- *Targeted Waste Retrieval* based on the risks associated with the wastes with *in situ* grouting (ISG) for subsurface geotechnical stabilization in retrieval areas and ISG for contaminant immobilization in selected non-retrieval areas.
- *Maximum Waste Retrieval* with ISG for subsurface geotechnical stabilization. For the SDA, wastes must be retrieved from all areas because of the likely presence of Rocky Flats Plant (RFP) transuranic (TRU) wastes as described in Appendix A.

Retrieved wastes that are not TRU in nature are assumed to be returned to their original burial sites for the SDA retrieval scenarios.

Additional Risk Considerations

Another way to characterize remedial actions is by how aggressively the waste areas must be remedied. Installation of a surface barrier helps reduce the flux of water to buried wastes, but does not impact the source of contamination or the potential risks if the surface barrier fails. Other remedial alternatives considered represent a progression from less to more aggressive approaches where contaminants would be immobilized in-place to those involving retrieving the highest risk wastes for treatment and disposal elsewhere.

For example, in the Idaho Site SDA it appears as though nitrates, Tc-99, and Sr-90 would present the largest *short-term* risks after VOCs are extracted (Holdren et al.

2002). Thus the remedial decision concerns whether buried wastes can be effectively managed in-place or must be retrieved to be protective based on waste form, geospatial distribution of contaminants, temporal nature of potential risks, and retrievability of the risk-driving contaminants. The risk reduction to potential receptors achieved through the proposed retrieval action should be balanced by the worker and public exposure and accident risks posed by the action.

The different areas where wastes were buried within the SDA and BCBG present very different hazards and risks;[84] therefore, both the remedial actions and approach to risk evaluation must be able to incorporate spatial and temporal differences in risk. The approach used in this research may be used to evaluate the site as a whole, or applied to sub-areas (or "hot-spots") that may warrant specific remedial approaches.

SDA: Qualitative Remedial Alternatives Risk Evaluation

As illustrated for Phase 2C in Figure 10 (Chapter III), remedial alternative risks were evaluated by completing the following steps:

- A *task list* was developed in conjunction with a *management flow diagram* which included the primary subtasks and sequence required to implement the alternative.
- A *risk flow diagram* was developed that indicated the sequence of activities that have the potential to pose significant health risks to workers or the general public. *Conceptual site models* for the remedial actions and final protective states were also developed.
- A set of uniform terminologies and categories were previously developed to characterize both hazards and knowledge gaps in a meaningful fashion as described in Chapter III.

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[84] For example, the SDA pits and trenches where Rocky Flats Plant wastes containing volatile organic compounds and TRU-contaminated wastes and sludges were buried pose decidedly different risks than those areas where liquid wastes containing fission products were disposed of in augur holes.

- A detailed *hazard analysis* was developed. For each subtask, the following were determined: task frequency, what can potentially go wrong, how likely is the adverse event to occur, severity of the consequences, impacted population, basis for characterizing the risk, and contribution of the subtask to overall risk.

- A detailed *gap analysis* describing key knowledge barriers, missing information, and uncertainties was performed. For each primary subtask, knowledge gaps were identified and then characterized by: what information is missing, how important the missing information is, and how large the knowledge gaps.

- An *integrated hazard and gap analysis* was performed and the most important potential risks and information gaps were provided in a summary table.

SDA: Task List Development

From the preliminary qualitative evaluation of the risks posed by the remedial alternatives proposed for the SDA, a great deal of commonality amongst the various process steps comprising the alternatives was discovered (Brown et al. 2005). The set of ten process steps listed in Table 12 appeared to characterize the various process steps needed to complete the manage-in-place and retrieval alternatives. The process steps and corresponding task lists were defined as generically as possible for application to both the SDA and the Oak Ridge Bear Creek Burial Grounds (BCBG).

Exhibit 5 provides a generic set of task lists corresponding to the process steps in Table 12, combinations of which comprise the remedial alternatives for the SDA and BCBG. For each alternative, the information in Table 12 was used to determine those process steps that were involved and then the task lists in Exhibit 5 were used to describe the steps needed to execute the remedial alternative. For this research, the order of the process steps was assumed to be the same as that in Table 12. The next step was to define the *management flow diagrams* to more fully and transparently convey the activities required to carry out the remedial alternatives.

Table 12. General Process Steps Needed to Disposition DOE Buried Wastes[a]

Process Step	MIP			RTD	
	1A. No Action	1B. Surface Barrier	1C. *In Situ* Grouting	2A. Targeted RTD[b]	2B. Maximum RTD[b]
1. Burial Site Characterization	√	√	√	√	√
2. *In Situ* Grouting (ISG) for Subsidence Control		√		√	√
3. ISG for Subsidence Control and Contaminant Immobilization			√	√	
4. Excavate, Retrieve, and Segregate Buried Wastes				√	√
5. *Ex Situ* Treatment (e.g., Compaction)				√	√
6. Package Retrieved Wastes				√	√
7. Intermediate Storage of Retrieved and Packaged Wastes and On-site Disposal of non-TRU Wastes and Contaminated Soil				√	√
8. Surface Barrier Selection, Preparation, and Emplacement		√	√	√	√
9. Long-term Stewardship Activities	√	√	√	√	√
10. Off-site Shipment and Disposal at WIPP				√	√

a. Two basic alternatives have been identified for dispositioning DOE buried wastes: 1) manage the waste in place (MIP) or 2) retrieve, treat, and dispose (RTD) the wastes.

b. The two options associated with the retrieve, treat, and dispose (RTD) alternative include A) targeted retrieval of Rocky Flats Plant (RFP) TRU wastes where ISG for contaminant immobilization is performed in non-retrieval areas or B) full or maximum retrieval of RFP TRU wastes from all waste areas. ISG is thus only needed for subsurface stabilization in the maximum RTD case for the SDA.

SDA: Management Flow Diagrams

As indicated in Table 12, all process steps were not required for each remedial alternative. The characteristics of the buried waste site and the extent to which remediation must be exercised impacted the decision logic and thus the risks associated with the remedial actions. To evaluate these considerations, *management flow diagrams* were developed for remedial alternatives. A management flow diagram for a remedial alternative consists of the general process steps that must be completed—and the order in which they are undertaken—to provide a protective final state for the buried wastes.

Exhibit 5. Generic Task Listing for Potential SDA Remedial Alternatives

1. Burial Site Characterization

1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site

1.2 Complete analysis of historic, current, and planned remedial activities

1.3 Complete conceptual site model(s) for the SDA

2. *In Situ* Grouting (ISG) for Subsurface Stabilization

2.1 Determine performance criteria and requirements for ISG based on performance standards

2.2 ISG development and treatability testing (including necessary planning and Quality Assurance/Quality Control)

2.3 Install ISG equipment

2.4 Grout designated areas to stabilize subsurface (against subsidence) prior to surface barrier installation—it is assumed that an enclosure will not be needed for this process step

2.5 Dismantle ISG equipment, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

2.6 Dispose ISG equipment under the surface barrier

3. *In Situ* Grouting (ISG) for Subsurface Stabilization and Contaminant Immobilization

3.1 Determine performance criteria and requirements for ISG based upon relevant waste acceptance criteria, performance standards, and future land-use decisions

3.2 ISG development and treatability testing (including necessary planning and Quality Assurance/Quality Control)

3.3 Install In Situ Grouting equipment and enclosure

3.4 Grout selected areas (e.g., soil vault rows) to immobilize subsurface contamination prior to surface barrier installation

3.5 Assuming same equipment can be used, dismantle, move, and install ISG equipment (but not enclosure) to those areas requiring stabilization against subsidence

3.6 Grout needed areas to stabilize subsurface (against subsidence) prior to surface barrier installation

3.7 Dismantle ISG equipment and enclosure, test for contamination, and decontaminate selected equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

3.8 Dispose ISG equipment and enclosure under the surface barrier

4. Excavate, Retrieve, and Segregate Buried Waste

4.1 Identify appropriate retrieval methods (and assume no additional testing required)

4.2 Determine extent to which buried wastes must be retrieved based on relevant waste acceptance criteria, performance standards, future land use decisions, and possible future legal decisions

4.3 Plan and manage retrieval of buried waste (including preparation of work plans, safety analyses, and other pertinent reviews and activities as well as obtaining any necessary permits)

4.4 Excavate soil overburden and store soil

4.5 Install retrieval equipment for selected contaminated waste areas

4.6 Retrieve wastes from selected areas (noting that spent fuel or analogous materials or pyrophoric or unstable materials may be discovered that must be handled specially)

4.7 Excavate soil underburden (if used)

4.8 Segregate retrieved material into TRU and non-TRU (e.g., low-level and mixed low-level waste) fractions where any specially-handled material will be segregated further

4.9 Temporarily store retrieved and segregated wastes and soil

4.10 Back fill areas from which wastes have been retrieved by initially interring the excavated overburden (and assuming fill material will come from the same borrow area used for surface barrier emplacement)

4.11 Dismantle retrieval equipment and facilities, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

4.12 Dispose retrieval equipment and appropriate facilities under the surface barrier

5. *Ex Situ* Treatment (e.g., Compaction)

5.1 Determine *Ex Situ* Treatment requirements and methods based upon performance standards

5.2 Develop *Ex Situ* Treatment technology and perform treatability studies (including necessary planning and Quality Assurance/Quality Control)

5.3 Construct necessary *Ex Situ* Treatment facilities and install equipment

5.4 Perform *Ex Situ* Treatment on retrieved and segregated wastes and soil (if needed)

5.5 Dismantle *Ex Situ* Treatment equipment and necessary structures, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

5.6 Dispose *Ex Situ* Treatment equipment and necessary structures under the surface barrier

Exhibit 5, Continued

6. Package Retrieved Wastes and Soils

- 6.1 Install packaging equipment if needed
- 6.2 Transfer treated wastes to packaging facility
- 6.3 Package non-TRU low-level and mixed low-level wastes and soils for on-site storage
- 6.4 Package any TRU wastes for shipment to the Waste Isolation Pilot Plant (WIPP)
- 6.5 Special Materials (e.g., spent fuel, pyrophoric materials, etc.) will be handled on a case-by-case basis

7. Intermediate Storage of Retrieved and Packaged Wastes and On-site Disposal of non-TRU Wastes and Contaminated Soil

- 7.1 Construct or identify necessary intermediate storage facilities
- 7.2 Store wastes prior to disposal (e.g., there is a 225-day wait period following final packaging before a drum can be certified for transport to WIPP)
- 7.3 Plan and manage the waste transfer from storage to the original burial site location
- 7.4 Transfer non-TRU wastes and contaminated soil from storage to original burial location for disposal—note that overburden and equipment internment were described in previous steps

8. Surface Barrier Selection, Preparation, and Emplacement

- 8.1 Define performance criteria and requirements for surface barrier emplacement based upon relevant waste acceptance criteria, performance standards, and future land-use decisions
- 8.2 Prepare work plans and safety analyses and obtain necessary permits (including those for borrow area)
- 8.3 Determine type of barrier required based upon performance criteria, requirements, and other relevant information—the Idaho Site Preliminary Evaluation of Remedial Alternatives called for the modified RCRA Subtitle 'C' cap used at the Idaho Site CERCLA Disposal Facility although an evapotranspiration (ET) cap is currently favored and should be protective for arid to semi-arid conditions. For example, RCRA Subtitle 'C' cap has been installed at selected areas in the Oak Ridge Bear Creek Burial Grounds
- 8.4 Prepare the burial site for surface barrier installation including grading and construction of necessary containment buildings and structures
- 8.5 Install surface barrier over the original burial site (which may be a phased installation depending upon closure of any currently operating waste disposal activities) and transport necessary fill material from the designated borrow area

9. Long-term Stewardship Activities for the Original Burial Site

- 9.1 Determine long-term monitoring, maintenance, and institutional controls (e.g., physical and administrative land-use restrictions) needed to ensure that residual buried contamination will be left in a protective state based upon, in part, future land use decisions and possible failure mode scenarios
- 9.2 Implement long-term monitoring (including sampling and analyses) and institutional controls
- 9.3 Routine maintenance, repair, and replacement
- 9.4 Non-routine maintenance, repair, and replacement

10. Off-site Shipment and Disposal at the Waste Isolation Pilot Plant (WIPP)

- 10.1 Plan and manage the waste shipments (including carrier/conveyance designation, preparing necessary plans for the route and security, coordinating the shipment with DOE and State/Local Governments, preparing the Hazardous/Radiological Shipment manifests, and performing the transportation Health Physics survey)
- 10.2 Load TRU Waste Packages into Appropriate Carrier
- 10.3 Load Appropriate Carriers on Appropriate Conveyances
- 10.4 Transport TRU Waste in Appropriate Conveyances to WIPP via road and/or rail
- 10.5 Off-load TRU wastes at the WIPP
- 10.6 Store TRU wastes at the WIPP
- 10.7 Dispose of TRU wastes at the WIPP

Fortunately, like the task list information, the management flow information can be modularized to a great extent for reuse. The management flow diagrams by process step are provided in Figure 18. Management flow diagrams for the remedial alternatives were constructed by connecting various component flow diagrams (from Figure 18) into overall management flow diagrams. An example of an overall management flow diagram for the manage-in-place remedial alternative is illustrated in Figure 19. It is interesting to note that, because of the fairly linear nature of the process steps (in that few of the steps could be performed in parallel), the process steps themselves can be conceived of in terms of the *pinch-points* represented by the process steps.

Management flow diagrams were considered so important to the description and transparency of the remedial process that they were used as the basis for other important risk analysis elements. These elements included the hazard and gap analyses (and tabulations) and *risk flow diagrams*. These risk elements will be described in the sections to follow.

The management flow diagram can be used to evaluate the SDA as a whole, or, applied to sub-areas of the SDA to evaluate which areas or "hot-spots" warrant specific targeted remedial actions. The manage-in-place alternatives, with the exception of the "No Action" option, can be described by the single diagram shown in Figure 19. This alternative was based on the assumption that no wastes would be retrieved in the future. The management flow diagram for the retrieve, treat, and dispose (RTD) alternative is constructed in a similar fashion and is provided in Figure 20.

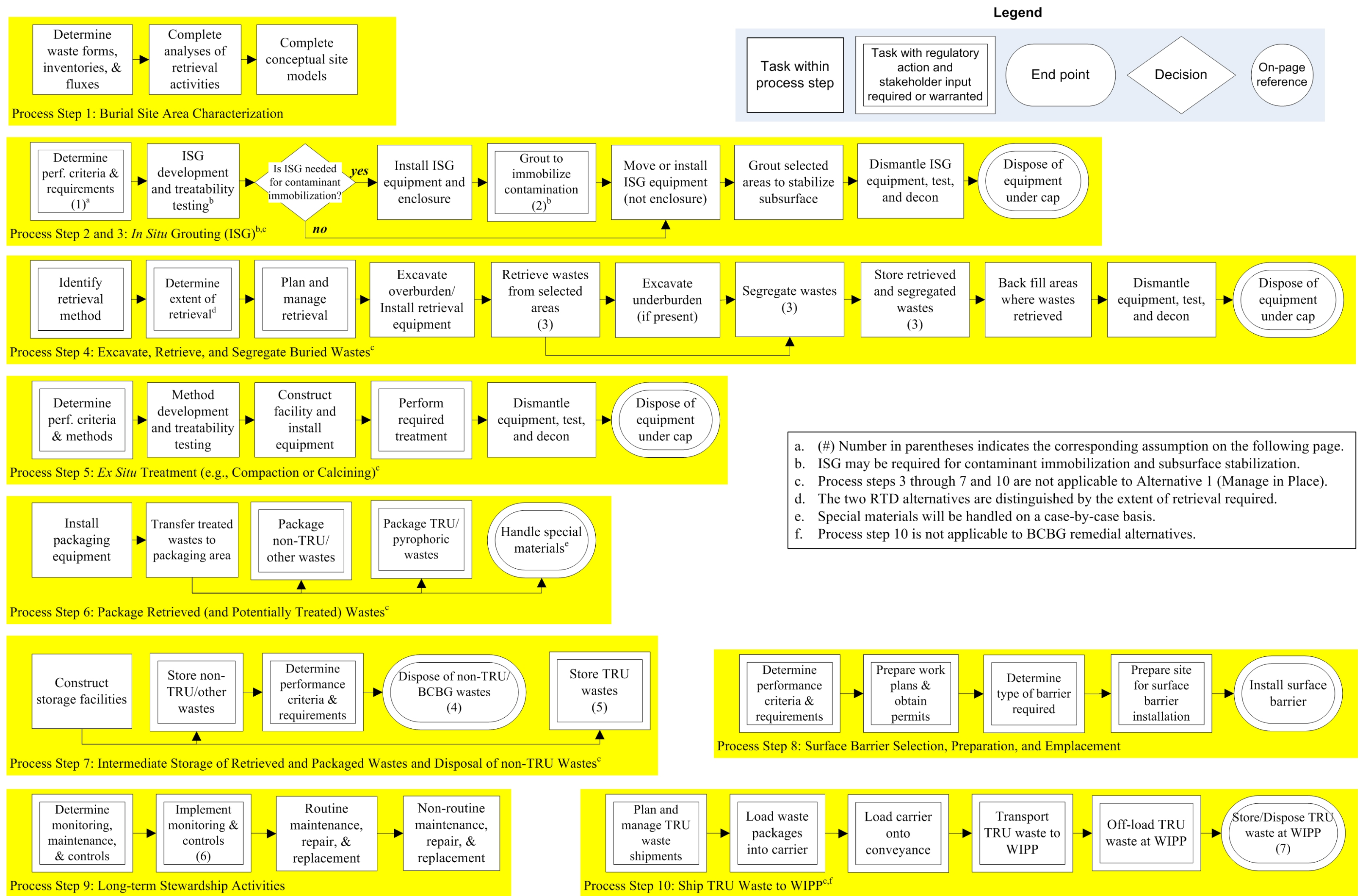

Figure 18. Component Process Flow Diagrams by Remedial Process Step used to Generate Management Flow Diagrams for Complete Remedial Alternatives.

Assumptions for Figure 18: Management Flow Diagrams by Remedial Process Step

1. Any on-going early remedial actions must be taken into account when defining final performance and acceptance requirements. For example, the impact of the SDA organic contamination in the vadose zone (OCVZ) treatment system is not considered in this analysis. However, continued use of the OCVZ treatment system could reduce groundwater risks associated with VOCs significantly (USDOE-ID 2004b; Zitnik et al. 2002), and thus the analysis should be overpredict risks.
2. The RTD alternatives assume that *in situ* grouting (ISG) will be adequate to immobilize the contaminants of potential concern in selected areas prior to installation of the selected engineered surface barrier. In situ grouting may be required for geotechnical stabilization even though buried wastes will be retrieved.
3. The RTD alternatives assume that facilities will be in place to both retrieve and segregate any wastes retrieved from the burial site. For example, during SDA retrieval activities high-level waste (HLW) or spent nuclear fuel (SNF) may be encountered as well as TRU, low-level, and mixed low-level wastes (Schofield 2002). If HLW or SNF material is encountered, Idaho Site personnel have proposed that it will be removed to a separate cell, grouted, and contained in-place in either a cell or trench (Schofield 2002). However, if this material is high-level waste, it may be required to be disposed of as such, that is, underground, in a deep geologic repository. For the BCBG, the wastes that require segregation are likely unstable, explosive, or pyrophoric in nature (SAIC 1996a; b).
4. The RTD alternatives assume that the appropriate regulatory and physical mechanisms are in place to manage and store low-level waste on-site. For example, the low-level and mixed low-level wastes may be emplaced near the center of the SDA prior to capping (Zitnik et al. 2002). All BCBG wastes will be disposed of on-site.
5. The RTD alternatives assume that there will be a facility that can store the packaged TRU containers until they can be shipped to WIPP.
6. Both the MIP ad RTD alternatives assume that there are both the regulatory and funding mechanisms in place to support the appropriate long-term stewardship of the burial site after capping.
7. The RTD alternative may require that there will be sufficient capacity at WIPP to store the TRU waste retrieved from the burial site (and then treated and packaged).

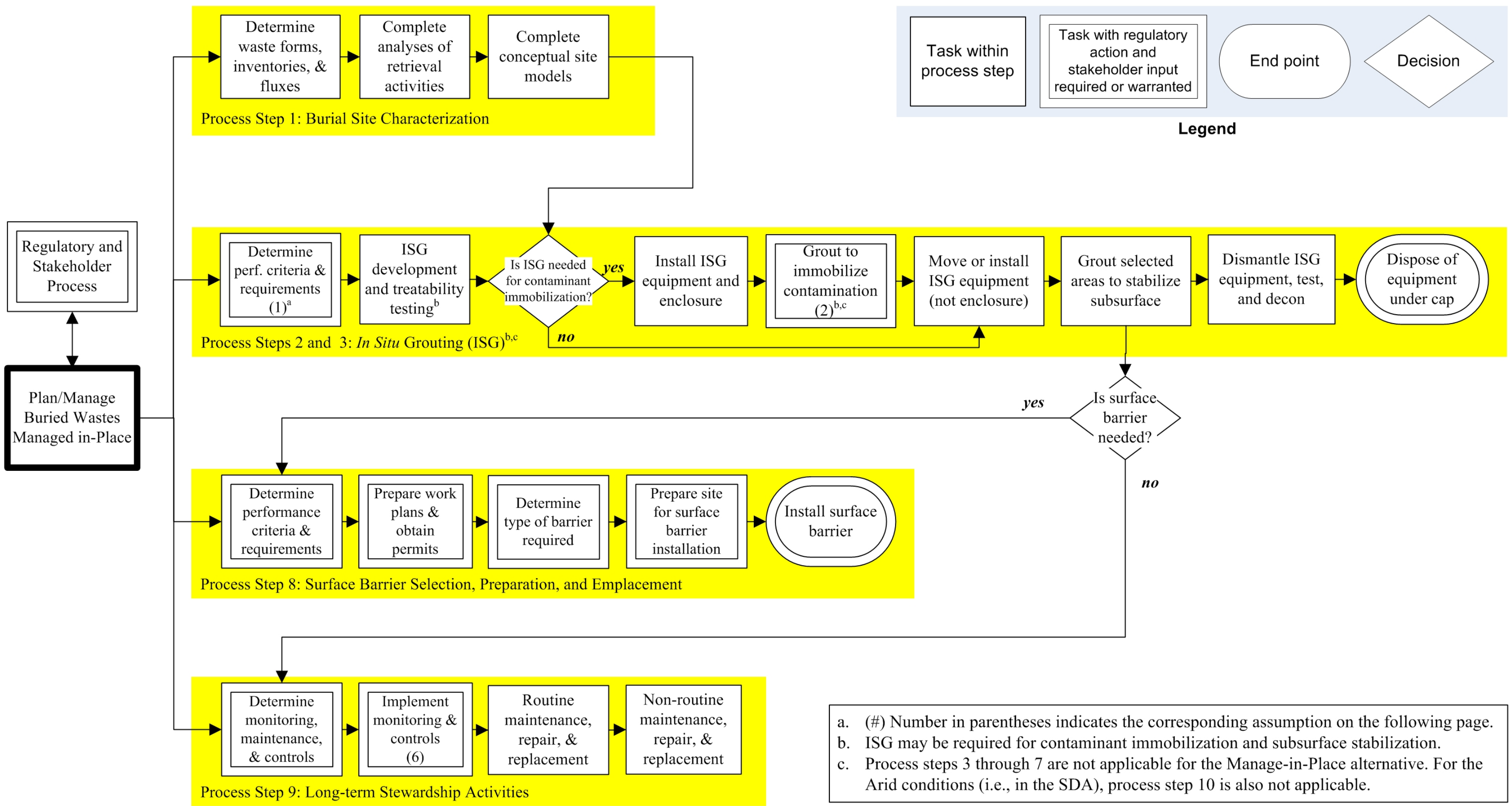

Figure 19. Management flow diagram for the SDA Manage-in-Place Alternatives (excluding the "No Action" Alternative). Assumptions refer to those for the Modules provided in Figure 18.

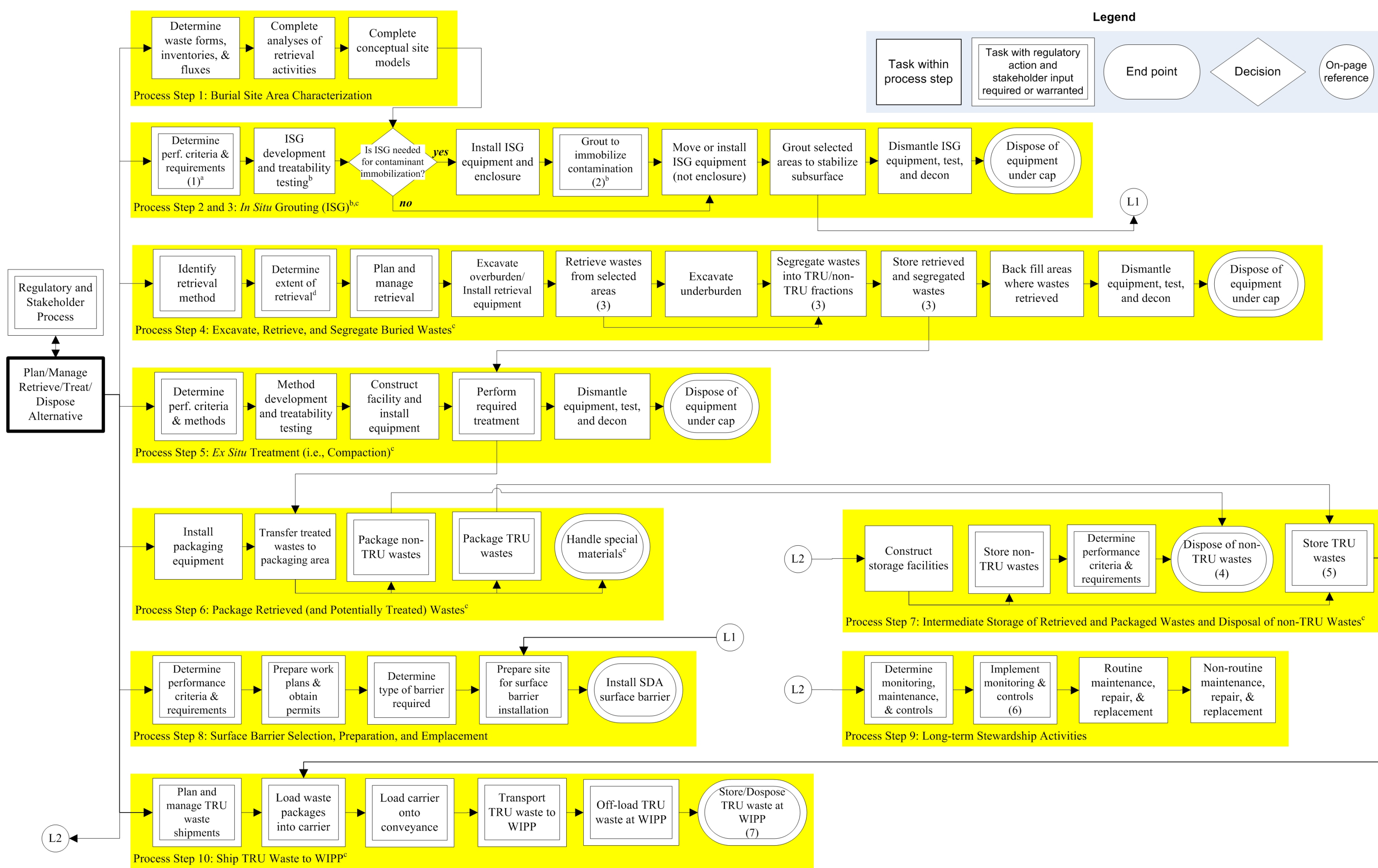

Figure 20. Management flow diagram for the SDA Retrieve, Treat, and Dispose Alternatives. Assumptions refer to those for the Modules provided in Figure 18.

SDA: Integrated Elements of the Remedial Alternative Risk Evaluation

As described in Chapter III, the next step in the risk analysis methodology would be to perform hazard and gap analyses for the various proposed remedial alternatives. Then risk flow diagrams would be developed to indicate the sequence of steps that have the potential to pose significant human health risks to workers or the general public. However, from experience using the risk analysis framework to evaluate risks for the SDA, it was apparent that a great deal of duplication of effort would be needed if these steps were executed sequentially instead of in an integrated or parallel fashion.

Remedial action evaluations including the hazard and gaps analyses and development of risk flow diagrams were performed in parallel using expert judgment and readily available information. These general elements of risk were based on the results of an evaluation of the potential risks and uncertainties associated with the process steps comprising a remedial alternative.

The process step evaluation can be conceptualized as shown in Figure 21 where the tasks comprising the process step are analyzed for significant hazards and uncertainties. An integrated remedial action conceptual site model (CSM) is also developed. The integrated CSM for remedial actions (i.e., Figure 11 in Chapter III) is particular to this risk analysis framework and provides an excellent basis for examining both potential exposure and accident risks in an integrated diagram. The results of the procedure outlined in Figure 21 are lists of significant risks and uncertainties for the remedial alternative that are then used to construct the corresponding risk flow diagrams.

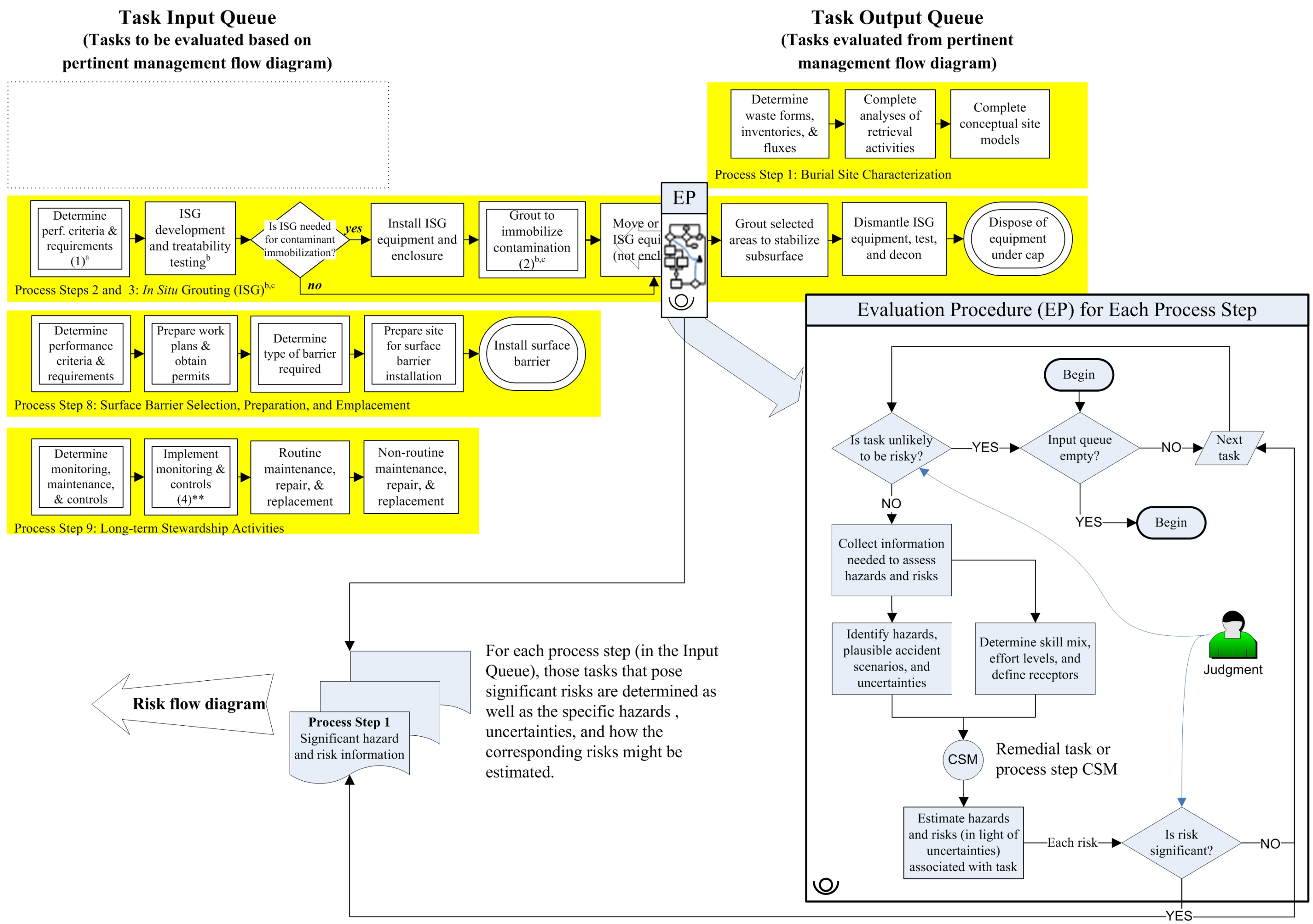

Figure 21. Conceptualization of the Evaluation Procedure for Process Steps to Generate the List of Significant Hazards and Risks for Input to the Risk Flow Diagram. The SDA Manage-in-Place Alternative is used for this Example.

SDA: Qualitative Hazard Analysis

Whereas new remedial alternative evaluations should be integrated as suggested in the previous section, the preliminary evaluation of the risks for proposed SDA remedial actions was completed in an earlier phase of this research (Brown et al. 2005). In this analysis, detailed hazard analysis tables were developed for the manage-in-place and retrieval alternatives. For each remedial task, the following information was determined and tabulated in an easy-to-read format:

- the task frequency,
- what can potentially go wrong,
- how likely is the adverse event to occur,
- the severity of the consequences,
- the impacted population,
- the basis for characterizing the risk, and
- the contribution of the subtask to overall risk of the remedial alternative.

The preliminary qualitative evaluation of the risk previously compiled for the SDA (Brown et al. 2005) was updated based on new information[85]. The resulting detailed hazard analysis tables for the SDA are provided in Appendix A. For example, the hazard analysis table for the SDA site characterization step is provided in Table 13 where the categories used in the hazard analyses were defined in the exhibits in Chapter III.

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[85] The changes made have to do with the process steps considered in the analyses and not the results of the original analysis. For example, *in situ* vitrification and thermal desorption are not considered implementable options (Holdren et al. 2007) and thus their results have been omitted.

Alternative 1: Manage in Place
1A. No Action Option

Table 13. Hazard Evaluation for Manage-in-Place Alternative, No Action Option (1A)

1. BURIAL SITE CHARACTERIZATION							
Task	**Task Frequency**	**What can go wrong? (Failure mode event)**	**How likely is it? (Event probability)**	**What is the severity of the consequences?**	**Who is the impacted population?**	**What is the risk evaluation basis?**	**Overall Contribution to Risk**
1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site[a]	Occasional[b]	• Construction-related traumatic injury • Radiological uptake via dust inhalation • Toxic VOC uptake via inhalation • Dose from external radiation • Heat stress or hypothermia	• Possible • Unlikely • Possible • Possible • Possible	• Critical • Critical • Critical • Marginal • Critical	• Worker • Worker • Worker • Worker • Worker	Judgment and similar activity	• Significant • Low • Significant • Low • Significant
1.2 Complete analysis of remedial activities[c]	Occasional[b]	• Office hazards not considered[d]	• Not considered	• Not considered	• Not considered	Judgment and similar activity	• Not considered
1.3 Complete conceptual model(s) for the burial site	Occasional[b]	• Office hazards not considered[d]	• Not considered	• Not considered	• Not considered	Judgment and similar activity	• Not considered
TASKS 2 THROUGH 8 ARE NOT APPLICABLE							

a. There is an on-going integrated probing project in the SDA to identify the extent of contamination. This is will include site preparation, surveys and mapping, probehole installation and testing, and sampling and data collection (Miller 2003)

b. "Occasional" in this context refers to an activity that is conducted a single time, however, over a long period of time.

c. There are hazards associated with the Pit 4 Accelerated Retrieval (USDOE-ID 2004a) and Beryllium Block Grouting (Lopez 2004; Lopez and Schultz 2004) Projects. However, the tasks associated with these projects have been omitted because they are common to all alternatives and will be completed before and regardless of what remedial alternative is selected.

d. Office hazards (e.g., carpal tunnel syndrome, tripping, slipping, etc.) are not usually considered significant nor do they relate to chemical or radioactive exposure and thus are not considered.

SDA: Summary of the Major Hazards

Based on the preliminary hazard analysis performed in an earlier phase of this research (Brown et al. 2005), the most significant hazards were from the *in situ* grouting (ISG) process step. *In situ* grouting can be used to immobilize subsurface contaminants and/or stabilize the subsurface against subsidence. The ISG step includes a subtask (described below) that appears to be both *probable* (in terms of likelihood) and *severe* (in terms of consequences) as defined in Exhibit 1 in Chapter III.

The intent of process design and implementation is to mitigate hazards and minimize unacceptable risks to the extent practical. However, all possible hazards and risks cannot be completely mitigated, and, as a result, adverse outcomes may still occur. Therefore, the identification of risks is important not only for process selection, but also to carry out the intended remedial actions as safely as possible. The hazards most likely to be problematic for the SDA are described below.

Failure of High-Pressure Grout System Resulting in Projectiles or Grout Release and Injuries (In Situ Grouting). According to the *Preliminary Documented Safety Analysis* for *in situ* grouting (ISG) in the SDA (Abbott and Santee 2004), a failure of the high-pressure grout system is *anticipated* during grouting operations that would result in projectiles or grout release and possible worker injury or fatality[86]. An ISG system similar to that planned for the SDA failed during tests at the Idaho Site and generated a projectile that injured a worker (Abbott and Santee 2004). No radioactive or hazardous material is used

[86] According to Table 3-9 in the preliminary safety analysis for *in situ* grouting in the SDA (Abbott and Santee 2004), the likelihood category is *anticipated* and the consequence category is *moderate*. *Anticipated* means the event is expected to occur during the lifetime of the facility and is the most probable likelihood category in the safety analysis. *Moderate* means that there is likelihood of on-site contamination and worker exposures of up to 100 rem (TEDE) and is the second highest of four consequence categories.

in the grouting system; however, it may be possible that contaminated grout under pressure is transported to the surface. Furthermore, a "failure could generate a projectile or release high-pressure grout with sufficient energy to cause a fatality" (Abbott and Santee 2004). Thus it appears reasonable to classify this failure event as both *probable* and *severe* based on the categories defined in Chapter III. Any impacts would be restricted to the site. The hazards associated with ISG would be compounded if the technique is used for both contaminant immobilization and subsurface stabilization.

Injuries and Exposure due to Excavation and Related Material-Handling Activities (Retrieve, Treat, and Dispose Alternatives). In addition to the ISG process steps, steps employed in the retrieve, treat, and dispose (RTD) alternative have at least one hazard considered to be *high-risk*[87]. The consequences from these hazards tend to be either traumatic injuries from excavation-related or tote-bin handling activities or exposure due to containment system failure or disturbance of contaminated soil. For example, the *Excavate, Retrieve, and Segregate Buried Waste* process step described in Appendix A poses the following three *high-risk* hazards:

- Contaminated soil removal resulting in radiological/toxic chemical exposure
- Loaded tote-bin dropped outside confinement area releasing radioactive material
- Cave-in occurs during excavation operation and buries a worker or worker is otherwise injured during excavation operations.

Although none of the above hazards are deemed *probable* with *severe* consequences, the fact that there are *three high-risk* hazards highlights the potential difficulties in retrieving and handling wastes from the SDA.

[87] *High-risk* hazards are defined as those from events with likelihood/consequence combinations deemed as 1) *probable* and either *critical* or *severe* or 2) *possible* and *severe* (using the definitions in Chapter III).

Failure of Long-Term Stewardship (Manage-in-Place Alternative). The risks to the general public associated with managing buried wastes in-place depend largely on the effectiveness of long-term stewardship activities. Failure of long-term stewardship may result in any of the following: site intrusion, inappropriate land or natural resource use, population encroachment, or contamination of the underlying aquifer. Each of these failure mechanisms has the potential to impact a large number of receptors.

Necessary long-term stewardship activities will have three primary components: maintaining performance of the engineered containment systems; maintaining institutional controls to prevent intrusion, encroachment, and inappropriate land or natural resource use; and effective monitoring strategies for both engineered systems and institutional controls. For monitoring strategies to be effective, they must provide warning of system degradation before failure occurs. For engineered containment systems, this implies monitoring the integrity of caps and moisture and contaminant movement in the vadose zone. This type of monitoring should be construed as preemptive, rather than the basis for a regulatory point of compliance. For institutional controls, regular evaluation of the effectiveness of these controls is needed. Public acceptance of these measures will largely depend on the credibility of DOE and the financial and legal mechanisms established for insuring long-term stewardship.

SDA: Qualitative Uncertainty and Gap Analysis

The nature of the baseline and short-term risk assessments (Holdren et al. 2006; Holdren et al. 2007; Holdren et al. 2002; Schofield 2002) performed by Idaho Site personnel for the SDA indicated that there are uncertainties and gaps in knowledge that must be addressed prior to completing a comprehensive analysis of the risks posed by disposition. A detailed analysis of uncertainties and gaps in knowledge was performed as part of a previous research effort (Brown et al. 2005) and the pertinent information[88] is provided in Appendix A. An example of the uncertainty results for the SDA site characterization is provided in Table 14 using definitions from Exhibit 3 in Chapter III.

SDA: Summary of the Key Uncertainties Relevant to All Remedial Alternatives

The uncertainties and knowledge gaps that were considered to be of highest priority for resolution are described in this section on an overall, as well as a process-specific, basis. Key information gaps included those that were both *critical* (from a safety standpoint) and *large* (indicating little or no information was available) based on the definitions provided in Chapter III.

Geospatial Distribution of Wastes and Waste Forms. The inventories and geospatial distributions of the contaminants of potential concern are highly uncertain, and they drive both the evaluation of risk and remedial alternatives. Knowledge of the specific location of the risk driving contaminants is required to evaluate the effectiveness of proposed remedial actions. For example, if nitrates originally present in waste packages are now widely dispersed in the disposal area and vadose zone, waste retrieval actions would be ineffective in reducing the risk associated with nitrate contamination. Similar concerns exist with respect to the fission products. In addition, knowledge of the geospatial distribution of both risk-driving contaminants and wastes that potentially can lead to high radiation doses to workers is needed to achieve protection of remedial workers.

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[88] As indicated above, the *in situ* vitrification and thermal desorption process steps are no longer considered implementable by Idaho Site personnel (Holdren et al. 2007) and this information is omitted.

Alternative 1: Manage in Place
1A. No Action Option

Table 14. Gap Analysis for Manage-in-Place Alternative, No Action Option (1A)

1. BURIAL SITE CHARACTERIZATION					
Task	**What information is missing?**	**How important?**	**How large a gap?**	**Sources**[a]	**Comment**
1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site	• Potential for facilitated plutonium transport through the vadose zone • Presence and location of spent fuel or similar high-activity material • Saturated zone contaminant transport properties and model validity • Vadose zone contaminant transport properties and model validity • Geospatial distribution of contaminants and waste forms • Physical and chemical forms • Release mechanisms and rates • Infiltration rate into burial site • Locations to insert probes to determine extent of contaminant migration	• Critical • Critical • Important • Important • Critical • Inconsequential[b] • Inconsequential[b] • Inconsequential[b] • Important	• Large • Large • Intermediate • Large • Large • Intermediate • Intermediate • Intermediate • Intermediate	• (Batcheller and Redden 2004) • STRE, PERA, 2nd Addendum • (INEEL 2005) • (INEEL 2005) • ABRA • ABRA • ABRA • ABRA • (Miller 2003; Salomon 2004)	The gaps in knowledge, particularly those relating to the presence and location of spent fuel (or analogous) material and possible facilitated plutonium transport, are high risk/large gap that can lead to significant risks to workers and the general public.
1.2 Complete analysis of remedial activities	• Pit 4 Accelerated Retrieval Project • Extraction of organic contaminants in the vadose zone (OCVZ)[c] • On-going low-level and mixed low-level waste disposal operations	• Inconsequential • Important • Inconsequential	• Large • Large • Intermediate	• (USDOE-ID 2004a; Wooley 2004) • (Housley 2004; USDOE-ID 1994) • IRA, ABRA	The tasks related to Pit 4 and OCVZ tend to reduce risks further and thus can be omitted to provide a reasonable bounding case; however, there are on-going low-level and mixed low-level waste disposal activities that will increase the inventory.

a. STRE is the Short-term Risk Evaluation (Schofield 2002), IRA is the Interim Risk Assessment (Becker et al. 1998), PERA is the Preliminary Evaluation of Remedial Alternatives (Zitnik et al. 2002), ABRA is the Ancillary Basis for Risk Analysis (Holdren et al. 2002), 2nd Addendum is the Addendum to RI/FS (Holdren and Broomfield 2004), and RBES is the Draft Idaho Site Risk-Based End State Vision document (USDOE-ID 2004b).

b. These gaps are considered to have a small impact on the overall task because reasonable assumptions (e.g., solubility-limited releases) can be made to provide reasonably conservative estimates for the contaminant fluxes from the burial site.

c. Since 1996, soil vapor extraction has been employed to remove organic contamination in the vadose zone (OCVZ) below the Subsurface Disposal Area (SDA). The vadose zone has been contaminated by volatile organic compounds migrating from the buried wastes (Housley 2004).

Presence and Location of High-Level Waste, Spent Fuel, or Similar High-Activity Material. Schofield (2002) stated that "[d]uring retrieval activities, high-level waste and possibly spent nuclear fuel may be encountered [in the SDA]." Zitnik et al. (2002) suggested that "[w]aste buried in the SDA before 1970 contains small quantities of irradiated fuel material…." Finally, Holdren and Broomfield (2004) indicated:

> "Some shipments to the SDA contained waste that is similar to spent nuclear fuel or high-level waste and may exhibit some characteristics of these waste forms" where "[the above] assumption reflects information developed since the First Addendum through review of waste shipment and inventory records. Waste similar to spent nuclear fuel or high-level waste may require specific attention in modeling (e.g., contaminant inventories and release and transport mechanisms) and in analyzing alternatives (e.g., safety issues related to exposure rates, potential security concerns, and interference with remedial technologies such as retrieval and ISG)." (Holdren and Broomfield 2004)

Therefore, there is uncertainty concerning the presence, type, and amount of high-level waste (HLW) and spent nuclear fuel (SNF) or similar material in the SDA. Special requirements are necessary when handling these types of material due to high activities.

If HLW or SNF was buried in the SDA, then this waste must be disposed of as such and the operations needed for their management added to the RTD risk evaluation. For example, the Nuclear Waste Policy Act of 1982 (as amended) specifies that high-level radioactive waste must be disposed of in a deep geologic repository (USPL 1982). Furthermore, the 1995 Settlement Agreement[89] among the State of Idaho, DOE, and the U.S. Department of the Navy stated that "DOE shall treat all high-level waste currently at [the Idaho Site] so that it is ready to be moved out of Idaho for disposal by a target date of 2035."

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[89] The 1995 Settlement Agreement among the State of Idaho, DOE, and Department of the Navy was located found at http://www.deq.state.id.us/inl_oversight/contamination/settlement_agreement_entire.cfm (accessed March 13, 2008).

Baseline Risk Assessment ("No Action" Alternative). Before examining risks for remedial alternatives, baseline risks and uncertainties should be evaluated from available information so there is a basis for comparison. For baseline conditions, no *additional* remedial actions[90] are assumed to be taken to treat the wastes buried in the SDA (Lopez 2004; Lopez and Schultz 2004; USDOE-ID 2004b). However, environmental monitoring, maintenance, and institutional controls would be implemented.

All baseline risk analyses performed for the wastes buried in the SDA pose "unacceptable long-term risk to human health and the environment" (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002). Sufficient information was available for many contaminants of potential concern (COPCs) to estimate risks to receptors quantitatively using best inventory estimates supplemented by one-at-a-time sensitivity and uncertainty analyses. However, risks for other contaminants of interest without sufficient inventory and/or toxicological information were addressed qualitatively.

Because mathematical modeling was required to assess the risks posed in any proposed scenario, a significant part of the gap analysis can be conceptualized in terms of the modeling effort required. Several methodological, release, transport, and fate aspects must be adequately addressed before the model can be useful for assessing risks. In general, these aspects for a given COPC include the

- ability of the model used to adequately describe the true situation,
- release and source term (including flux to the surface or subsurface media),
- surface transport through both air and water,

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[90] For example, some waste will be retrieved during the Pit 4 Accelerated Retrieval Project (USDOE-ID 2004a). Beryllium blocks, which became radioactive after being used as reflectors in Idaho Site test reactors, were buried in the SDA. These blocks were grouted in-place to immobilize C-14; however, this does not preclude future retrieval if necessary (Lopez 2004; Lopez and Schultz 2004). There is also an on-going removal of organic contamination from the vadose zone using vacuum vapor extraction.

- subsurface transport including both vadose and groundwater zones,
- exposure mechanisms, and
- receptors and impacts.

Many of these aspects are described by the conceptual site model (CSM) (ASTM 1995; USDOE 2003); an example of the baseline CSM for the SDA was shown in Figure 17.

Several modeling aspects are likely to be either uncertain or unknown (i.e., gaps). A previous research effort provided a detailed analysis of the known gaps in knowledge for baseline conditions including not only modeling tasks but also the long-term stewardship and institutional control activities (Brown et al. 2005). The results of the detailed gap analysis for the SDA were provided in Appendix A. An example of the gap analysis was provided in Table 14. The information gaps for SDA baseline conditions that were both *critical* (from a safety standpoint) and *large* (i.e., little if anything was known) were

- Potential for facilitated plutonium transport and
- Presence and location of spent nuclear fuel or similar high-activity material

These knowledge gaps were relevant to all SDA remedial alternatives in this research.

Potential for Facilitated Plutonium Transport. Estimates indicate that more than a metric ton of plutonium may have been buried in the SDA (Sentieri 2002; Sentieri 2003a; b; 2004)[91]. Under oxidizing and consolidated conditions, plutonium has been found to be

[91] Large quantities of fissionable material were buried in the SDA. Thus one might consider the potential for a criticality accident to be both *high-risk* and a critical information gap. However, the preliminary safety analyses (Abbott and Santee 2004; Abbott 2003; Santee 2003) for proposed remedial actions indicate that any conceivable criticality accident would have a frequency less than once in 10,000 years. The conclusion from these safety analyses are supported by criticality analyses for the SDA (Sentieri 2002; Sentieri 2003a; b; 2004), which indicated that criticality accidents were either extremely unlikely or not credible.

fairly immobile in the environment. However, plutonium can move significant distances when present as fine particulates, chelated, or under reducing conditions or if it is present as, transformed into, or attached to a colloid (Flury and Harsh 2003).

If plutonium is of small particle size (i.e., less than 1 μm), it can be transported as a colloid (Batcheller and Redden 2004) or it can form an intrinsic colloid by polymerization (Flury and Harsh 2003). As much as 75 kg of the plutonium buried in the SDA may be of the size that could form colloids[92]. Plutonium can form aqueous complexes with organic materials, such as EDTA, which is present in the SDA (Bates 1993; Becker et al. 1998). When reduced to a soluble form, plutonium can sorb to other, natural "colloidal" material such as zeolites or clays, which are ubiquitous in the subsurface (Flury and Harsh 2003). When in colloidal form or attached to colloidal particles, plutonium can move through the subsurface unretarded (Flury and Harsh 2003).

Colloid concentrations in natural subsurface systems tend to be low resulting in limited facilitated colloidal transport, and resulting colloids may not be stable over the long distances necessary to reach the aquifer. The stability of colloids and distances over which facilitated transport can occur is a strong function of specific field conditions. There is insufficient data to test or calibrate colloid transport models[93]. Thus facilitated transport of plutonium cannot be ruled out and presents a potentially significant gap in information that must be addressed to accurately assess the risks posed by the wastes buried in the SDA.

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[92] The 95% upper confidence limit on the colloid-sized plutonium in the SDA is 4.9% of 1100 kg (best estimate) to 1500 kg (upper bound) (Batcheller and Redden 2004).

[93] Limitations include: few appropriate laboratory (or column) studies, insufficient field data that can be used to test or calibrate colloid transport models, and no link made between the laboratory and field characteristics (Batcheller and Redden 2004). Often the column studies used to estimate the transport parameters do not consider preferential flow pathways.

Other Important Knowledge Gaps. There were other knowledge gaps that, even though not classified as *critical* and *large*, were still important. These included:

- Carcinogenic and non-carcinogenic risks were estimated using standard exposure parameters at the downgradient Idaho Site boundary for a hypothetical 100-year institutional control (IC) period (Holdren and Broomfield 2004). In the remedial investigation/feasibility study (RI/FS) for the SDA (Holdren and Broomfield 2004), only a single acute well-drilling, intruder scenario was evaluated. A residential groundwater ingestion scenario was evaluated to 10,000 years at the SDA boundary after completion of the 100-year IC period (Holdren and Broomfield 2004). Additional exposure scenarios and pathways need to be evaluated that are more relevant to projected, future land use and local values.

- It is uncertain whether *in situ* grouting (ISG) will have to be used to immobilize COPCs in selected areas of the SDA. There was also uncertainty as to the extent of ISG needed for contaminant immobilization and subsurface stabilization.

- Because the final applicable or relevant and appropriate requirements (ARARs) for the SDA will not be defined until the Record of Decision is finalized, the regulatory requirements for SDA remedial actions may change from those assumed in this report.

- A significant amount of fill, or "borrow," material will be required to backfill areas in the SDA, prevent subsidence, and complete the surface barrier.[94] Under the most favorable circumstances, this material can be taken from the spreading areas less than a mile from the SDA. However, it is possible that this material will have to be moved (by truck) from a location more than 64 km (40 miles) from the SDA (Schofield 2002). This activity would substantially increase risks and costs for this alternative because as many as 159,000 truck loads (at 17 m^3 per load) of material would be required if a RCRA type "C" cap is employed.

SDA: Summary of Key Process-Specific Uncertainties and Gaps in Knowledge

Possible Future Legal Decisions and Resulting Actions (Retrieve, Treat, and Dispose Alternatives). These remedial alternatives involve retrieving buried wastes from the SDA, segregating and treating the retrieved wastes, and disposing the non-TRU wastes at the

[94] Zitnik (2002) states that "[p]reliminary assessments indicate that suitable materials are available from borrow areas on and off the INEEL. However, this project would require extensive excavation within the designated areas. For example, approximately 3.5 million yd^3 [or over 159,000 truck-loads] of silt loam materials would be required to complete construction of the [RCRA-type] cover. Assuming this was retrieved from a single pit with an average extraction depth of 20 ft, it is projected that the pit surface would cover approximately 100 acres [or approximately the size of the SDA]."

Idaho Site and the TRU wastes at Waste Isolation Pilot Plant (WIPP). As described in Chapter III, there are two options associated with this alterative that differ in the extent to which buried wastes would be retrieved.

The extent to which buried waste must be retrieved from the SDA is controversial and may ultimately be the result of legal decisions concerning the disposition of Rocky Flats Plant (RFP) waste buried in the SDA before 1970. The 1995 Settlement Agreement between the State of Idaho, DOE, and the Department of the Navy states that[95]

> "DOE shall ship all transuranic waste now located at INEL, currently estimated at 65,000 cubic meters in volume, to the Waste Isolation Pilot Plant (WIPP) or other such facility designated by DOE, by a target date of December 31, 2015, and in no event later than December 31, 2018."

The Idaho Site and DOE indicated that the approximately 65,000 m^3 of transuranic waste referred to in the Settlement Agreement is that stored in the Transuranic Storage Area and did not include waste buried in the SDA. However, in *United States of America v. Dirk Kempthorne* (*USA v. Kempthorne*, Civil Case No. 91-0054-S-EJL), the judge ruled that "all" meant *both stored and buried* transuranic and *all* high-level wastes. DOE appealed this ruling to the U.S. Court of Appeals for the Ninth Circuit (NAS 2005). In December 2004, the decision was reversed and the case remanded to district court indicating that all evidence (including the source of the 65,000 m^3 estimate) must be considered[96]. However, in March 2008 the Ninth Circuit Court of Appeals upheld the original decision requiring DOE to remove all transuranic waste from the SDA.

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[95] The 1995 Settlement Agreement among the State of Idaho, DOE, and Department of the Navy was located found at http://www.deq.state.id.us/inl_oversight/contamination/settlement_agreement_entire.cfm (accessed March 13, 2008).

[96] The reversal can be found at: http://www.ca9.uscourts.gov/coa/memdispo.nsf/pdfview/120304/$File/03-35470.PDF (accessed March 13, 2008).

Decisions have gone back and forth on this issue since these decisions. The outcome of such legal actions in the future may dictate the extent to which wastes buried in the SDA will have to be retrieved and/or immobilized and the concomitant risks. The Idaho Site has little control over such legal matters; however, because such actions may dictate the remedial actions that must be taken, these actions also directly influence the risks associated with the SDA remediation.

SDA: Suggestions for Information Gap Resolution

Uncertainties, even large ones, can be managed using well-established statistical techniques. However, missing information is much more difficult to capture in a risk (or any other) analysis. Therefore, the uncertainty and gap analysis in this research was focused on the missing information that might significantly impact risk estimates.

The path forward for resolving knowledge gaps is part of the on-going CERCLA process, and it is hoped that this research could be useful as input to that process. For example, an assessment of the size and impact of each knowledge gap indicated in this research could be used to determine resolution order. Information concerning risk reductions achievable, residual risks, inventories, geospatial distributions, waste forms, release, fate, and transport of contaminants in the SDA should be collected.

Any additional data collection effort should be focused on risk-drivers. An assessment of the potential to uncover high-radiation material (especially spent nuclear fuel or similar material) should be performed early in the process. Contingencies should be developed to prepare for the possibility that a future court decision mandates the extent to which buried transuranic wastes from the Rocky Flats Plant must be retrieved.

SDA: Risk Flow Diagrams

The hazards and gaps for the SDA remedial alternatives were previously evaluated as illustrated in Appendix A (Brown et al. 2005). These results were converted into corresponding risk flow diagrams. Typically, the risk flow diagram generation should be integrated with the results of the detailed hazard and gap analyses as suggested in the methodology in Chapter III and illustrated in Figure 21.

The appearance of the risk flow diagrams developed in previous research (Brown et al. 2005) were changed. The task flow information that served as the structural basis for the previous version of these diagrams duplicated much of the information in the corresponding management flow diagrams. Thus the management flow diagram was selected as the basis of the risk flow diagram as well to enhance transparency. The risk flow diagrams for the SDA manage-in-place and retrieve, treat, and dispose alternatives are illustrated in Figure 22 and Figure 23, respectively.

The risk flow diagram indicates by relative degree of shading and hatching the maximum hazard classification for a task. The overall contribution to risk for the process step, which is summarized in the next section and Table 15, is represented by the degree of shading and hatching. The *in situ* grouting and long-term stewardship steps were those that posed the greatest life-cycle risks from dispositioning the buried wastes in the SDA.

Conceptual site models (CSMs) for remedial process steps and final disposition states were also developed. A complete set of CSMs for SDA process steps and final states was developed in a previous effort (Brown et al. 2005). Generic CSMs for buried waste sites are provided in Chapter III and can be use as the basis for future such analyses.

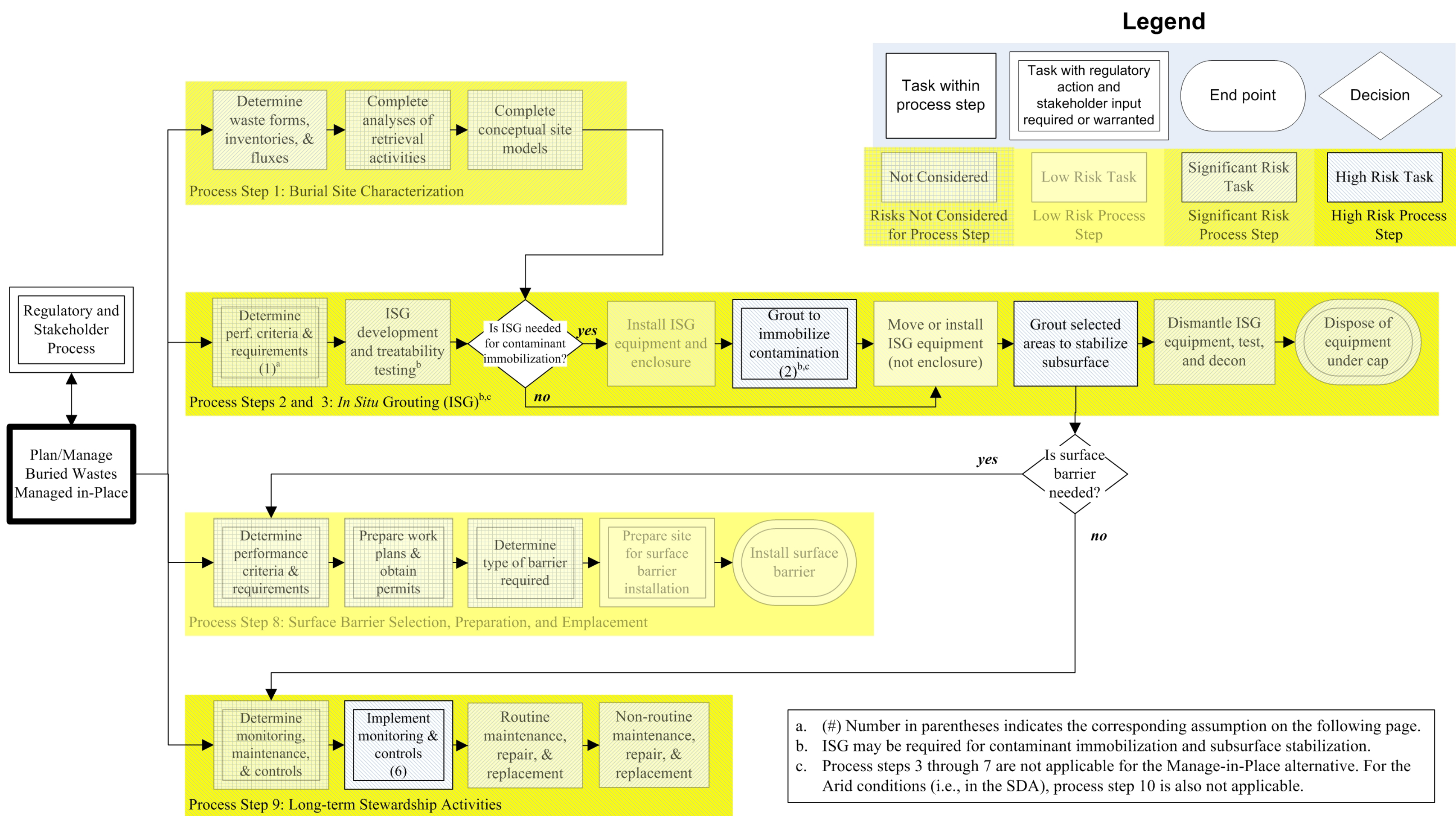

Figure 22. Risk flow diagram for the SDA Manage-in-Place Alternatives. The degree of transparency and type of hatching represent the degree of risk for the process step. Assumptions refer to those for the Modules provided in Figure 18.

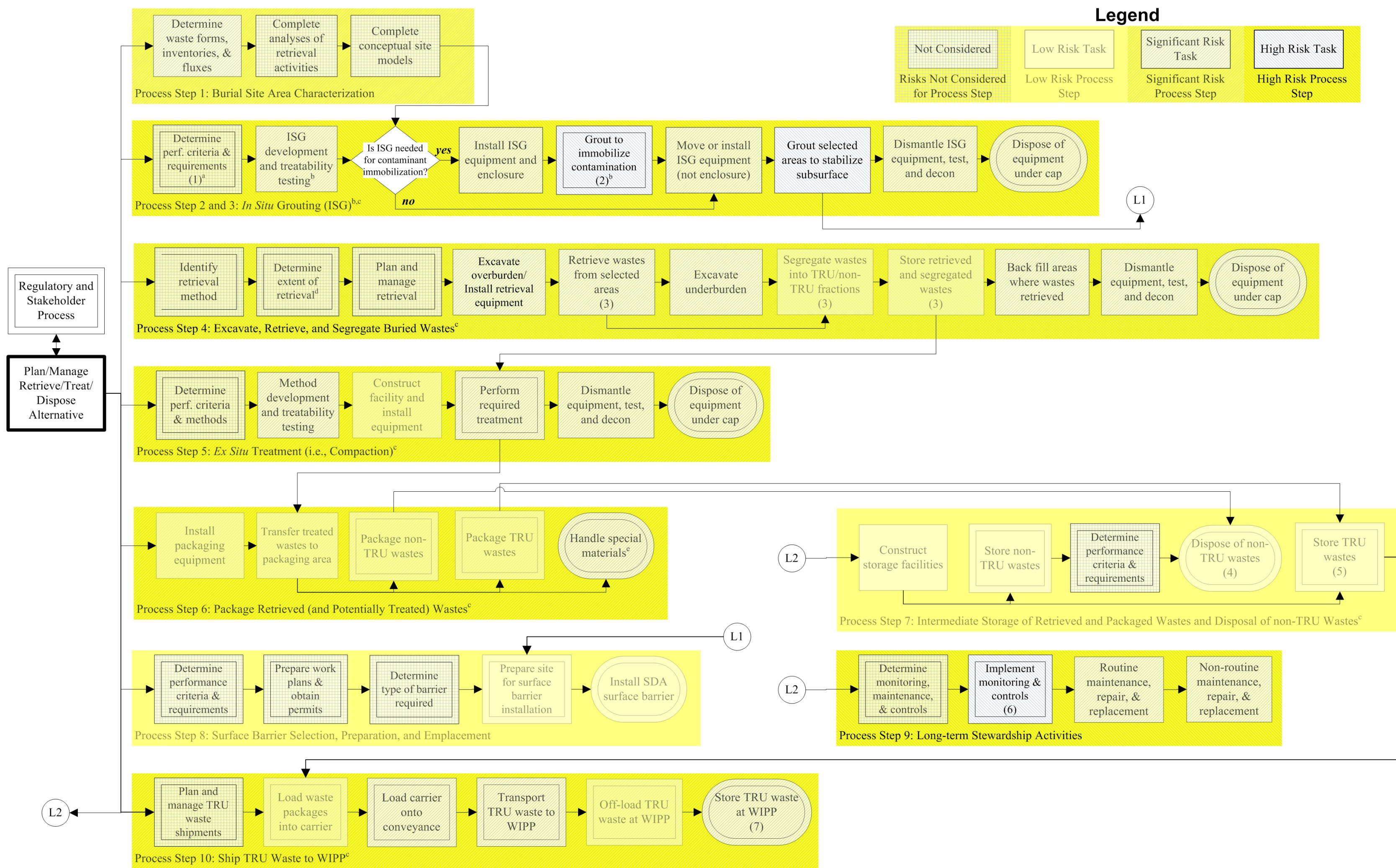

Figure 23. Risk flow diagram for the SDA Retrieve, Treat, and Dispose (RTD) Alternatives. The degree of transparency and type of hatching represent the degree of risk for the step. Assumptions refer to those for the Modules provided in Figure 18.

SDA: Integrated Gap and Hazard Analysis Summary

From discussions with Idaho Site stakeholders, one of the most important sources of information for potential remedial alternatives for a buried waste site was the integrated gap and hazard analysis summary provided in Table 15. This summary was derived from the detailed hazard and gap analyses (Appendix A) used to define the risk flow diagrams in the previous section.

The most significant hazards were from the *in situ* grouting (ISG) process steps. *In situ* grouting was required to immobilize subsurface contaminants of concern and stabilize the subsurface against subsidence. As indicated by Brown et al. (2005), this process step presented significant hazards in that it included a subtask that appeared to be both *probable* and *severe* (in terms of consequences) based on the definitions in Chapter III. Furthermore, the risks presented by the ISG step were compounded by the fact that there were also high priority information gaps associated with knowing where the grouting would be needed (for either subsidence control or contaminant immobilization).

SDA: Preliminary Comparison of Remedial Alternatives

Integrating the hazard and gap analyses allows for a qualitative ranking of proposed remedial alternatives in terms of risk, human health, environmental, and programmatic factors as dictated in the risk analysis framework and methodology described in Chapter III (and especially Phase 2C in Figure 10). When assessed for these factors and in the context of the numerous assumptions and value judgments made when assessing risks and uncertainties, the proposed remedial alternatives for the SDA buried wastes can be ranked.

Table 15. Summary of the Most Important Human Health Risks and Knowledge Gaps for the SDA Remedial Alternatives

Process Step	1A. Baseline/No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the consequences?	Who is impacted?	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C)
1. Burial Site Characterization	√	√	√	√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Potential for facilitated plutonium transport • Possible presence and location of high-rad material • Geospatial distribution of wastes and waste forms	Significant (0,3,2,2)
2. ISG for Subsurface Stabilization		√		√	√	• Failure of high-pressure grout system resulting in projectiles or grout release and injuries	• Probable	• Severe	• Worker	• No *high-priority* gaps	High (1,4,13,1)
3. ISG for Immobilization and Stabilization			√	√	√	• Failure of high-pressure grout system resulting in projectiles or grout release and injuries	• Probable	• Severe	• Worker	• Geospatial distribution of wastes and waste forms	High (2,4,22,1)
4. Excavate, Retrieve, and Segregate Buried Wastes				√	√	• Contaminated soil removal resulting in radiological/toxic chemical exposure • Loaded tote-bin dropped outside confinement area releasing radioactive material • Traumatic injury (e.g., cave-in occurs during excavation)	• Probable • Probable • Possible	• Critical • Critical • Severe	• Worker • Worker • Worker	• Future legal decisions and resulting actions • Geospatial distribution of wastes and waste forms	Significant[d] (0,11,37,3)

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Exhibit 1 (Chapter III).

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Exhibit 3 (Chapter III).

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. The numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The fact that there are three *high-risk* hazards associated with this process step makes it highly significant from a risk perspective so much so that is may need to be considered a *high-risk* step even though there are no events that would be considered both *probable* (in terms of likelihood) and *severe* (in terms of consequences).

Table 15, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[b]	How likely is it?	What are the consequences?	Who is impacted?	Highest Priority Information Gap(s)[c]	Overall Contribution to Risk[d] (H,S,L,N/C)
5. Ex Situ Treatment				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• No *high-priority* gaps	Significant (0,5,13,1)
6. Package Retrieved Wastes and Soil				√	√	• Containment/ventilation system failure and resulting exposure to radiological and toxic substances	• Possible	• Severe	• Worker	• No *high-priority* gaps	Significant (0,2,17,0)
7. Intermediate Storage and On-Site Disposal				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• No *high-priority* gaps	Low (0,0,5,0)
8. Surface Barrier Selection, Prep, and Emplacement		√	√	√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• No *high-priority* gaps	Low (0,0,8,3)
9. Long-term Stewardship (LTS)	√	√	√	√	√	• Failure of LTS (MiP Options) • Failure of LTS (RTD Options)	• Probable • Probable	• Severe • Critical	• Public • Public	• Geospatial distribution of wastes and waste forms	High (1,7,8,1)
10. Off-Site Shipment and WIPP Disposal				√	√	• Injuries of operation of heavy equipment	• Probable	• Critical	• Worker	• No *high-priority* gaps	Significant (0,3,11,1)

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Exhibit 1 (Chapter III)..

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Exhibit 3 (Chapter III)..

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. The numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The fact that there are three *high-risk* hazards associated with this process step makes it highly significant from a risk perspective so much so that is may need to be considered a *high-risk* step even though there are no events that would be considered both *probable* and *severe* (in terms of consequences).

For the manage-in-place (MIP) alternative, there were no remedial options that would be considered *low-risk*. For example, baseline conditions (i.e., represented by the "No Action" alternative) were considered *high-risk* because nothing would be done to isolate subsurface contamination from percolating water. Even though long-term monitoring, maintenance, and institutional controls would be instituted, this option must be considered *high-risk*. The *high-risk* hazard for this option was associated with failure of long-term stewardship actions. Such activities would be required for any proposed remedial option; however, a surface barrier would be installed for all but the "No Action" option and thus the risks would be lower unless a catastrophic barrier failure occurs.

All remedial alternatives considered for the SDA would be classified as *high-risk* primarily because they all employed one or more of the *high-risk* process steps (i.e., *in situ* grouting and long-term stewardship) illustrated in the integrated summary in Table 15. However, this fact does not preclude a rank-ordering amongst the various possible remedial alternatives. For example, significant programmatic risk was associated with the waste retrieval options represented by the retrieve, treat, and dispose (RTD) alternative. Another factor that impacts the overall risk would be how extensively a *high-risk* process step might be employed. A number of the remedial options employed *in situ* grouting (ISG) for both subsurface stabilization and contaminant immobilization.

Significant risks were also associated with the retrieval process employed in the RTD alternative. Assuming that risk increases with increasing waste retrieval, one would assume that the maximum Rocky Flats Plant (RFP) transuranic (TRU) waste retrieval option would present more risk than the targeted RFP TRU waste retrieval option. Based on these assumptions, one *possible* rank-ordering of the SDA remedial options in terms of risk would produce the following:[97]

No Action >> Maximum RTD > Targeted RTD > ISG > Surface Barrier

However, the above rank-ordering is subjective and based on numerous assumptions concerning how to compare the *qualitative* risk classifications in this report and other value judgments[97]. Furthermore, not all of the options (i.e., No Action) would be considered satisfactory from a regulatory perspective (where an alternative must be protective of human health and the environment). A quantitative assessment of risks or different remedial requirements may produce a different rank-ordering than that above.

SDA: Interpreting the Overall Risk Classification

The results of applying the risk analysis framework to this point suggest no clear, ideal remedial choice for dispositioning the wastes buried in the SDA. One could qualitatively rank-order the remedial options based on assumptions concerning relative risks and remedial requirements; however, additional programmatic and regulatory information or analysis of risks and uncertainties would be required to select an acceptable remedial choice. Many of these issues are being addressed as part of the Idaho Site CERCLA process for the SDA (Holdren et al. 2006; Holdren et al. 2007).

Risk analyses that define risk reduction and residual risks for proposed remedial actions provide very informative inputs to the decision-making process. A step in this

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[97] The rank-ordering is based on the following assumptions: 1) risk increases with increased waste retrieval, 2) employing *in situ* grouting (ISG) for both subsurface stabilization and contaminant immobilization is higher risk than when ISG is used for only subsurface stabilization, and 3) not containing the wastes using a surface barrier would have the potential to impact by far the greatest number the public, which would overwhelm any reduced worker risks.

direction was taken in the SDA feasibility study where targeted retrieval actions were considered (Holdren et al. 2007). Contaminants of concern should be characterized based on their geospatial distributions and waste forms in the context of the overall SDA disposition, which allows risk drivers to be identified and managed appropriately. For example, retrieval of buried RFP TRU wastes (including the majority of TRU contaminants, perhaps one-half of the nitrate source, and much of the volatile organic material that has not migrated) would have little impact on many other contaminants of concern (e.g., Tc-99, Sr-90, etc.) and the short-term exposure risks presented by the SDA.

Results of this phase of the framework indicated that the lowest risk option for the SDA would be removal of the highly mobile contaminants (i.e., volatile organics and nitrates) that pose significant short-term risks followed by containment using a surface barrier (Brown et al. 2005). Containment will reduce the flux of water to the remaining contaminants and reduce their ability to migrate to the environment. However, the estimated risk associated with this option is strongly dependent on the effectiveness of long-term stewardship activities. Failure of these activities may result in any or all of the following: site intrusion, inappropriate land use, population encroachment, contamination of the sole-source aquifer, etc. Each of these failure mechanisms has the potential to impact a large number of people in the (possibly distant) future.

Based on the risk and uncertainty information developed for the SDA remedial alternatives, it appears that a remedial alternative can be selected from the group studied that will satisfy the cleanup goals defined in this research. Because remedial alternatives have been defined to target high-risk areas within the SDA, no additional remedial alternatives need to be considered.

Oak Ridge Bear Creek Burial Grounds (BCBG) Risk Analysis

The risk analysis framework developed for this research was applied to the Bear Creek Burial Grounds (BCBG) on the Oak Ridge Reservation (ORR). This site was selected for evaluation because, when considered in juxtaposition with the Idaho Site Subsurface Disposal Area (SDA), it tended to bracket the types of contaminants, hazards, and conditions expected from Department of Energy (DOE) buried waste sites.

Considerable information exists for potential remedial actions that might be applied to the SDA (Holdren et al. 2007; Schofield 2002; Zitnik et al. 2002). No comparable information was found for the BCBG. Surface barriers have been installed on the BCBG; however, some of the barriers appear to be in areas where the buried waste is periodically inundated by near surface groundwater (SAIC 1996a).

Because of the lack of specific information concerning possible remedial actions for the BCBG, the risk evaluation focused on differences in hazards and uncertainties presented by the sites or that would impact the effectiveness of the remedial actions originally proposed for the SDA. These remedial actions were applied to both sites so the results for the sites could be compared directly.

BCBG: Conceptual Site Model (CSM) Development

The Bear Creek Burial Grounds (BCBG) are located within the Beak Creek Valley, an area mostly contained in the Oak Ridge Reservation as illustrated in Figure 15 approximately 20 miles northwest of Knoxville, Tennessee. The valley is over 10 miles long and runs from the eastern end of the Oak Ridge Y-12 Plant to the Clinch River. As illustrated in Figure 16, multiple individual waste units are located in the valley containing hazardous and radioactive wastes derived primarily from Y-12 Plant operations. Groundwater has been contaminated throughout at least the eastern 3 miles of the valley, including commingled plumes from different sources (SAIC 1996a).

The conceptual site model (CSM) in Figure 24 describes baseline BCBG conditions and potential exposure risks to the general public and workers. Various DOE reports were used as the basis for hazard identification, exposure assessment, and receptor evaluation (SAIC 1996a; b; c; d; e; f). Because the BCBG has been in operation for many decades and has contaminated extensive areas of the environment including soil, surface water, and groundwater, some "early" remedial actions (i.e., surface barriers and leachate collection) have been taken to treat problem areas of the site (SAIC 1996a).

BCBG: Qualitative Baseline Uncertainty and Gap Analysis

The primary uncertainties that impact the ability of the risk assessor to estimate risks to potentially impacted receptors are the inventories and geospatial distributions of the radioactive and hazardous contaminants. Whereas uranium wastes dominate those that were buried in the BCBG, the waste form may be a much more important issue from a safety perspective. Although there are exposure concerns related to uranium and other wastes in the BCBG, unstable, explosive, and pyrophoric materials were also buried that, if they require retrieval, would pose significant and unique hazards to remedial workers. For example, pyrophoric uranium exposed to air presents not only an exposure hazard (from inhalation of uranium in the fire plume) but also a risk from potential thermal burns. These hazards are compounded by the fact that these materials were buried in many areas in the BCBG and their locations are not always well-known.

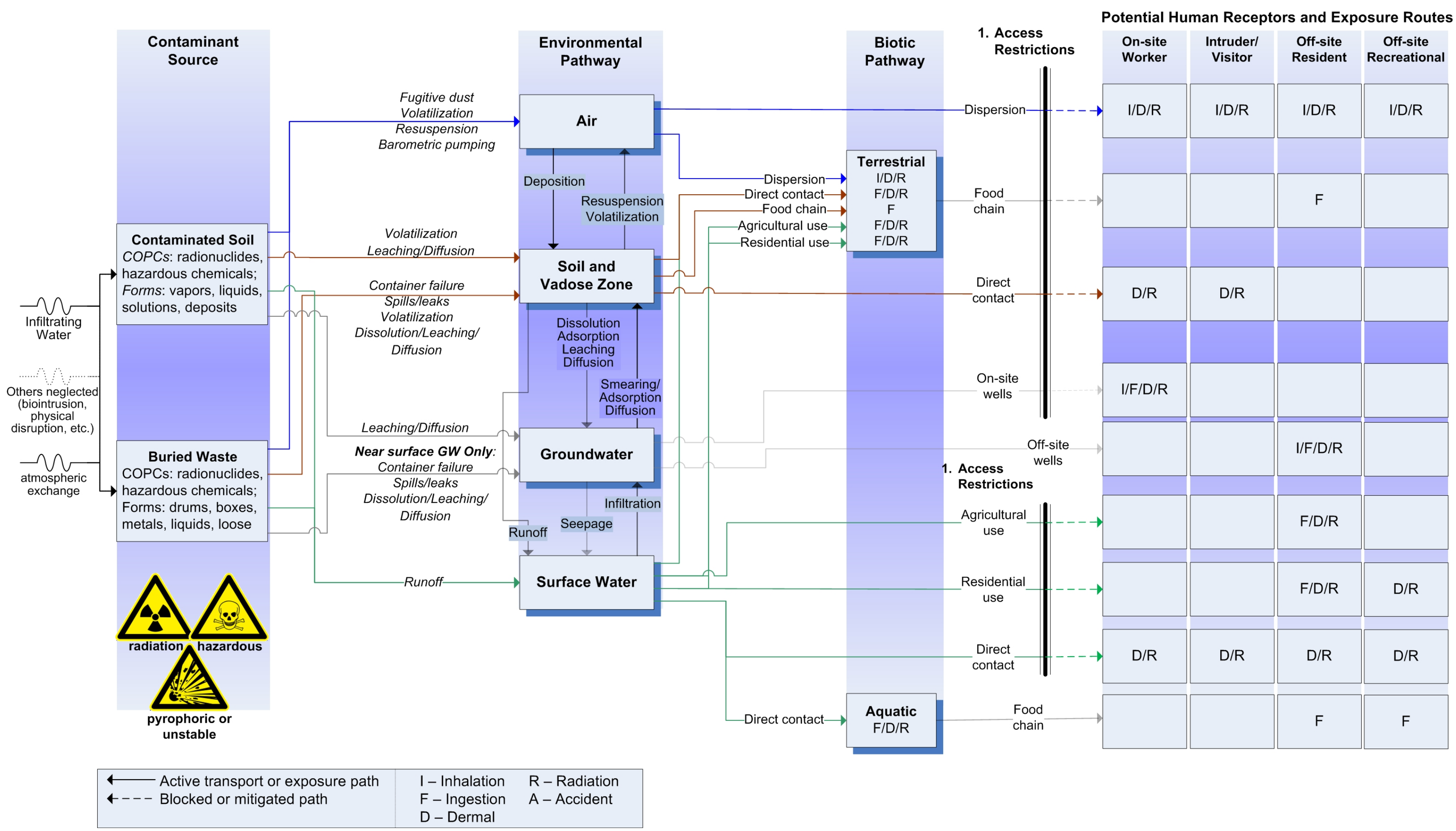

Figure 24. Baseline conceptual site model (CSM) for exposures to chemicals and radionuclides from the Bear Creek Burial Grounds (BCBG) **before any *additional* remedial actions are undertaken**.

Narrative for Figure 24: Baseline Conceptual Site Model (CSM) for the Oak Ridge Bear Creek Burial Grounds (BCBG)

Bear Creek Valley and the Bear Creek Burial Grounds contain several of the primary waste disposal units at the Oak Ridge Reservation (USDOE-ORO 2004). Large volumes of solid hazardous and radioactive wastes (particularly contaminated with uranium from Y-12 operations) were buried in trenches located at the BCBG. Hazardous liquids are known to have been disposed of at various locations including the BCBG. Soils, groundwater, and surface water at each of these sites including the BCBG are known to be contaminated. Contaminants in the BCBG include VOCs and metals in groundwater and VOCs, metals, and radionuclides in surface water, soils, waste materials, and leachates. Organic contamination of environmental media tends to be more widespread than inorganic and radionuclide contamination.

The following barrier (or step taken to mitigate impacts) is shown in Figure 24:

1. The Oak Ridge Reservation has restricted access to prevent intrusion by the public (USDOE-ORO 2004).

For very long-lived and reasonably mobile radionuclides that are already in the environment surrounding the BCBG, a pathway exists from the BCBG to the Clinch River via the surface water pathway. The only potential “barriers” that exist are decay, dispersion, and dilution for the radionuclides and dispersion and dilution for the nitrates and heavy metals. Two potential barriers are not shown on the BCBG CSM because of issues with their effectiveness. RCRA Subtitle 'C' barriers have been installed over some areas of the BCBG that appear to experience periodic inundation with near-surface groundwater. Leachate collection has also been initiated in areas of the BCBG that may help reduce the flux of contaminants to the environment (SAIC 1996a).

Contaminant migration has been detected in the environment around the BCBG (SAIC 1996a; c). Results from environmental monitoring provide a snapshot, albeit an uncertain one, of the extent of contaminant migration (but only in those locations sampled). The baseline uncertainty information used in this analysis was taken from the most recent remedial investigation report (SAIC 1996a; b; c; d; e; f).

BCBG: Qualitative Baseline Risk Evaluation

Extensive studies have been completed to evaluate the potential risks associated with the wastes buried in the BCBG (SAIC 1996a; d). Solid and liquid wastes were disposed of in a series of unlined trenches (SAIC 1996a). Uranium-contaminated wastes including pyrophoric metallic uranium fines, chips, and cuttings dominate the material disposed in the BCBG with a total estimated uranium mass of 19×10^6 kg (40×10^6 lb). In addition to the pyrophoric uranium metal, unstable materials including reactive and explosive materials were buried.

Liquid waste disposal has resulted in volatile organic compound (VOC) contamination in groundwater that may have reached depths of almost 200 m (600 ft). Contaminants in the BCBG include VOCs and metals in groundwater and VOCs, metals, and radionuclides in surface water, soils, waste materials, and leachates. Organic contamination of environmental media tends to be more widespread than inorganic and radionuclide contamination. The peak baseline risks for the SDA contaminants were previously provided in Table 10. Without additional information, the baseline peak risks for the BCBG wastes described in this table will be used to evaluate the risk drivers for BCBG buried wastes.

BCBG: Preliminary Overall Assessment and Cleanup Goals

The overall cleanup goals for the BCBG were simple to define. Like the SDA, the BCBG is being cleaned up under the auspices of CERCLA, and the nine evaluation criteria defined for the SDA were used as the preliminary cleanup goals for the BCBG.

As illustrated in Chapter III, the other important outcomes of Phase 1 include decisions as to whether or not there is sufficient information to require the site undergo remedial action. An affirmative answer was obvious based on the remedial investigation results for the BCBG. The contaminants of potential concern (COPCs) for the BCBG were the same as those being studied in the most recent remedial investigation report (SAIC 1996a). Examples of the BCBG COPCs are provided in Table 10

BCBG: Screening Quantitative Baseline and Residual Risk Evaluations

To determine the extent to which the BCBG must be remedied, quantitative estimates of baseline risks were used to identify contaminants of potential concern and define quantitative acceptance goals to assure that cleanup would be completed to the extent necessary to be protective. The next phase in the risk analysis framework, Phase 2A: *Screening Quantitative BRA and Preliminary Acceptance Goals (PAGs) Definitions* (Figure 9 in Chapter III) describes the process to provide the quantitative baseline risk estimates needed to identify contaminants of concern and define acceptance goals. The remedial investigation reports developed for the BCBG (SAIC 1996a; e) provided more than adequate quantitative baseline human health risk estimates for BCBG site cleanup. No additional updates to the conceptual site models, scenarios, uncertainty analysis, or cleanup goals were deemed necessary.

The final step in the screening quantitative baseline risk analysis phase is to define quantitative assessment goals corresponding to the cleanup goals defined in Phase 1. Because the BCBG is being cleaned up under CERCLA, the basic acceptance goals were those corresponding to the EPA 10^{-6} to 10^{-4} risk criteria for carcinogens or a Hazard Quotient of unity for noncarcinogens or relevant ARARs.

After quantitative estimates of the baseline risks associated with the BCBG have been generated, acceptable remedial alternatives must be identified. Without additional information, remedial alternatives similar to those defined for the SDA in Table 11 were assumed to apply to BCBG disposition. The appropriate remedial alternatives for the BCBG are provided in Table 16.

The manage-in-place options were the same as those for the SDA. The retrieve, treat, and dispose options were also similar; however, the extent of retrieval for the BCBG was based on the hydrologic conditions of the waste areas. Targeted retrieval actions were applied to areas that are perennially inundated with groundwater and extended to those areas that are, at a minimum, inundated during storm events. Retrieved wastes cannot be returned to their original burial sites and are instead moved to an area not impacted by shallow groundwater and inundation.

It was assumed that the residual risk levels obtained from Phase 2B evaluation (Figure 9, Chapter III) corresponded to the most restrictive of either the EPA risk levels or ARARs. No other information (e.g., conceptual site models, uncertainties, scenarios, cleanup goals, etc.) developed to this point required updating. The next phase (Phase 2C in Figure 10, Chapter III) involved determining the life-cycle risks associated with potential remedial alternatives as input to the informed decision-making process.

Table 16. Possible Bear Creek Burial Grounds (BCBG) Disposition Alternatives

	Stabilization	Surface Barrier	*In Situ* Treatment	*Ex Situ* Treatment	On-Site Disposal
1. Manage-in-Place Alternative (3 options)					
1A. No Action option[a]					
1B. Surface barrier with *in situ* grouting (ISG) for stabilization	√	√			√
1C. ISG for immobilization and stabilization with surface barrier	√	√	√		√
2. Retrieve/Treat/Dispose Alternative (2 options)					
2A. Targeted BCBG waste retrieval with surface barrier[b]		√	√	√	√
2B. Maximum BCBG waste retrieval with surface barrier[b]		√	√	√	√

a. This option represents baseline conditions and is required under CERCLA for comparison purposes. Although no further actions would be taken to reduce contaminant mobility, toxicity, or volume, long-term monitoring, maintenance, and institutional controls would be instituted.

b. The retrieval alternatives are based on the hydrologic conditions of the waste areas in the BCBG. The options presented concern targeted retrieval of wastes from areas perennially inundated with groundwater versus maximum retrieval of BCBG wastes in areas that are, at a minimum, inundated during storm events as described in Appendix D. The wastes retrieved would be segregated, treated, packaged, and disposed of on-site. Unstable, explosive, and pyrophoric materials were buried in the BCBG and may be retrieved. If these types of materials are unearthed during retrieval, then additional segregation and storage tasks would be necessary. However, the lack of inventory and quantitative information makes estimating the risks associated with such retrieval efforts difficult.

BCBG: Qualitative Remedial Alternatives Risk Evaluation

Characterization of risk for a remedial alternative requires the identification of adverse events, the likelihood of the adverse event, the severity of the consequences, the temporal nature of the risks, and the affected population. As illustrated for Phase 2C in Figure 10 (Chapter III), qualitative remedial alternative risks are evaluated by completing a set of six steps for each alternative[98]. These steps are described below.

[98] As illustrated previously for the Idaho Site SDA, these steps involve development of a task list, a risk flow diagram and conceptual models, detailed hazard and gap analyses, and an integrated hazard and gap analysis summary.

BCBG: Task List Development

From a preliminary evaluation of the potential risks posed by the remedial alternatives proposed for the BCBG, a great deal of commonality was discovered amongst the various process steps for remedial alternatives for buried waste sites (Brown et al. 2005). The first nine process steps listed in Table 12 for the Subsurface Disposal Area (SDA) appeared to describe the steps needed to complete remedial options for the BCBG. The revised process step table for the BCBG wastes is provided in Table 17.

Table 17. Process Steps Needed to Disposition BCBG Buried Wastes[a]

	MIP			RTD	
Process Step	**1A. No Action**	**1B. Surface Barrier**	**1C. *In Situ* Grouting**	**2A. Targeted RTD[b]**	**2B. Maximum RTD[b]**
1. Burial Site Characterization	√	√	√	√	√
2. *In Situ* Grouting (ISG) for Subsurface Stabilization		√		√	√
3. ISG for Subsurface Stabilization and Contaminant Immobilization			√	√	
4. Excavate, Retrieve, and Segregate Buried Wastes				√	√
5. *Ex Situ* Treatment (e.g., Compaction)				√	√
6. Package Retrieved Wastes				√	√
7. Intermediate Storage of Retrieved and Packaged Wastes and On-site Disposal of Wastes and Contaminated Soil				√	√
8. Surface Barrier Selection, Preparation, and Emplacement		√	√	√	√
9. Long-term Stewardship Activities for the BCBG	√	√	√	√	√

a. Two basic alternatives have been identified for dispositioning the BCBG buried wastes: 1) manage the waste in place (MIP) or 2) retrieve, treat, and dispose (RTD) the wastes.

b. The two options associated with the retrieve, treat, and dispose (RTD) alternative include A) targeted retrieval of wastes in areas perennially inundated with groundwater and *in situ* grouting (ISG) is performed for contaminant immobilization in non-retrieval areas or B) maximum retrieval of wastes from all areas that are, at a minimum, inundated during storm events. ISG is thus only needed for subsurface stabilization in the maximum RTD case for the BCBG.

Exhibit 6 provides the updated set of task lists corresponding to the BCBG process steps in Table 17. For each alternative, the information in Table 17 was used to determine which process steps were involved and then the task lists in Exhibit 6 were used to identify the steps needed to execute the remedial alternative. The order of the process steps was assumed to be the same as that in Table 17. The next step is to define the *management flow diagrams* to more fully and transparently convey the activities required to carry out the remedial alternatives.

BCBG: Management Flow Diagrams

The characteristics of the buried waste site and the extent to which remediation must be exercised impact the decision logic and thus the risks associated with the remedial actions. To evaluate these considerations, *management flow diagrams* were developed for the BCBG remedial alternatives. The manage-in-place remedial options, with the exception of the "No Action" alternative, can be described by the diagram in Figure 19 originally developed for the SDA. The management flow diagram for the retrieve, treat, and dispose (RTD) alternative is provided in Figure 25.

Exhibit 6. Generic Task Listing for Potential BCBG Remedial Alternatives

1. Burial Site Characterization

1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site

1.2 Complete analysis of historic, current, and planned remedial activities

1.3 Complete conceptual site model(s) for the BCBG

2. *In Situ* Grouting (ISG) for Subsurface Stabilization

2.1 Determine performance criteria and requirements for ISG based on performance standards

2.2 ISG development and treatability testing (including necessary planning and Quality Assurance/Quality Control)

2.3 Install ISG equipment

2.4 Grout designated areas to stabilize subsurface (against subsidence) prior to surface barrier installation—it is assumed that an enclosure will not be needed for this process step

2.5 Dismantle ISG equipment, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

2.6 Dispose ISG equipment under the surface barrier

3. *In Situ* Grouting (ISG) for Subsurface Stabilization and Contaminant Immobilization

3.1 Determine performance criteria and requirements for ISG based upon relevant waste acceptance criteria, performance standards, and future land-use decisions

3.2 ISG development and treatability testing (including necessary planning and Quality Assurance/Quality Control)

3.3 Install In Situ Grouting equipment and enclosure

3.4 Grout selected areas to immobilize subsurface contamination prior to surface barrier installation

3.5 Assuming same equipment can be used, dismantle, move, and install ISG equipment (but not enclosure) to those areas requiring stabilization against subsidence

3.6 Grout needed areas to stabilize subsurface (against subsidence) prior to surface barrier installation

3.7 Dismantle ISG equipment and enclosure, test for contamination, and decontaminate selected equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

3.8 Dispose ISG equipment and enclosure under the surface barrier

4. Excavate, Retrieve, and Segregate Buried Waste

4.1 Identify appropriate retrieval methods (and assume no additional testing required)

4.2 Determine extent to which buried wastes must be retrieved based on relevant waste acceptance criteria, performance standards, future land use decisions, and possible future legal decisions

4.3 Plan and manage retrieval of buried waste (including preparation of work plans, safety analyses, and other pertinent reviews and activities as well as obtaining any necessary permits)

4.4 Excavate soil overburden and store soil

4.5 Install retrieval equipment for selected contaminated waste areas

4.6 Retrieve wastes from selected areas (noting that unstable, explosive, and pyrophoric materials may be discovered that must be handled specially)

4.7 Excavate soil underburden (if used)

4.8 Segregate retrieved material into pyrophoric and other fractions where any specially-handled material (e.g., unstable, explosive, etc.) will be segregated further

4.9 Temporarily store retrieved and segregated wastes and soil

4.10 Back fill areas from which wastes have been retrieved by initially interring the excavated overburden (and assuming fill material will come from the same borrow area used for surface barrier emplacement)

4.11 Dismantle retrieval equipment and facilities, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

4.12 Dispose retrieval equipment and appropriate facilities under the surface barrier

5. *Ex Situ* Treatment (e.g., Calcining)

5.1 Determine *Ex Situ* Treatment requirements and methods based upon performance standards

5.2 Develop *Ex Situ* Treatment technology and perform treatability studies (including necessary planning and Quality Assurance/Quality Control)

5.3 Construct necessary *Ex Situ* Treatment facilities and install equipment

5.4 Perform *Ex Situ* Treatment on retrieved and segregated wastes and soil (if needed)

5.5 Dismantle *Ex Situ* Treatment equipment and necessary structures, test for contamination, and decontaminate equipment (where remaining, contaminated equipment will be disposed of by placing under surface barrier)

5.6 Dispose *Ex Situ* Treatment equipment and necessary structures under the surface barrier

Exhibit 6, Continued

6. Package Retrieved Wastes and Soils

- 6.1 Install packaging equipment if needed
- 6.2 Transfer treated wastes to packaging facility
- 6.3 Package treated pyrophoric wastes for on-site storage
- 6.4 Package remaining wastes and soils for on-site disposal
- 6.5 Special Materials (e.g., shock-sensitive, unstable, etc.) will be handled on a case-by-case basis

7. Intermediate Storage of Retrieved and Packaged Wastes and On-site Disposal of Wastes and Contaminated Soil

- 7.1 Construct or identify necessary intermediate storage facilities
- 7.2 Store wastes prior to disposal
- 7.3 Plan and manage the waste transfer (including performance requirements) from storage to the original burial site location
- 7.4 Transfer wastes and contaminated soil from storage to original burial location for disposal—note that overburden and equipment internment were described in previous steps

8. Surface Barrier Selection, Preparation, and Emplacement

- 8.1 Define performance criteria and requirements for surface barrier emplacement based upon relevant waste acceptance criteria, performance standards, and future land-use decisions
- 8.2 Prepare work plans and safety analyses and obtain necessary permits (including those for borrow area)
- 8.3 Determine type of barrier required based upon performance criteria, requirements, and other relevant information. For example, RCRA Subtitle 'C' cap has been installed at selected areas in the Oak Ridge Bear Creek Burial Grounds
- 8.4 Prepare the burial site for surface barrier installation including grading and construction of necessary containment buildings and structures
- 8.5 Install surface barrier over the original burial site and transport necessary fill material from the designated borrow area

9. Long-term Stewardship Activities for the Original Burial Site

- 9.1 Determine long-term monitoring, maintenance, and institutional controls (e.g., physical and administrative land-use restrictions) needed to ensure that residual buried contamination will be left in a protective state based upon, in part, future land use decisions and possible failure mode scenarios
- 9.2 Implement long-term monitoring (including sampling and analyses) and institutional controls
- 9.3 Routine maintenance, repair, and replacement
- 9.4 Non-routine maintenance, repair, and replacement

BCBG: Integrated Elements of the Remedial Alternative Risk Evaluation

The next steps in the risk analysis methodology would be to perform hazard and gap analyses for proposed remedial alternatives. Then risk flow diagrams would be developed to indicate the sequence of steps that have the potential to pose significant human health risks. However, from experience using the risk analysis framework, it was apparent that a great deal of duplication of effort was needed if these steps were executed sequentially instead of in an integrated or parallel fashion.

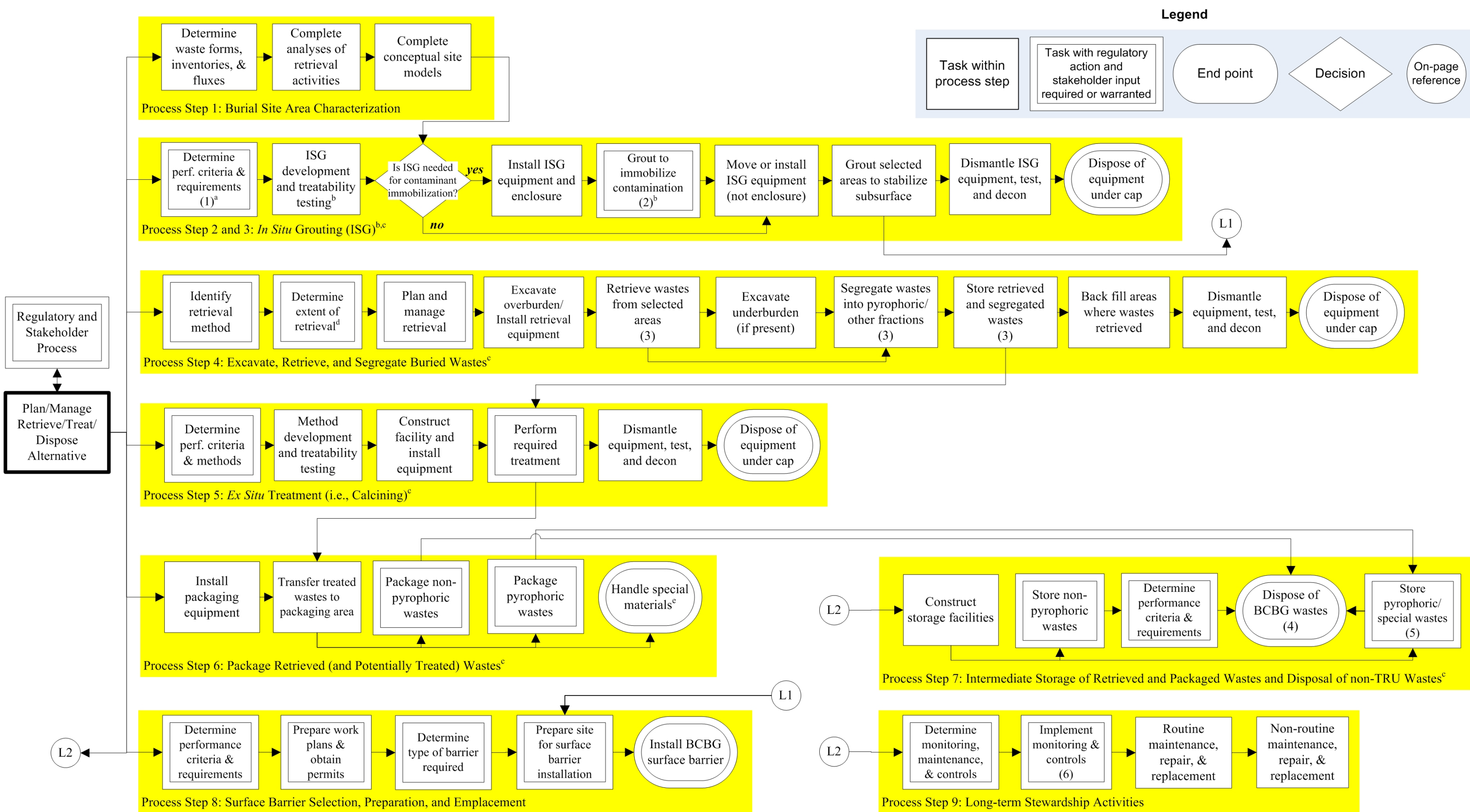

Figure 25. Management flow diagram for the BCBG Retrieve, Treat, and Dispose Alternatives. Assumptions refer to those for the Modules provided in Figure 18.

Remedial action evaluations including hazard and gaps analyses and risk flow diagrams were performed in parallel as shown in Figure 21 where tasks were analyzed for significant hazards and uncertainties. An integrated remedial action conceptual site model (CSM) was also developed as illustrated in Figure 11 from Chapter III. This diagram provided an excellent basis for presenting both potential exposure and accident risks in a single diagram. The results of the procedure described in Figure 21 were lists of significant risks and uncertainties that were then used to construct risk flow diagrams.

BCBG: Qualitative Hazard Analysis

A qualitative evaluation of the risks posed by proposed remedial actions for the Oak Ridge Bear Creek Burial Grounds (BCBG) was completed as illustrated in Appendix B based on available remedial investigation information and the guidelines defined in previous research (Brown et al. 2005). An example of the results of the detailed hazard analysis for the BCBG is provided in Table 18 (using the risk categories defined in the exhibits in Chapter III) for the manage-in-place "No Action" option.

When compared to SDA hazards, the outstanding additional hazards posed by BCBG clean up were associated with the large quantities of unstable, explosive, and pyrophoric materials buried throughout the BCBG (SAIC 1996a). These hazards really came into play during excavation and waste retrieval operations when shock-sensitive materials might be disturbed, incompatible materials mixed, or pyrophoric materials exposed to air. Oak Ridge personnel and their regulators consider the areas containing these materials to be too hazardous to sample; an inventory analysis was conducted to characterize buried wastes in these areas in lieu of sampling and analysis (SAIC 1996b).

Alternative 1: Manage in Place
1A. No Action Option

Table 18. Hazard Evaluation for Manage-in-Place Alternative, No Action Option (1A)

1. BURIAL SITE CHARACTERIZATION							
Task	**Task Frequency**	**What can go wrong? (Failure mode event)**	**How likely is it? (Event probability)**	**What is the severity of the consequences?**	**Who is the impacted population?**	**What is the risk evaluation basis?**	**Overall Contribution to Risk**
1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site[a]	Occasional[b]	• Disturbing unstable or uncovering pyrophoric materials during sampling • Construction-related traumatic injury • Radiological uptake via dust inhalation • Toxic VOC uptake via inhalation • Dose from external radiation • Heat stress or hypothermia	• Probable • Possible • Unlikely • Possible • Possible • Possible	• Severe • Critical • Critical • Critical • Marginal • Critical	• Worker • Worker • Worker • Worker • Worker • Worker	Judgment and similar activity	• High • Significant • Low • Significant • Low • Significant
1.2 Complete analysis of remedial activities	Occasional[b]	• Office hazards not considered[c]	• Not considered	• Not considered	• Not considered	Judgment and similar activity	• Not considered
1.3 Complete conceptual model(s) for the burial site	Occasional[b]	• Office hazards not considered[c]	• Not considered	• Not considered	• Not considered	Judgment and similar activity	• Not considered
TASKS 2 THROUGH 8 ARE NOT APPLICABLE							

a. According to the Bear Creek Valley remedial investigation information, areas likely containing unstable (i.e., shock-sensitive or explosive) and pyrophoric materials were considered too dangerous to characterize via sampling (SAIC 1996b).

b. "Occasional" in this context refers to an activity that is conducted a single time, however, over a long period of time.

c. Office hazards (e.g., carpal tunnel syndrome, tripping, slipping, etc.) are not usually considered significant nor do they relate to chemical or radioactive exposure and thus are not considered.

On the other hand, because of the small masses of plutonium and other similar materials buried in the BCBG, any criticality concerns or those associated with facilitated transport of plutonium were not pertinent to BCBG waste disposition. Because all BCBG waste disposal was assumed to take place on-site (because no TRU waste was buried), those risks associated with transfers of TRU waste to the WIPP would not be applicable.

BCBG: Summary of the Major Hazards

The most significant hazards for disposition of Idaho Site SDA wastes were from the *in situ* grouting (ISG) process step. The results of the preliminary hazard analysis for the BCBG were provided in Appendix B and indicate that ISG presented similar hazards as for the SDA. However, unlike for the SDA, additional unacceptable hazards would be presented if BCBG waste areas containing unstable or pyrophoric wastes were excavated. The unstable or pyrophoric nature of the wastes made even the comparatively simple act of sampling hazardous to workers performing the characterization (SAIC 1996b). The following list describes the hazards most likely to be problematic for the BCBG.

Disturbing Unstable or Uncovering Pyrophoric Materials during Sampling (Site Characterization for Both Alternatives). Large quantities of unstable, explosive, and pyrophoric materials were buried in the BCBG (SAIC 1996a; b). Sampling operations in the areas containing these materials may disturb shock-sensitive materials or expose pyrophoric materials to air resulting unacceptable exposure and injury risks to workers (SAIC 1996a; e). Based on the detailed results in Appendix B, the hazards associated with sampling of BCBG areas that might uncover pyrophoric or disturb unstable materials were classified as *probable* and *severe* based on the definitions in Chapter III.

Failure of High-Pressure Grout System Resulting in Projectiles or Grout Release and Injuries (In Situ Grouting for Both Alternatives). Without specific Oak Ridge information for many of the steps involved in disposition, the BCBG hazard analysis was performed using Idaho Site information to supplement available BCBG information. A brief description of each important hazard is provided here with additional details provided in the corresponding SDA detailed hazard analysis results under the same title.

According to the *Preliminary Documented Safety Analysis* for SDA *in situ* grouting (ISG) (Abbott and Santee 2004), a failure of a high-pressure grout system would resulting in projectiles or grout release and subsequent worker injury or even fatality was *anticipated* during operations. No radioactive or hazardous material is used in the grouting system; however, radioactive or hot grout might be carried to the surface. The impacts would be restricted to the BCBG site and site personnel.

Disturbing Pyrophoric or Unstable Materials Resulting in Exposure Injury or Fatality (Retrieve, Treat, and Dispose Alternative). Large quantities of unstable, explosive, and pyrophoric materials were buried in the BCBG (SAIC 1996a; e). Excavation and retrieval operations were likely to disturb shock-sensitive materials, cause incompatible materials to be mixed, or expose pyrophoric materials to air resulting in unacceptable exposure and injury risks (SAIC 1996a; b; c; d; e; f). Based on the detailed BCBG hazard analysis in Appendix B, the hazards associated with the excavation and retrieval tasks that may uncover pyrophoric or disturb unstable materials were classified as *probable* and *severe*.

The BCBG Walk-In Pits, containing unstable and explosive materials, have been closed by capping under RCRA (SAIC 1996a; e). However, the analysis performed in this research did not address what has been done but instead what should have been done or be done when considering the life-cycle risks for site disposition. This is not to say that the same decision (i.e., to manage these wastes in-place) would not be made when considering life-cycle disposition risks; however, approaching the remedial decisions in a consistent and transparent manner should lend independence and a degree of believability to the remedial decision and might even result in lower life-cycle risks.

Injuries and Exposure due to Excavation and Related Material-Handling Activities (Retrieve, Treat, and Dispose Alternatives). Other steps employed in the excavation and retrieval process step included at least one hazard that was considered to be *high-risk*[99] that was not related to unstable or pyrophoric materials. The consequences from these risks tended to be either traumatic injuries from excavation-related or tote-bin handling activities or exposure from containment failure or disturbance of contaminated soil. For example, the BCBG excavation step posed three *high-risk* hazards although none of these were deemed *probable* with *severe* consequences (translating into a *high* overall contribution to risk). However, the fact that there were *three high-risk* hazards in addition to those associated with managing pyrophoric and unstable materials highlighted the potential difficulties in retrieving and handling wastes from the BCBG.

Failure of Long-Term Stewardship (Manage-in-Place Alternative). Risks to the general population associated with managing buried wastes in-place depend largely on the effectiveness of long-term stewardship activities. Failure of long-term stewardship may

[99] *High-risk* hazards were defined as those from events with likelihood/consequence pairs deemed as 1) *probable* and either *critical* or *severe* or 2) *possible* and *severe* (using the definitions in Chapter III).

result in any of the following: site intrusion, inappropriate land or natural resource use, population encroachment, or continued contamination of the surrounding environment. Each of these failure mechanisms has the potential to impact a large number of people.

BCBG: Qualitative Uncertainty and Gap Analysis

The baseline risk assessments performed under CERCLA indicated that there were uncertainties and gaps in knowledge that must be addressed prior to completing a comprehensive analysis of the risks posed by dispositioning the BCBG wastes. The SDA uncertainty results in Appendix A were used to supplement available BCBG information and the resulting detailed gap and uncertainty analysis results are provided in Appendix B. An example of the uncertainty and gaps results for BCBG site characterization is provided in Table 19 based on the definitions in Chapter III. The knowledge gaps[100] or missing pieces of information that were considered to be of highest priority for resolution are provided in this section on an overall, as well as a process-specific, basis.

BCBG: Summary of the Key Uncertainties and Gaps in Knowledge Relevant to All Remedial Alternatives

Presence and Location of Unstable, Explosive, or Pyrophoric Materials. In SAIC (1996e), it was stated that:

> "In addition to the risk/hazard associated with exposure to chemicals in [Bear Creek Valley], the BCBG site presents potential physical hazards associated with the burial of pyrophoric and explosive agents. An incompatibility study… concluded that it is reasonable to expect that excavation at BCBG would foster additional reactions through re-exposure to air and unavoidable mixing as wastes are handled." (SAIC 1996e)

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[100] Key information gaps were those that are both *critical* (from a safety standpoint) and *large* (indicating little or no information is available) based on the definitions provided in Chapter III.

Alternative 1: Manage in Place
1A. No Action Option

Table 19. Gap Analysis for Manage-in-Place Alternative, No Action Option (1A)

1. BURIAL SITE CHARACTERIZATION					
Task	**What information is missing?**	**How important?**	**How large a gap?**	**Sources**	**Comment**
1.1 Determine contaminant waste forms, inventories, distributions, and fluxes from the burial site	• Presence and location of pyrophoric or explosive material • Saturated zone contaminant transport properties and model validity • Vadose zone contaminant transport properties and model validity • Geospatial distribution of contaminants and waste forms • Physical and chemical forms • Release mechanisms and rates • Infiltration rate into burial site	• Critical • Important • Important • Critical • Inconsequential[a] • Inconsequential[a] • Inconsequential[a]	• Large • Intermediate • Large • Large • Intermediate • Intermediate • Intermediate	• (SAIC 1996a; b) • (SAIC 1996a; d) • (SAIC 1996a; d) • (SAIC 1996a; d) • (SAIC 1996a; b; d) • (SAIC 1996a; b; d) • (SAIC 1996a; d)	The gaps in knowledge, particularly those relating to the presence and location of pyrophoric, unstable, shock-sensitive, or explosive material are high risk/large gap that can lead to significant risks to remedial workers.
1.2 Complete analysis of remedial activities	• Impacts of BCBG RCRA closure actions on the ability to disposition remaining BCBG wastes	• Important	• Large	• (SAIC 1996a; e)	RCRA actions have been completed on certain BCBG areas including capping. This may have an impact on the ability to disposition the wastes.
1.3 Complete conceptual model(s) for the burial site	• Contaminant transport pathways • Exposure methods • Residential and worker scenarios	• Critical • Critical • Important	• Small • Small • Intermediate	• (SAIC 1996a; e) • (SAIC 1996a; d) • (SAIC 1996a; d)	An important gap is whether the four land uses (i.e., current maintenance worker, future industrial worker, future recreational receptor, and future resident) evaluated in the RI report are the only important scenarios.

a. These gaps are considered to have a small impact on the overall task because reasonable assumptions (e.g., solubility-limited releases) can be made to provide reasonably conservative estimates for the contaminant fluxes from the burial site.

It was also stated in SAIC (1996e) that:

> "Numerous factors influence the likelihood of pyrophoric and explosive events, making a quantitative assessment of physical hazards potential impractical. These factors include chemical properties and physical form; whether the chemicals are containerized; quantities of chemicals; the age of the chemicals…; local environmental conditions…; contact among reactive chemicals; and the types of maintenance and/or remediation activities that could apply significant pressure, thermal energy, or other disruptions to the site. In general, the longer waste sits/settles, the more untrustworthy it becomes... In addition, the lack of an exact waste inventory contributes significant uncertainty to a qualitative or quantitative analysis of the physical hazard potential." (SAIC 1996e)

Thus there are uncertainties concerning the presence, type, and amount of explosive and pyrophoric materials in the BCBG. Special requirements would be necessary if handling these types of material due to their potential exposure and injury potential.

Geospatial Distribution of Wastes and Waste Forms. The inventories and geospatial distributions of contaminants were highly uncertain, and they drove the evaluation of site disposition risks. Knowledge of the locations of the risk-driving contaminants would be required to estimate the effectiveness of proposed remedial actions. For example, if volatile organic compounds (VOCs) originally placed in the burial grounds are now widely dispersed in the environment (which is the case for the BCBG), retrieval actions would be ineffective in reducing risks associated with VOC contamination. In addition, attempting to sample waste areas to characterize the geospatial distribution of risk-driving contaminants may lead to unacceptable risks to workers (SAIC 1996a; b; e).

Baseline Risk Assessment ("No Action" Alternative). As dictated in the risk analysis methodology, available baseline risk information should be evaluated before proposed remedial actions are evaluated so there is a basis for comparison. The baseline risk analyses performed for the wastes buried in the BCBG suggested that these wastes posed unacceptable risks to human health and the environment (SAIC 1996a; e; f). Sufficient analytical data were available for many contaminants to estimate risks to receptors quantitatively. However, risks from other contaminants that might be of concern (e.g., future risks, those without sufficient inventory and/or toxicological information, etc.) were either predicted or addressed qualitatively.

Because a modeling effort was required to assess the exposure risks posed in any proposed scenario, the gap analysis could be conceptualized in terms of the modeling effort required. There were methodological, release, transport, and fate aspects of modeling that has to be addressed before assessing risks. These key aspects (i.e., model uncertainty, source release, contaminant transport, receptors, etc.) were discussed when presenting the SDA gap analysis earlier in this chapter and will not be repeated.

The results of the detailed gap analysis for the BCBG are provided in Appendix B. The information gap that was both *critical* (from a safety standpoint) and *large* (i.e., little if anything is known) was

- Presence and location of pyrophoric or explosive material

This knowledge gap was relevant to all remedial alternatives considered in this research.

Other Important Knowledge Gaps. There were a number of knowledge gaps that, even though they were not classified as *critical* and *large*, were still important enough to emphasize. These include:

- It is uncertain whether *in situ* grouting (ISG) would have to be used to immobilize contaminants of potential concern in selected areas of the BCBG. There was also uncertainty as to the extent of ISG needed for both contaminant immobilization, if necessary, and subsurface (geotechnical) stabilization.

- Because the final applicable or relevant and appropriate requirements (ARARs) for the BCBG will not be defined until the Record of Decision is finalized, the regulatory requirements for BCBG remedial actions may change from those assumed in this report.

BCBG: Summary of Key Process-Specific Uncertainties and Gaps in Knowledge

Possible Future Legal Decisions and Resulting Actions (Retrieve, Treat, and Dispose Alternative). The BCBG retrieval alternative involved retrieving wastes, segregating and treating the retrieved wastes, and disposing the wastes on-site[101]. As described in Chapter III, there were two options associated with this alterative that differed in the extent to which buried wastes would be retrieved. These options may be based, to a large extent, on future legal decisions. For example, Records of Decision are still pending for the Bear Creek Valley burial grounds (including the BCBG) and groundwater. Future decisions based on these agreements may influence the remedial actions taken at the BCBG and impact the life-cycle risks to both workers and the general public.

BCBG: Risk Flow Diagrams

The risks for each remedial alternative were evaluated and the results converted into a risk flow diagram. As recommended in the risk analysis methodology in Chapter III (i.e., Phase 2C in Figure 10), risk flow diagram development was integrated with the detailed hazard and gap analyses (Appendix B) using the method illustrated in Figure 21. The risk flow diagrams for the manage-in-place and retrieve, treat, and dispose alternatives are provided in Figure 26 and Figure 27, respectively.

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[101] For the purpose of this research, the wastes retrieved from the BCBG were assumed to be disposed of in the BCBG in an area not prone to inundation.

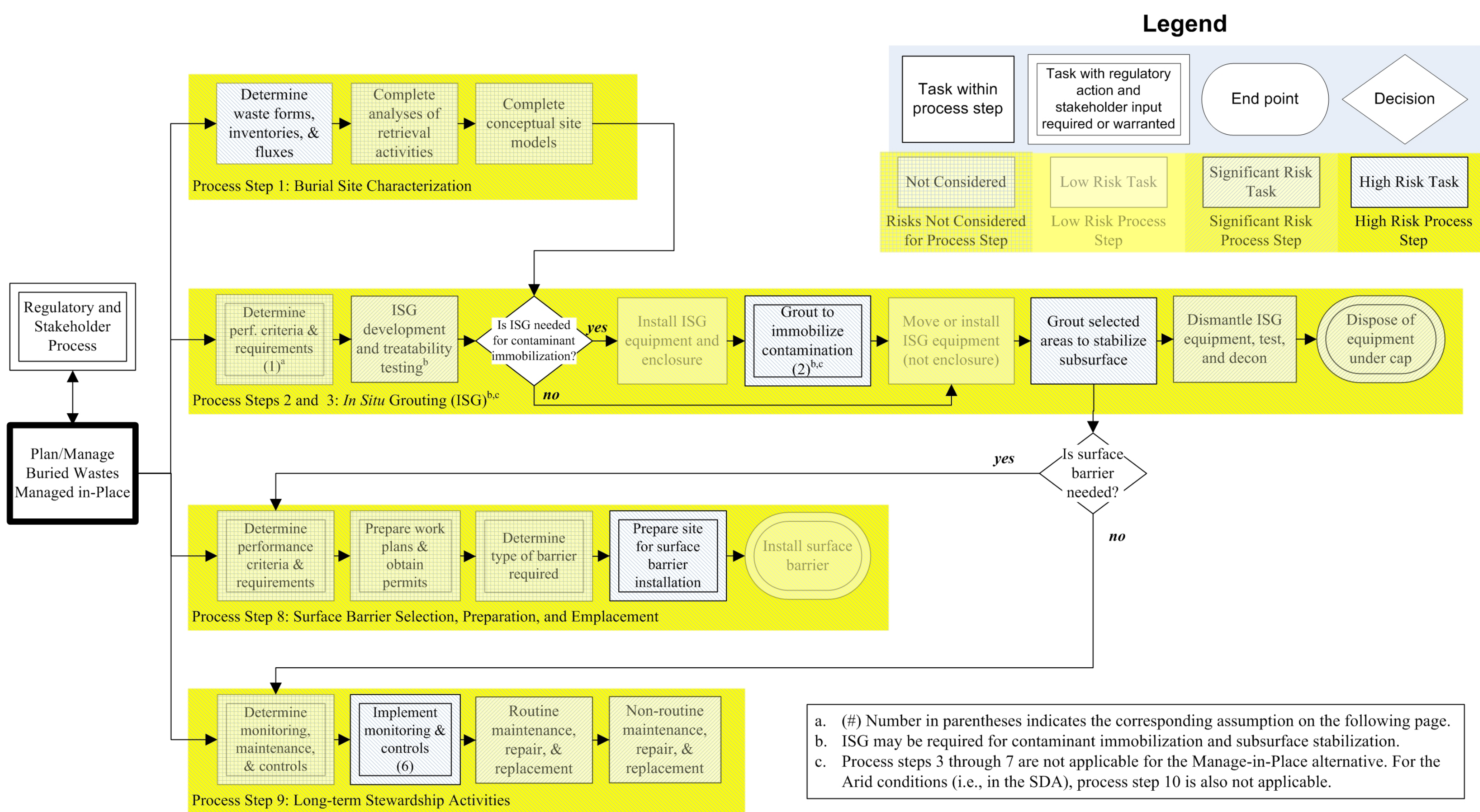

Figure 26. Risk flow diagram for the BCBG Manage-in-Place Alternatives. The degree of transparency and type of hatching represent the degree of risk for the step. Assumptions refer to those for the Modules provided in Figure 18.

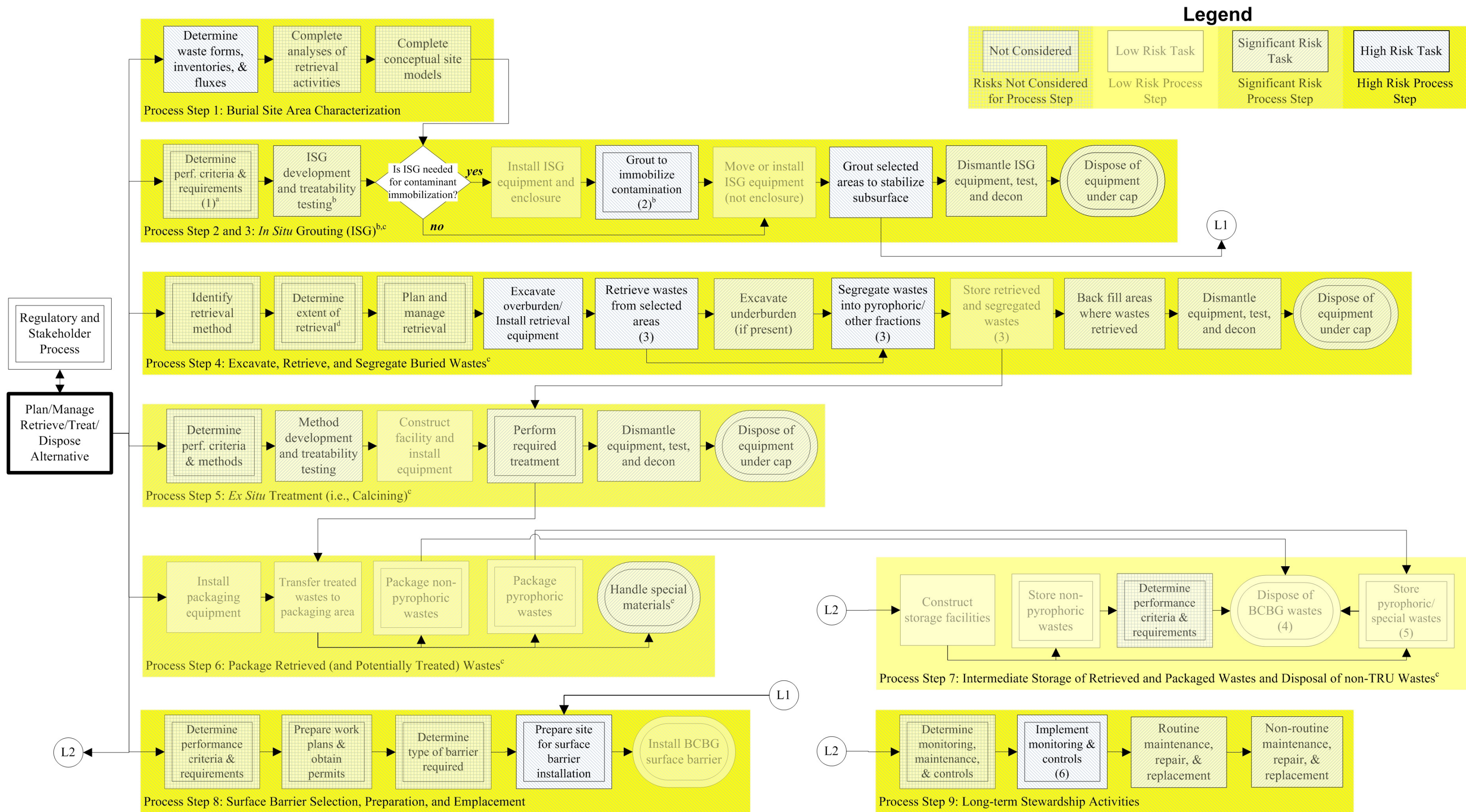

Figure 27. Risk flow diagram for the BCBG Retrieve, Treat, and Dispose (RTD) Alternatives. The degree of transparency represents the degree of risk for the process step. Assumptions refer to those for the Modules provided in Figure 18.

When compared to the risk flow diagrams for the Idaho Site SDA in Figure 22 and Figure 23, there were significant differences between the risk results for the BCBG and SDA despite obvious similarities in the process steps needed for site disposition. For the manage-in-place alternatives, the *in situ* grouting and long-term stewardship steps posed high overall risks for both sites. However, because of the inherent hazards associated with the pyrophoric and unstable materials buried in the BCBG, all BCBG process steps were considered *high-risk*. Even the comparatively simple act of characterizing the site via sampling presented unacceptably high risks to workers. For the BCBG, risks to workers were further compounded for these *high-risk* areas that would be excavated and the wastes treated for disposal back in the Oak Ridge Reservation.

BCBG: Integrated Gap and Hazard Analysis Summary

The integrated gap and hazard analysis summary table allows the results of the detailed hazard and gap analyses (Appendix B) for possible remedial alternatives to be summarized. The summary table for the BCBG remedial alternatives is provided in Table 20. Like the results for the SDA, significant hazards were expected from failure of long-term stewardship activities and the use of *in situ* grouting (ISG) for either subsurface stabilization or contaminant immobilization. However, the similarities to the SDA results tended to end here. All other BCBG process steps either had additional *high-risk* tasks (i.e., *ex situ* treatment and packaging) or were considered to have a *high* overall contribution to risk. Thus, disposition of BCBG wastes generally appeared to be a higher risk proposition (to workers) than that for SDA wastes primarily because of the presence of unstable and pyrophoric materials in the BCBG. The less these inherently hazardous materials are disturbed, the less risky site disposition would become.

Table 20. Summary of the Most Important Human Health Risks and Knowledge Gaps for the BCBG Remedial Alternatives

Process Step	1A. Baseline/No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the consequences?	Who is impacted?	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C)
1. Burial Site Characterization	√	√	√	√	√	• Disturbing unstable or uncovering pyrophoric materials during sampling	• Probable	• Severe	• Worker	• Presence and location of pyrophoric materials • Geospatial distribution of wastes and waste forms	High (1,3,2,2)
2. ISG for Subsurface Stabilization		√		√	√	• Failure of high-pressure grout system resulting in projectiles or grout release and injuries	• Probable	• Severe	• Worker	• No *high-priority* gaps	High (1,4,13,1)
3. ISG for Immobilization and Stabilization			√	√	√	• Failure of high-pressure grout system resulting in projectiles or grout release and injuries	• Probable	• Severe	• Worker	• Geospatial distribution of wastes and waste forms	High (2,4,20,1)
4. Excavate, Retrieve, and Segregate Buried Wastes				√	√	• Uncovering pyrophoric materials resulting in exposure or injury • Contaminated soil removal and rad/toxic chemical exposure • Loaded tote-bin dropped outside confinement area releasing radioactive material • Cave-in occurs during excavation operations and buries worker • Exposure or injury from handling pyrophoric materials	• Probable • Probable • Probable • Possible • Probable	• Severe • Critical • Critical • Severe • Severe	• Worker • Worker • Worker • Worker • Worker	• Geospatial distribution of wastes and waste forms • Future legal decisions and resulting actions	High (3,11,35,3)

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Chapter III.

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Chapter III.

c. The overall contribution for a process step is based on the hazard information provided in Appendix B using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. The numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

Table 20, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the consequences?	Who is impacted?	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C)
5. Ex Situ Treatment (e.g., Calcining)				√	√	• Containment/ventilation system failure and fire, explosion, injury • Fire or explosion during operations resulting in injury or exposure	• Probable • Probable	• Critical • Critical	• Worker • Worker	• No *high-priority* gaps	Significant (0,7,15,1)
6. Package Retrieved Wastes and Soil				√	√	• Containment/ventilation system failure and resulting exposure • Explosion during operations resulting in traumatic injury	• Possible • Possible	• Severe • Severe	• Worker • Worker	• Presence and location of unstable, explosive, and shock-sensitive materials	Significant (0,3,14,0)
7. Intermediate Storage and On-Site Disposal				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• No *high-priority* gaps	Low (0,0,11,1)
8. Surface Barrier Process Steps		√	√	√	√	• Uncovering pyrophoric materials resulting in exposure or injury	• Probable	• Severe	• Worker	• No *high-priority* gaps	High (1,0,7,3)
9. Long-term Stewardship (LTS)	√	√	√	√	√	• Failure of LTS (MiP Options) • Failure of LTS (RTD Options)	• Probable • Probable	• Severe • Critical	• Public • Public	• Geospatial distribution of wastes and waste forms	• High (1,7,8,1) • Significant (0,8,8,1)

a. *High-risk* hazards are 1) probable with either critical or severe consequences or 2) possible with severe consequences based on the definitions in Chapter III.

b. *High-priority* gaps are critical (in terms of safety) and large (meaning little or no information is available) as indicated in Chapter III.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. The numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

BCBG: Preliminary Comparison of Remedial Alternatives

Integrating the hazard and gap analyses allows for a qualitative ranking of proposed remedial alternatives in terms of risk, human health, environmental, and programmatic factors. When assessed for these factors in the context of the numerous assumptions and value judgments made when assessing risks and uncertainties, the remedial alternatives for BCBG buried wastes were ranked as those were for the Idaho Site SDA. Many of the assumptions and values judgments were the same for both SDA and BCBG risk analyses.

Like for the SDA, the remedial alternatives proposed for the BCBG were classified as *high-risk* because they employed one or more of the *high-risk* process steps described in the risk-flow diagrams in Figure 26 and Figure 27 and the integrated summary table in Table 20. However, this fact did not preclude a rank-ordering among the proposed remedial alternatives. For the purpose of rank-ordering based on the qualitative information generated using the risk analysis framework, it was assumed that risk increased with the increasing application of the *high-risk in situ* grouting (ISG) and waste retrieval process steps. Furthermore, the lack of containment of the wastes would have the potential to impact by far the greatest number the public and thus overwhelm any reduced worker risks[102].

Based on the above assumptions, one *possible* rank-ordering of the remedial options in terms of life-cycle risk would produce the following:

No Action > Surface Barrier >> Maximum RTD > Targeted RTD > ISG

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[102] The issues surrounding containment for BCBG wastes were different than those for the SDA because of the shallow groundwater impacting the BCBG wastes. Therefore, capping these areas would likely have little, if any, impact on contaminant migration.

which was considerably different than the corresponding rank-order for the SDA. However, like any such rank-ordering, the BCBG rank-ordering was subjective and based on assumptions concerning the comparison of the *qualitative* risk classifications defined in this report and other value judgments.

BCBG: Interpreting the Overall Risk Classification

The results of the qualitative hazard and gap analyses for the Bear Creek Burial Grounds (BCBG) identified no clear, ideal choice for dispositioning the wastes buried in the BCBG from the alternatives considered. One could qualitatively rank-order the proposed remedial options based on assumptions concerning the relative risks and remedial requirements; however, additional programmatic and regulatory information (i.e., risk, cost, regulatory requirement, etc.) would be required to identify an acceptable remedial choice.

The development of risk assessments that define risk reduction and remaining risks (in the context of the entire Bear Creek Valley) for proposed remedial actions would provide very informative inputs to the decision-making process. It would be necessary to characterize the contaminants of potential concern based on their geospatial distributions and waste forms, which would allow risk drivers and their potential impacts over time to be identified and managed appropriately. For example, capping areas prone to inundation would have little, if any, impact on contaminant transport and may even make future characterization and remedial actions more difficult and risky.

The results of the qualitative evaluation in this research indicated that the lowest risk option for the BCBG was to immobilize contaminants of concern in-place using *in situ* grouting to reduce their ability to migrate to the environment. It is important that unstable wastes not be disturbed and that pyrophoric materials are not exposed to air even during characterization activities. However, the risk evaluation for with this alternative was strongly dependent on the effectiveness of the *in situ* treatment process. This alternative also reduced the impact of any failures in the long-term stewardship.

Conclusions and Recommendations

From the risk and uncertainty information developed for the remedial alternatives for the Idaho Site Subsurface Disposal Area (SDA) and Oak Ridge Bear Creek Burial Grounds (BCBG), it appeared that a remedial alternative for each site could be selected that likely has minimum life-cycle risks. For the SDA, it appeared that placing a surface barrier on the site would provide the minimum life-cycle risk remedial option for the arid SDA buried waste site. However, because factors other than risk must be considered to make a risk-informed decision, retrieval actions targeted on the highly mobile wastes that represented the highest, short-term risks[103] might be the risk-informed decision that would be selected by regulators.

For the BCBG areas, especially those containing unstable and pyrophoric materials, retrieval actions may be prohibitively hazardous for remedial workers. Thus, even though retrieval actions may be warranted for other BCBG wastes, some assurance is required that unstable wastes will not be disturbed and pyrophoric materials will not be exposed to oxygen. However, even the comparatively simple act of characterizing the wastes may be highly hazardous if the presence and location of the highly hazardous materials in the BCBG are not well-known. If the uncertainties in the presence and

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[103] One issue that must be considered is whether, by the time remedial actions can be taken to influence highly mobile contaminants, it is too late to make a significant difference in the risks posed by site wastes.

location for the unstable and pyrophoric materials cannot be reduced to an acceptable level, then the management of the wastes in-place would likely be the minimum life-cycle alternative.

The most important gaps in information that must be resolved to improve risk communication and enable risk-informed decisions were identified as a result of this research. The most important gaps concerned i) risk reductions potentially achieved through implementation of each remedial option and remaining risks after remedial actions in the context of closure activities, ii) geospatial distributions of the risk-driving contaminants of concern (including unstable and pyrophoric materials in the BCBG), and iii) future court decisions or agreements that may mandate the extent to which buried wastes must be retrieved from the site for treatment and disposal. These findings appeared to be consistent with recent National Academy findings and recommendations that suggest that the effort, exposure, and costs associated with retrieval, immobilization, and disposition of transuranic wastes buried without intent for retrieval may not warrant the corresponding risk reduction that would be achieved (NAS 2005). The risk assessment framework defined in this research provides a foundation for answering these types of questions concerning potential remedial alternatives.

When relevant risk factors were taken into account, the lowest life-cycle risk option for the SDA was likely to be extraction of the highly mobile contaminants that posed significant risks followed by in-place containment. For the BCBG wastes, the preferred option was the immobilization of wastes using *in situ* treatment. If the presence and location of unstable and pyrophoric materials in the BCBG can subsequently be well-characterized, then targeted retrieval actions may provide the lowest-risk remedial option for the BCBG. Efforts are underway at the Idaho Site to better characterize the spatial distribution of key contaminants. The greatest risk reductions may be achieved through remedial actions targeted towards retrieval of contaminants presenting the greatest threats to receptors rather than through removal actions based on waste origin or other non-risk factor unless integrated into the risk-informed decision-making process.

The aforementioned National Academies report stresses the importance of balancing public health with worker and environmental risks, costs, achievability, and other site-specific factors when developing a risk-based approach (NAS 2005). The aforementioned observations were made in the context of possible exemption of certain high-level and transuranic wastes from disposal in a geologic repository; however, the observations have broader implications and applicability. Furthermore, these observations are consistent with the risk analysis framework approach developed in this research.

The preliminary risk and gap characterizations for the SDA were developed using the extensive document database provided in the Administrative Record for the Idaho Site CERCLA process[104]. Related materials pertaining to other DOE sites were also utilized when available. Idaho Site personnel provided additional insights and answered extensive questions. The primary source for the BCBG evaluation was the Oak Ridge CERCLA remedial investigation report (SAIC 1996a; b; c; d; e; f).

Further detailed analysis and evolving information may identify additional considerations that impact risk characterization for the remedial alternatives considered in this report. The risk characterization process should be viewed as a vehicle for gathering, organizing, and evaluating information to inform the management and decision

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[104] The Idaho Site Administrative Record can be accessed via http://ar.inel.gov (accessed March 13, 2008).

processes. The results of the qualitative analyses performed for the SDA and BCBG suggest that quantitative risk analysis of the two buried waste sites (i.e., the next phase in the risk analysis framework in Chapter III) may provide both a better understanding of potential life-cycle risks and thus a clearer basis on which to make remedial decisions.

This research was developed to promote a broader discussion among DOE, regulators, public representatives, and the general public on the most appropriate path forward for disposition of DOE buried wastes. Risk is but one of several important aspects that must be considered in decisions impacting public welfare. Imperfect and incomplete information, inherent variability and uncertainty, and differences in individual values and perspectives will undoubtedly lead to differing views on the appropriate path forward. These differences highlight the necessity for a clearly defined and engaged stakeholder participation process as an integral part of the on-going decision and management process for wastes buried in the DOE Complex.

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CHAPTER V

A CONCEPTUAL BURIAL MODEL FOR DESCRIBING DEPARTMENT OF ENERGY (DOE) BURIED WASTE SITES

Despite the extensive qualitative risk and uncertainty analyses performed in Chapter IV, remedial actions could not be definitively identified for either Department of Energy (DOE) buried waste site studied. A *quantitative* analysis of the risks and uncertainties for buried waste site disposition is warranted as indicated in the risk analysis framework defined in Chapter III. To help manage the complexity of estimating risks for remedial decision-making, a conceptual burial model is defined in this chapter to embody, *for a screening-type analysis*, the contaminants, waste types, releases, fate and transport, exposures, and receptors representative of DOE buried waste sites.

The impacts of uncertainties and data gaps must be addressed when estimating exposures and risks from even the simplest buried waste site. Uncertainties likely include the types and concentrations of contaminants, locations of wastes and contaminant plumes, subsurface properties affecting contaminant fate and transport, meteorological conditions, etc. Even after extensive study, many uncertainties will remain large. Some information will remain unavailable despite the extent and expense of characterization. Even if the properties needed to describe current waste and site conditions could be accurately characterized, many of these critical properties will change over time, and the future values of these critical parameters cannot be predicted accurately.

To help manage the complexity of estimating risks for buried waste site disposition and decision-making under such large attendant uncertainties, a conceptual burial model is defined so that the aforementioned uncertainties are not problematic (for the *conceptual* buried waste site). The conceptual model will adequately represent, *for a screening-type risk analysis*, the contaminants, waste types, releases, fate and transport, exposures, and receptors representative of buried waste sites across the DOE. The decision then becomes how well the conceptual burial model developed in this research describes the DOE buried waste site in question and how meaningful will be the results.

Even more importantly, the usefulness of the conceptual burial model concept is much like that of the risk assessment itself, i.e., as an organizational tool to help focus attention on the factors critical to the decision-making process. These factors include critical assumptions made during modeling and the significant contributors to risk and impacts of uncertainty. DOE buried waste sites are sufficiently complex so that any tool shown to provide useful information for these sites should find broader applicability.

Conceptualizing the Buried Waste Site

A general conceptual burial drawing is provided in Figure 28. Exposure risks to the general public and workers from hazardous chemicals and radionuclides in the buried wastes and accident risks to workers are both represented. This drawing (as is any such drawing or model) is necessarily a highly simplified and idealized representation of the actual buried waste site and its disposition over time. For example, exposure risks to chemicals and radionuclides would be manifested over many years (perhaps millennia) to current and future receptors; however, exposure risks are presented in a way that might indicate a false simultaneity (i.e., misrepresent the timing) to one unfamiliar with the true nature of the risks.

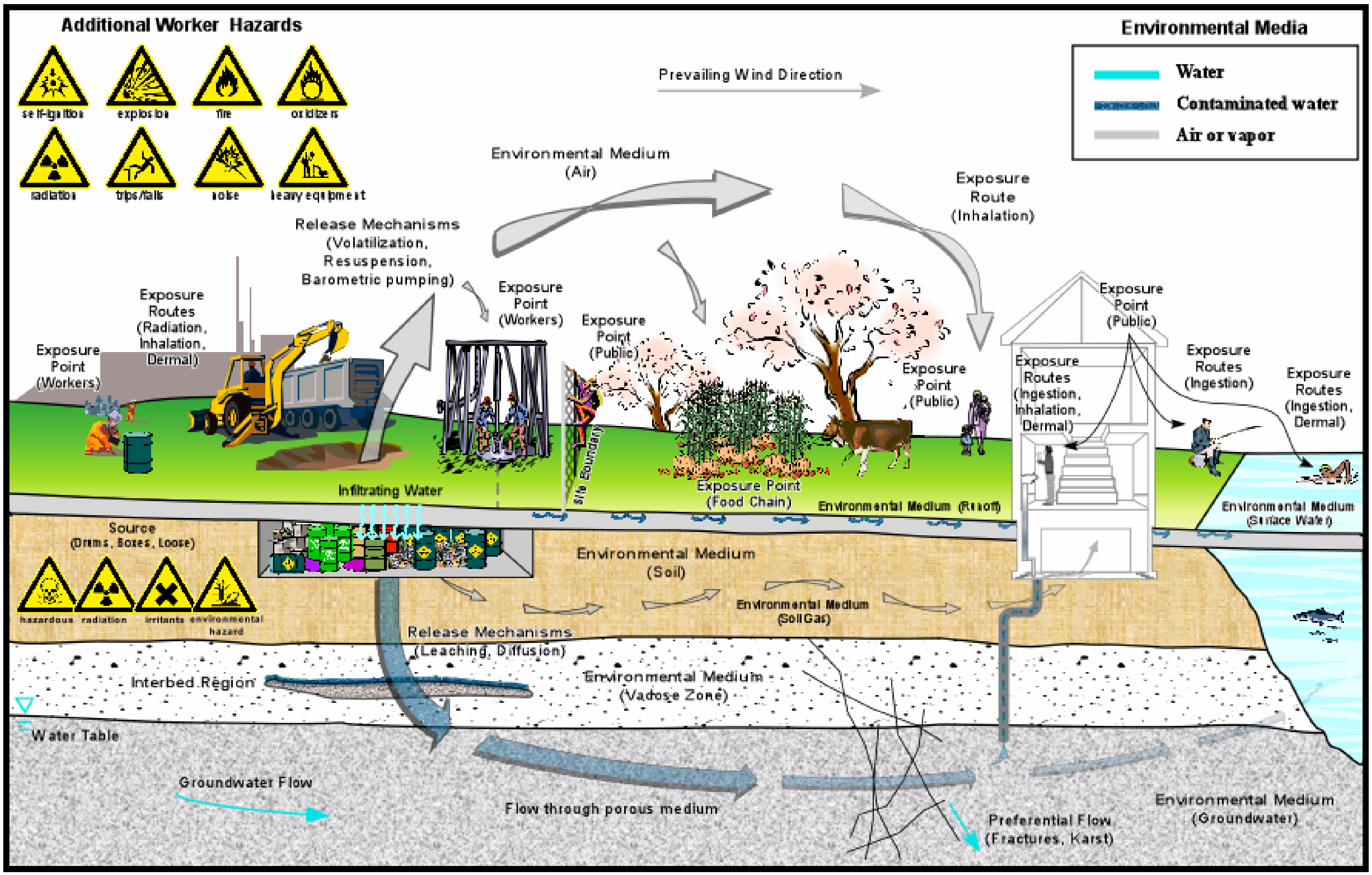

Figure 28. Conceptual burial drawing representing risks from chemical and radionuclide exposures and short-term remedial activities. The degree of change in shading indicates the time to impact. Adapted from Figure 6-2 (ATSDR 2005).

The exposure risks represented in Figure 28 include inhalation of contaminants via the air pathway; radiation and dermal contact to intruders; ingestion, inhalation, and dermal contact via the soil gas and groundwater pathways; ingestion of contaminated fish; or ingestion or dermal contact of contaminated surface water during recreational use. *Exposure* risks may be significantly higher to workers because of their proximity to the wastes, especially upon excavation, retrieval, and handling operations.

One simplification comes about in how the temporal dimension of risk comes into play; the various risks may be present over millennia, which is difficult to represent in a conceptual drawing like the one provided in Figure 28. In Figure 28 the temporal dimension in exposure (and thus risk) is represented by the change in shading of the pathway arrow—the relative shading represents the time to effect. Other simplifications include how to best represent temporal and spatial variations in subsurface conditions affecting the fate and transport of contaminants over long distances and geologic time.

The conceptual burial drawing should be seen as one of the central organizing principals of the risk assessment process. Errors associated with drawings and other conceptualizations (e.g., conceptual site models) are notoriously difficult to capture (NAS 2005). The impact of these errors must be considered because uncertainties in conceptual models can dominate the overall uncertainties in risk estimates (Magnuson 2004; Meyer and Gee 1999; NAS 2005). Thus indications of uncertainties and information gaps for the drawings, conceptual site models, and other conceptualizations must be considered on the results of the assessment process. Most attempts at capturing conceptual model uncertainty have been focused on examining the sensitivity of risk estimates to changes resulting from different possible conceptualizations.

The exposure risks to hazardous chemicals and radionuclides that are often the focus of the risk assessment process are not the only ones faced by receptors. However, the dominant sources of actual risks for workers, especially during remedial activities like excavation and waste retrieval, are from more mundane accidents including slips, trips, or falls (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). Some studies have indicated that the remedial activities themselves (e.g., excavation increasing hazardous volatile contaminant emissions) may increase public health risks (Brett et al. 1989).

Occupational risks, indicated by the warning symbols in the upper left-hand corner of Figure 28, are often neglected in selecting remedial actions for a buried waste site (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). Furthermore, significant risks (to both workers and the general public) from transporting retrieved wastes to more stable locations and those associated with residual contaminants and the final internment of wastes (and their future impacts) are often omitted from the decision-making process.

Both exposure and standard industrial risks are represented in the conceptual drawing in Figure 28. However, even though the drawing represents both types of risks, it may omit some risks (e.g., for specific remedial actions) posed by buried waste site disposition. The purpose of the drawing is to present representative risks and other important characteristics of the disposal process that might impact risks. Furthermore, the temporal and spatial variations in the risks posed by disposition of buried waste sites are difficult to capture using any drawing or conceptual model. Even though the drawing in Figure 28 conveys useful information, it does not represent the true nature of all risks for buried wastes. An additional diagram, the conceptual site model (CSM), is needed to help complement the information provided by the drawing in Figure 28.

General Conceptual Site Models for DOE Buried Wastes

For hazards related to the exposure of receptors to hazardous and radioactive contaminants, another way to think about *risk* is captured by the conceptual site model (CSM) (ASTM 1995; USDOE 2003). A CSM is "a written or pictorial representation of an environmental system and the biological, physical, and chemical processes that determine the transport of contaminants from sources through environmental media to environmental receptors within the system" (ASTM 1995). The CSM links sources of contaminants to receptors via release to environmental pathways that contaminants must migrate and exposure routes to which potential receptors may be exposed.

That is, unless there is i) a source of contaminants with potential health effects, ii) a mechanism of release of contaminants to the environment through which the contaminants may migrate, and iii) an exposure route through which potential receptors can be exposed to the contaminants, then there is *no risk from exposure* to the contamination. This conceptualization also explicitly brings potential receptors into the picture and provides a useful framework for examining exposure risks associated with buried waste disposition. Figure 8 in Chapter III shows a simplified CSM corresponding to the *human health exposure risks* for the conceptual drawing in Figure 28. The degree of shading of the pathways in the generic baseline CSM provides an indication of the temporal nature of the potential exposure risks.

The CSM in Figure 8 describes the relationship between buried wastes and exposure to potential receptors for baseline conditions, which are those manifested for the site if no remedial actions or other risk-mitigating steps would be taken. The exposure risks posed by site contaminants are problematic if they have already migrated into the environmental and if no remedial action was taken to protect potential receptors (e.g., workers, general public, biota, etc.). A lack of early remedial actions is not the case for many DOE buried wastes sites. Often early actions (e.g., vapor extraction, surface water controls, grouting, etc.) were taken to mitigate the highest risk drivers for contaminants in the site based on screening risk assessments using assumptions meant to maximize predicted exposure risks. Any such early actions should be represented on the CSM representing baseline conditions for the site.

The risks to workers and the general public from *exposure* to buried wastes and contaminated soils may often be significantly smaller than the standard industrial risks from accidents (e.g., slip, trips, falls, etc.) likely during remedial activities. Thus the baseline CSM in Figure 8 provides only part of the necessary risk picture for buried waste disposition. With some modification, the CSM can be used to represent both exposure and accident risks associated with remedial actions required for buried waste site disposition. An example of the new CSM was provided in Figure 11 in Chapter III.

One important difference between the baseline CSM in Figure 8 and that in Figure 11 (based on the CSM for exposure risk) is the inclusion of the non-exposure hazards (e.g., injury, explosion, fire, criticality, etc.) for remedial actions needed to disposition the site. Another difference is the explicit inclusion of exposure hazards (i.e., radiation, hazardous chemicals, irritants, and environmental hazards) associated with the buried wastes themselves as indicated in Figure 28. The hazards listed here should not be considered exhaustive and may change depending on the buried waste site and remedial action (e.g., characterization, excavation, *in situ* treatment, etc.) being considered. A unique CSM may be required for each remedial action step (Brown et al. 2005).

After required remedial actions have been completed (posing the industrial and exposure hazards illustrated in Figure 11 from Chapter III), the post-closure site will continue to present some degree of risk to both workers and the general public. For example, if buried wastes are either managed in-place or retrieved and treated for disposal elsewhere, at least some contamination will persist in or near the original buried waste site that presents a degree of risk. Post-closure risks should be tolerable from a regulatory perspective. It is how and what remedial actions are selected and to the extent the actions are executed and effective that will determine how much risk is posed by the site post-closure. Assuming, at a minimum, that a surface barrier will be installed to control infiltrating water and thus contaminant migration, the post-closure risks are illustrated by the generic conceptual site model provided in Figure 12 from Chapter III.

Critical Components of the General Conceptual Site Models for Buried Wastes

Risk assessments can be broadly categorized as either *predictive* (i.e., attempting to estimate future risks associated with remedial action or inaction) or *retrospective* (i.e., estimating effects from measurements of existing contaminant levels in specific media or receptors). The risk assessments in this research are inherently predictive because the future exposure and industrial risks to receptors for proposed remedial alternatives are evaluated. The very nature of predictive risk assessments dictates the use of mathematical abstractions of the pertinent physical processes (e.g., contaminant release, fate and transport, exposure, etc.) using models, many of them complex and requiring large numbers of input data and parameters. However, despite advances in fate and transport, exposure and effects, and other necessary models, even the most accurate models include considerable uncertainty as do the risk estimates obtained from their use.

Because models must be employed in predictive assessments and no mathematical model is perfectly accurate, model uncertainty will persist as will uncertainties in conceptual models and the data required as input. Thus uncertainty is an essential additional dimension to any risk estimate.

It should not be the business of risk assessment to attempt to nullify uncertainties and provide completely accurate risk predictions—which would be an expensive and doomed enterprise. Neither should the business of risk assessment be to diminish the role of uncertainty in assessing risks, instead the challenge is to reflect fully and appropriately the uncertainties in the risk information provided to decision makers (NAS 2005).

Consideration of uncertainties is an integral part of developing conceptual models, selecting mathematical models, and selecting input data and parameters used to estimate the risks associated with the disposition of buried wastes. Furthermore, the level of sophistication in mathematical models and uncertainty management strategy should be commensurate with the importance of the decision and complexity of the problem and analysis. For example, in a tiered risk analysis of the type described in Chapter III, the requirements of models, uncertainty analysis, and information requirements are likely to change significantly from the screening phase to the more detailed analysis phases that often require advanced mathematical models and considerable site-specific information.

The important components of the conceptual drawing in Figure 28 and further refined using conceptual site models can be condensed into a set of descriptors (i.e., measures and parameters) that should be addressed before estimating risks for buried waste site disposition. A representative set of these important components is provided in Table 21. More importantly, because exposure risks are directly related to concentrations in and exposure to contaminated environmental media, there are a few fundamental measures—the temporal contaminant fluxes bolded in Table 21—that, if known, could be used to predict exposure risks. Although it may requires hundreds, if not thousands, of inputs and parameters to model the fate and transport of contaminants from a burial site to receptors, the modeling effort is, in essence, estimating inter-media contaminant fluxes and exposure media concentrations (which vary both temporally and spatially).

The important components of conceptual models that should be considered when estimating the risks for buried waste site disposition can be condensed into a handful of fluxes that vary temporally and spatially. These fluxes—or "pinch-points"—characterize the transport phenomena and contaminant concentrations needed to estimate the risks associated with potential exposure of receptors to the contaminants buried in the waste site. Another way to look at this abstraction is to integrate the concept of pinch-points and the conceptual site models.

The integration of conceptual models and pinch-points is represented in Figure 29 where pinch-points (i.e., fluxes) are shown on red backgrounds. The confluence of releases for the various waste types to the air and soil and vadose zone are presented. However, for understandability only fluxes from the soil and vadose zone to groundwater and the corresponding exposure are shown on Figure 29. Analogous representations for the other combinations of fluxes (e.g., to air, soil, etc.) and receptors (e.g., workers, residents, etc.) that contribute significantly to the overall risk presented by the contaminants in the site should be represented on the diagram.

Table 21. Critical Components of the Conceptual Site Model for *Exposure Risks* from Buried Wastes (After Table 1 from Travis et al. (2004))

Category	Measures and Parameters Needed
Release and Transport Drivers	Meteorological conditions (including prevailing winds) and precipitation (e.g., rainfall, snow, etc.)
	Relative fractions of precipitation to burial site, evapotranspiration, surface water, vadose zone, and groundwater
	Dust resuspension rate and mass loading of soil in water
	Net fluxes of water infiltrating into burial site (percolation) and to surface water (runoff)
	Net fluxes of soil to the atmosphere (resuspension) and to surface water (runoff)
	Net flux of vapor from subsurface to atmosphere due to barometric pumping
Source Term	Waste types (e.g., loose, metal, boxes, drums, glass, etc.) and spatial distributions of types throughout burial site
	Spatial distributions of contaminants within types, buried waste site, and outside areas if migrated from site
	Distributions of contaminants available to contribute to source term (i.e., function of waste type, immobilization, and degradation)
	Barriers restricting or preventing infiltrating water or other driver for contaminant release
	Barriers restricting or preventing contaminant transport from buried waste site
	Rate of barrier degradation or outright barrier failure mechanisms
	Release rates of contaminants from wastes into burial site
	Contaminant fluxes through burial site boundaries
Burial Site	Topography including both site and potentially contaminated areas
	Dimensions of buried waste site and potentially contaminated areas
	Dimensions and depth to subsurface layers (e.g., vadose zone, water table, etc.)
Transport Properties	Physical and chemical characteristics (e.g., porosity, organic content, density, pH, hydraulic conductivity, etc.) for subsurface layers through which contaminants migrate
	Effect of seasonal meteorological conditions on transport properties
	Contribution of preferential flows (from fractured or karst conditions) to flow in each layer (in vadose zone and groundwater)
	Flux of water through each subsurface layer
	Direction and velocity distribution of groundwater flow
	Contaminant fluxes through each layer in the subsurface and resulting concentrations in media to which receptors may be exposed
Receptor Properties	Locations and densities of potential receptors
	Land use
	Exposure routes (e.g., inhalation, ingestion, etc.) to potential receptors
	Receptor "flux" through contaminated media
Exposure/Dose Factors	**Exposure rate and duration**
	Exposure to dose factors
	Bioaccumulation factors
	Exposures to contaminants and resulting computed doses
Risk Factors	Toxicity or slope factors for carcinogens or “no effect” levels for non-carcinogens
	Cumulative effects from multiple simultaneous exposures
	Computed risks resulting from exposures or doses

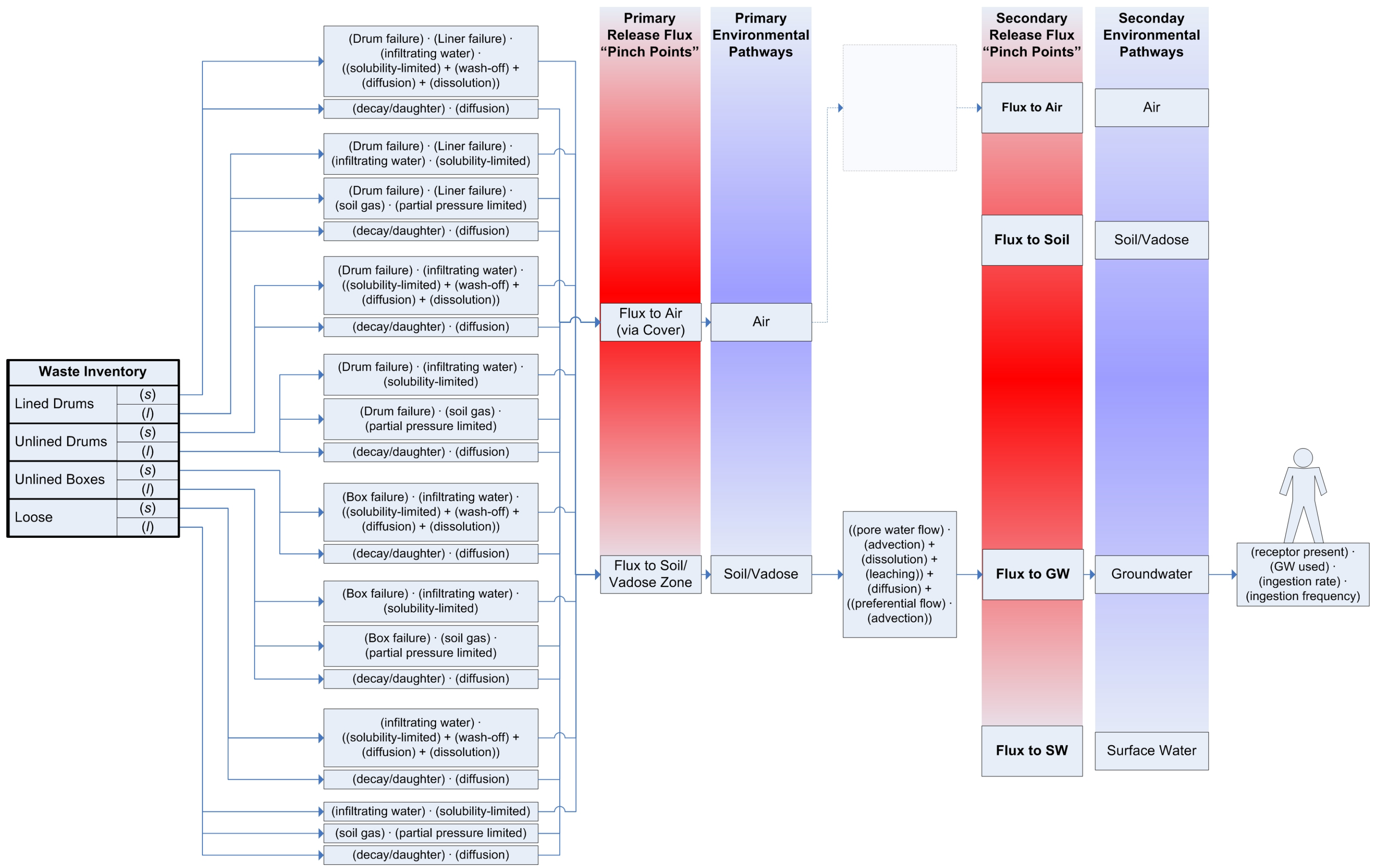

Figure 29. Abstracting the Important Components of the Drawing in Figure 28 using the Fluxes (or Pinch-Points) between the Various Exposure Media represented in the Drawing.

Simplifications Made for Screening Exposure Risk Modeling

Even when introducing the concept of pinch-points, the conceptual models needed to describe a buried waste site often become very complicated very quickly. For example, the contaminant fluxes moving through exposure media and the resulting concentrations vary both spatially and temporally. Potential human receptors tend to move into or out of contact with the contaminated media. Exposures to receptors (and corresponding risks) can vary considerably over time. Even the most sophisticated models developed to predict the risks associated with a buried waste site must include a number of simplifications. It is not the need for simplification that should be the issue; it is recognizing and considering the impacts of the simplifications made on the resulting risk predictions that should be the focus of the analysis.

The key components of the simplified conceptual model used for the *screening risk assessment* of buried waste sites are described in Table 22. The general basis for modeling are the guidelines to estimate radiation doses for exposures to residual contamination after decommissioning (Kennedy and Strenge 1992). Additional transport mechanisms are added where potentially important. More current information has been introduced including more recent screening models for radionuclides releases to the environment (NCRP 1996a; b). Volatile organic compounds and other important non-radioactive constituents are modeled[105]. More detailed information on key components and constituents are provided in Chapter VI describing the screening risk model.

[105] As a starting point, information for contaminants known to have been buried in the prototype sites will constitute the list of constituents included in modeling. For radioactive constituents, progeny will be included (Eckerman 2003a; ICRP 1983). Because there is potential for organic compounds to degrade to hazardous compounds (Lawrence 2006), potential degradation products will be included.

Table 22. Simplifications Used for the Critical Components of the Conceptual Model for Screening Risk Assessments

Key Component	Model Basis and Key Assumptions[a]	Likely Impact on Risk Estimate
Transport and exposure media	Contaminant transport through the various exposure media is assumed to be one-dimensional in nature (Tauxe 2004). A layer of air of fixed height is moving above the site at the average wind speed.	Contaminant transport is three-dimensional in nature and the boundary layer would not be fixed over time. One-dimensional transport should tend to overestimate transport and thus risk. For atmospheric transport, the use of reasonable bounding values including a shallow boundary layer depth should (but is not guaranteed to) overestimate exposure risk.
Net fluxes of water infiltrating into site (percolation) and to surface water (runoff)	Long-term (e.g., annual) average used for precipitation (and divided among evapotranspiration, runoff, and percolation) (Dwyer 2003)[b]. The average values used are assumed representative over the long simulation times considered.	Long-term estimates of risk are evaluated for screening purposes and long-term averages are used to describe water flux. Bounding values for these fluxes are not guaranteed to over-estimate long-term exposure risk. Episodic infiltration coupled with non-linearity in the Richardson equation results in under-estimation of net water velocity and transport.
Net flux of soil to the atmosphere via resuspension	Maximum particulate loading in air is used assuming all particulates are contaminated soil and PM_{10} (Tauxe 2004)[c]. It is assumed that the average particulate loading is constant over time and no deposition occurs.	Because the exposure risk is based on the maximum particulate loading in air and all particulates are assumed to be contaminated soil and none are deposited on the surface, the long-term risk via the atmosphere should be overestimated.
Net flux of vapor from subsurface to atmosphere via barometric pumping	The flux is based on overall transport efficiency, amplitude of the change in barometric pressure versus subsurface pressure, and number of cycles per year providing an annual fraction of contaminants to transport (Nilson 1991). The fraction is assumed to be independent of depth.	The site-specific information (e.g., fracture spacing, diffusivities, etc.) will be difficult to obtain although a range of potential transport efficiencies can be defined from the information given in (Nilson 1991). However, whether or not the risk would necessarily be overestimated cannot be determined *a priori*.
Dispersion of contaminants in air	The simple approach of using atmospheric dispersion factors (often denoted χ/Q or E/Q (USDOE 1997)) will be used to estimate the concentration in air a given distance from the burial site.	There can be large uncertainties in the atmospheric dispersion factors computed for a specific DOE site under a specific set of conditions. Again there is no a priori way of assuring that the risk would be overestimated.

a. Most of the key assumptions are taken from two basic references (Kennedy and Strenge 1992; NCRP 1996a). Assumptions not taken from these references are specifically cited in the table.

b. A water balance for the system is (Dwyer 2003): Precipitation (Pr) = surface Runoff (R) + Infiltration (I) where I = Evapotranspiration (ET) + lateral Drainage (D) + change in Storage (dS) + Percolation (Pe). For screening purposes, it is assumed that there is no change in water storage for the soil cover and that there is no lateral drainage (when the wastes are only covered with soil or that D is a part of the runoff for covered sites). Only surface runoff (R), percolation (Pe), and evapotranspiration (ET) remain; long-term averages of the percolation and runoff values are used in modeling.

c. PM_{10} is the fraction of particles with a diameter of less than or equal to 10 micrometers and has been the standard against which EPA has been measuring Clean Air Act compliance for particulates. An alternative model (a.k.a., "maximum resuspension") is based on the assumption that the contaminant loading is not a function of the soil concentration (Tauxe 2004).

Table 22, Continued

Key Component	Model Basis and Key Assumptions[a]	Likely Impact on Risk Estimate
Net flux of soil to surface water via runoff (erosion)	Water from runoff transports soil to near surface water (Dwyer 2003). Although runoff (assumed not a function of vegetative cover) is not pore water, runoff will "dilute" the water in (some fraction of) the soil pore spaces and transport soil to surface water. Transported soil will be replaced with "clean" soil resulting in no net change in soil depth.	Runoff can vary temporally based on many factors including vegetative cover, slope, etc. Runoff for simulation purposes will be a long-term average from historic information and, because it is a long-term average, will vary much less over time. However, because of the lack of site-specific information and the potential impacts of other factors, there is no guarantee that the risk will be overestimated.
Plant transport	An effective concentration ratio approach will be used to estimate the activity on plants (e.g., grasses and shrubs) growing above the burial site from root uptake and resuspension (Kennedy and Strenge 1992). Only elements (including radionuclides) are transported in this manner. Root distribution profile and rooting depth are time invariant (Jackson et al. 1996).	The degree of conservatism in plant transport is controlled by how concentration ratios are defined. Tauxe (2004) posed his deterministic calculations at the 50^{th}-percentile. The 95^{th}-percentile values will be used for deterministic estimates to overestimate exposure risk. Any organic constituents and their contribution to risk are ignored in this method. Time invariant root models may inadequately describe establishment of vegetation in cleared areas over long time-scales (Arora and Boer 2003).
Animal transport	The impact of burrowing creatures (assuming only ants and mammals) is effectively to transfer soil from one layer to another (either upward from burrowing or downward in collapse) (Tauxe 2004). All contaminants in all phases are transferred. Burrow depth and distribution are time-invariant.	There may be burrowing creatures omitted from those considered that may excavate more earth or burrow deeper (than those considered). The time-invariant burrow models may also poorly represent actual site conditions. Thus the method cannot be guaranteed to overestimate the exposure risks involved.
General inventory	All waste burials will be assumed to be simultaneous at the beginning of the simulation (with no decay-correction). The inventory will be divided into Waste Areas, each having distinct characteristics and comprised of two layers: the upper accessible to biota and the lower not. There will be at least two Waste Areas, one that would be a likely candidate for retrieval and the other not.	The decision to not decay correct the inventories will have the greatest impact on risks from short-lived radionuclides. Also the timing of when receptors will be exposed will be impacted by this decision. However, preliminary stochastic studies indicate that variation in the time to impact will vary greatly although the impact itself varies much less. Unfortunately there is guarantee of conservatism in the total exposure or risk estimates using this method.
Waste Types and Contaminant Concentrations	The contaminants in each Waste Area will be distributed between boxed, drummed, and loose wastes as they were originally buried. It is assumed that background contaminant concentrations are zero elsewhere. Bounding and stochastic inventories will be used.	These types cover the vast majority of expected waste forms. Other waste forms (e.g., glass, grouted, etc.) can be added if reasonable cases can be made. Bounding or stochastic inventories are needed to assure the exposure risks are likely overestimated and the comparison between exposure and industrial risks is as clear as possible.

a. Most of the key assumptions are taken from two basic references (Kennedy and Strenge 1992; NCRP 1996a). Assumptions not taken from these references are specifically cited in the table.

Table 22, Continued

Key Component	Model Basis and Key Assumptions[a]	Likely Impact on Risk Estimate
Source term: Contaminant Releases/Fluxes from Burial Site	Release of contaminants into the environment is a function of waste type (failure and release), Waste Area properties (partitioning, solubility, etc.), water flux through area, and vapor diffusion to soil layers.	The contaminant flux from the burial site is controlled by a large number of factors (e.g., inventory, waste type, partitioning, etc.) that can compete or reinforce each other over time. Thus there is way to guarantee that temporal exposure estimates are overestimated.
Decay Processes	Many contaminants of potential concern are radioactive; therefore, radioactive decay will be included (ICRP 1983). Organic compounds may degrade in the environment to more dangerous compounds; this degradation will be included (Lawrence 2006).	The radioactive decay for the contaminants that be of concern in this research are essentially error-free compared to the uncertainties in the other parameters needed to model risks associated the burial site. Because organic compounds degrade differently depending upon the mechanism and medium, there may be significant uncertainty introduced in the concentrations and thus risk by handling organic degradation as decay.
General Transport Processes	The transport processes (i.e., advection and diffusion) via which contaminants are distributed within the environment will incorporate solubility constraints and partitioning among exposure media represented in the model. For screening purposes, partitioning will be managed using linear partition coefficients (USEPA 1999).	The solubility and partition coefficients used in the model will be assumed fixed and thus not functions of the conditions within the exposure media. For example, one reason there is such large variation in the soil/water partition coefficients often used in transport models is that the linear partition model does not adequately describe the actual mechanisms involved. Large parameters variations are used to compensate for model inadequacies and no statement concerning the "conservatism" in the resulting risk can be made.
Vadose Zone Transport—Cap, Soil and Waste	These layers will be represented using well-mixed soil media that area assumed isotropic (Tauxe 2004). Transport occurs via advection and colloids (water), diffusion (air), barometric pumping (air), inundation and flooding (water), and plants and animals (biota).	The heterogeneity in properties and contaminant concentrations are not considered because the model is essentially a one-dimension transport model. Furthermore, transport is represented using long-term average values which may underestimate some short-term exposures and risks.
Vadose Zone Transport—Subsurface Layers	These vadose zone layers will be represented as fractured media with a coating of the relevant material. It is assumed that three layers (i.e., upper, interbed with infill, and lower) will adequately describe the relevant sites. Transport occurs via advection and colloids (water).	The selected prototypic sites have either fractured basalt (SDA) or karst (BCBG) vadose zones underlying the waste sites. The fracture or equivalent porous media descriptions of the vadose zones are used to develop fractured zones. The fracture zone descriptions are not unique, which may impact travel times. Vapor transport from the fractured zone (either barometric pumping or diffusion downward) is ignored. There is no guarantee that the exposures and risks are overestimated.

a. Most of the key assumptions are taken from two basic references (Kennedy and Strenge 1992; NCRP 1996a). Assumptions not taken from these references are specifically cited in the table.

Table 22, Continued

Key Component	Model Basis and Key Assumptions[a]	Likely Impact on Risk Estimate
Groundwater Transport	The saturated zone will be represented by two "pipes" (or fluid conduits): a shorter one representing flow that may impact a future on-site resident (via well water) and a longer one feeding impacted surface water (when appropriate). Transport occurs via advection and colloids (water) and the initial zone may receive inundation transport from the Waste Areas.	The use of such a conduit is based on the assumption that the transport in the saturated zone is advection-dominated and one-dimensional. Other process (e.g., retardation, longitudinal dispersion, in-growth, etc.) are included. There will be some impact on exposures and risks because the saturated zone is likely fractured also. Spatial variation in the plume concentration associated with this element can be addressed. Because of the confluence of effects, there is no guarantee that exposures and risks are overestimated.
Surface water transport	A series of well-mixed vessels will be used to model surface water flow (and produce numerical dispersion). Stream depth and width will be estimated from relationships in (NCRP 1996a). Transport occurs via advection (water) and colloids and suspended solids. Solids will remain suspended and no volatilization occurs. Surface water may receive flooding and inundation transport from the Waste Areas.	Simulating plug flow using a series of well-mixed tanks will not introduce significant error—dispersion is not a property of the medium but instead is used to describe heterogeneities in the flow. The number of vessels is chosen to produce an equivalent solution. Because neither settling nor volatilization is assumed to occur, the risk should be overestimated.
Exposure to Contaminants	The media (e.g., air, water, etc.) in or to which receptors are exposed to contaminants are assume to be well-mixed. Changes to exposure media, including the distribution of contaminants, are assumed to take place instantaneously.	Because contaminants are assumed to redistribute instantaneously and completely in exposure media, the impacts of heterogeneity are not included in the model. It is likely that these impacts are small relative to other variations in the system; however, exposures and risks are not necessarily overestimated.
Exposure to Dose Conversion—Radionuclides	Dose conversion factors (DCFs), used to convert from exposure to dose, are taken from International Commission on Radiological Protection (ICRP) Report 68 (ICRP 1995) (for workers) and ICRP Report 72 (ICRP 1996) (for the general public). The conversion factors are easily accessed using the Rad Toolbox (Eckerman 2003b).	The NRC uses dose as a metric to estimate risks associated with sites contaminated with radionuclides. The EPA prefers the use of slope factors to convert exposures to risks. Uncertainties in slope factors are neglected when in use[b]. Thus probabilistic assessments tend to be stochastic *exposure* assessments—not dose or risk assessments. For comparison purposes this will not be problematic.

a. Most of the key assumptions are taken from two basic references (Kennedy and Strenge 1992; NCRP 1996a). Assumptions not taken from these references are specifically cited in the table.

b. As illustrated in Chapter II, uncertainty and lack of knowledge are taken into account when defining slope factors (or reference doses) as upper bound risk factors for exposure to low doses of chemicals; however, this practice is different from characterizing the uncertainty in these factors for use in probabilistic risk analyses.

Table 22, Continued

Key Component	**Model Basis and Key Assumptions**[a]	**Likely Impact on Risk Estimate**
Exposure to Latent Cancer Risk Conversion—Radionuclides	This model will use the latest morbidity factors from either FGR-13 (Eckerman et al. 1999) or the Health Effects Assessment Summary Tables (HEAST) (USEPA 2001). The FGR-13 mortality conversion factors are used; these factors were taken from the FGR13CD database (Eckerman 2002) available from ORNL.	The NRC uses dose as a metric to estimate risks associated with sites contaminated with radionuclides. The EPA prefers the use of slope factors to convert exposures to risks. Uncertainties in slope factors are neglected when in use. Thus probabilistic assessments tend to be stochastic *exposure* assessments—not dose or risk assessments. For comparison purposes this will not be problematic.
Exposure to Latent Cancer Risk Conversion—Chemicals	The model will use the latest chemical slope (i.e., morbidity) factors from either the EPA IRIS database (USEPA 2006) or Health Effects Assessment Summary Tables (HEAST) (USEPA 2001). All risk factors were taken from the RAIS site (Dolislager 2006).	The EPA uses slope factors to convert from exposures to risks. Uncertainties in slope factors are not considered when used[b]. Thus probabilistic assessments tend to be stochastic *exposure* assessments—not dose or risk assessments. However, for comparison purposes this will not be problematic.
Exposure to Noncancer Risk Conversion—Chemicals	The model will use the latest chemical reference doses (RfDs) from the EPA IRIS database (USEPA 2006) to compute hazard indices corresponding to exposures. The conversion factors were collected from the RAIS site (Dolislager 2006).	The hazard quotient will be use to estimate noncancer risks for chemical exposure. Uncertainties in these factors are neglected when in use[b]. Thus probabilistic assessments tend to be stochastic *exposure* assessments—not dose or risk assessments. For comparison purposes, this will not be problematic.
Receptors	The following general public receptor scenarios will be considered: - Future on-site resident[c] - Future transient[c] - Off-site resident - Recreational user For workers, the following scenarios will be considered: - Direct worker - Support worker	Scenarios will be defined for both the general public (current and future) and workers. These scenarios (including transport and exposure pathways and parameters) will be defined to assure that any important effects will be captured. For deterministic analyses, bounding exposure parameters will be defined to overestimate resulting doses and risks.

a. Most of the key assumptions are taken from two basic references (Kennedy and Strenge 1992; NCRP 1996a). Assumptions not taken from these references are specifically cited in the table.

b. As illustrated in Chapter II, uncertainty and lack of knowledge are taken into account when defining slope factors (or reference doses) as upper bound risk factors for exposure to low doses of chemicals; however, this practice is different from characterizing the uncertainty in these factors for use in probabilistic risk analyses.

c. Future receptors will be exposed to site hazards only after the institutional control (IC) period has expired (USDOE 2000).

The Impacts of Simplifying Assumptions on Predicted Exposure Risks

It is very apparent when examining the information in Table 22 is that there is no way to guarantee, *a priori*, that risks computed from predicted exposures of radionuclides or chemicals will be overestimated even if that is the intent. For each assumption that produces increased transport of a contaminant through a given pathway (ostensively to overestimate the corresponding risk), transport through other pathways is necessarily reduced and the timing of transport through all pathways is impacted.

It is the confluence of risks that should be overestimated for screening purposes, which cannot be guaranteed by maximizing risks one exposure pathway at a time. The impacts of assumptions must be considered both individually (to estimate the maximum impact) and in combination to determine whether or not risks are overestimated. The impacts of the assumptions for maximizing risks can also differ from contaminant to contaminant. Although there is no way to guarantee that risks will be overestimated, it should be possible to assure that risks are not grossly underestimated, which is important when comparing exposure risks to non-exposure risks from standard industrial practices.

Simplifications Used for Screening Standard Industrial Risk Modeling

The primary difference between the screening risk analysis tool developed in this research and others is the integration of risks from *both* exposures to radionuclides and chemicals and more typical standard industrial accidents. Standard industrial risks are defined in this research to be the *non-exposure* risks associated with falls, explosions, transportation accidents, etc. These more mundane risks are often omitted when selecting remedial actions for buried waste sites even though these sites often resemble heavy construction areas and hazards are dominated by standard industrial risks (Applegate and

Wesloh 1998; Gerrard and Goldberg 1995). Table 23 from Brown et al. (2005) provides examples of *high-risk* hazards[106] that are likely when dispositioning buried waste sites. The *high-risk* hazards are dominated not by exposures to buried or retrieved wastes but instead by the standard industrial accidents associated with remedial activities. Accident risks should be considered when selecting a remedial alternative to provide a clear and comprehensive picture of the risks associated with site cleanup.

An excellent method for evaluating the risks for a process is to use the risk-triplet described by Kaplan and Garrick (1981) as a guide. The first step in evaluating the risk-triplet is to generate the set of scenarios describing what can go wrong. For engineered disposal sites, the list of features, events, and processes (FEP)[107] is developed and used to define a comprehensive set of scenarios (Kaplan et al. 2001; Park et al. 2002; Swift et al. 1999). However, a FEP analysis for potential remedial actions would likely be prohibitive for the type of screening risk analysis developed in this research. Existing FEP analyses should be used whenever possible.

There are many different types of workers (e.g., construction, support, office, managerial, etc.) and activities required during remedial activities. The combinations of workers and activities would quickly swell to a complicated set of scenarios for which data are unlikely available. Thus scenario analysis for the screening analysis will be simplified to a single, yet meaningful, general scenario for the screening risk analysis.

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[106] Table 23 is organized using the risk-triplet concept of Kaplan and Garrick (1981). *High-risk* hazards are those considered to be 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences as defined in Chapter III.

[107] *Features* (e.g., waste form, fractures, etc.) are physical, chemical, thermal, or temporal characteristics of the system. *Processes* (e.g., percolation) are typically phenomena and activities that have gradual, continuous interactions with the system. *Events* (e.g., volcanism) are, in general, discrete occurrences.

Table 23. Summary of *High-Risk* Hazards for the Idaho Site Subsurface Disposal Area (SDA) Remedial Alternatives (Brown et al. 2005)

Process Step	**No Action**	**Retrieval**	**Manage in Place**	**What can go wrong?**[a]	**How likely is it?**[a]	**What are the consequences?**[a]	**Who is impacted?**
Characterization	√	√	√	• No high-risk hazards	• N/A	• N/A	• N/A
In Situ Grouting		√	√	• Failure of high-pressure grout system resulting in projectiles of grout release	• Probable	• Severe	• Worker
Excavation and Waste Retrieval		√		• Soil removal resulting in radiological or toxic exposure • Loaded tote-bin dropped outside confinement releasing radioactivity • Traumatic injury (e.g., excavation cave-in)	• Probable • Probable • Possible	• Critical • Critical • Severe	• Worker • Worker • Worker
Ex Situ Treatment		√		• No high-risk hazards	• N/A	• N/A	• N/A
Packaging		√		• Containment/ventilation system failure resulting in exposure	• Possible	• Severe	• Worker
Intermediate Storage		√		• No high-risk hazards	• N/A	• N/A	• N/A
Surface Barrier Installation		√	√	• No high-risk hazards	• N/A	• N/A	• N/A
On-Site Disposal		√ √ 	√ √	• Traumatic injury (e.g., during preparation or interring) • Failure of engineered system • Failure of engineered system	• Possible • Probable • Probable	• Severe • Severe • Critical	• Worker • Public • Public
Monitoring and Maintenance	√	√	√	• No high-risk hazards	• N/A	• N/A	• N/A
Off-Site Disposal		√		• Traumatic injuries from heavy equipment operation	• Probable	• Critical	• Worker

a. These three questions comprise the *risk-triplet* concept proposed by Kaplan and Garrick (1981).

In general, workers' activities and their relationship to risk can be abstracted in the following manner. The risk-related questions concerning what can go wrong and what are the consequences can be integrated into a single scenario-consequence pair that a worker is either injured or killed (*consequences*) while working (*scenario*). It is important to place the evaluation of standard industrial risks on a firm theoretical basis like has been done for exposure risks. The *probability* that a worker will either be injured or killed is assumed proportional to the time worked for screening risk analysis. These risks can be represented by:

$$\underbrace{\text{Injuries}(1/\text{year})}_{\textit{Consequences}} = \underbrace{(\text{Hours worked}/\text{year})}_{\textit{Probability}} \times \underbrace{(\text{Injuries}/\text{Hours worked})}_{\textit{Scenario-Consequences}} \qquad [4]$$

$$\text{Fatalies}(1/\text{year}) = (\text{Hours worked}/\text{year}) \times (\text{Fatalities}/\text{Hours worked}) \qquad [5]$$

A great deal of effort has been expended in collecting the statistics needed to relate time worked to injuries and fatalities. (USDOL 2005).

The accuracy of standard industrial risks for a screening level analysis is directly related to the estimates of the levels of effort required to perform the work and the statistics describing the potential risk while working. Typically a great deal of effort is needed to perform even high-level estimates of the levels of effort and skills mix required to perform the work. The uncertainties in the statistics used to relate the time worked to risk are based on finding pertinent statistics for the type of worker and conditions. For example, many of the accepted statistics are only as good as the reporting systems that collect the data. General statistics may be inappropriate for specific site locations. As for exposure risks, site-specific information should be used whenever it is available.

The Impacts of Simplifying Assumptions on Predicted Standard Industrial Risks

Although there are uncertainties in the standard industrial risks estimated for a remedial action, it is likely that these uncertainties are significantly smaller than those estimated for exposure risks, especially exposure risks to low doses of hazardous chemicals. Consider the following analogy (using injury risk as an example that also holds for fatality risk):

$$
\begin{array}{ccccc}
\text{Injuries}\left(1/\text{year}\right) & = & \left(\text{Hours}/\text{year}\right) & \times & \left(\text{Injuries}/\text{Hours}\right) \\
\underset{\textit{Morbidity}}{\text{Latent cancers}\left(1/\text{year}\right)} & = & \underset{\textit{Exposure}}{\left(\text{Intake}/\text{year}\right)} & \times & \underset{\textit{Slope factor}}{\left(\text{Latent cancers}/\text{Intake}\right)}
\end{array}
\qquad [6]
$$

For example, morbidity risk is often used to determine whether or not a contaminated site requires action and the corresponding cleanup levels. The cleanup analysis does not account for uncertainties in the slope factor used (and often not in the intake rate). Often the uncertainty in predicted exposure and slope factor can be several (and perhaps as many as seven) orders of magnitude (Linkov and Burmistrov 2003). For injury or fatality risk parameters, labor estimates or injury or fatality rates are unlikely to be off by several orders of magnitude. Whereas slope factors are derived from models relating low doses to risk, injury or fatality factors are estimated from data from actual injuries and fatalities. Thus the standard industrial risks are likely to be more certain than those for exposure.

Conclusions

To help manage the complexity of estimating risks for buried waste site disposition and making remedial decisions under large uncertainties, a conceptual burial model is defined so that the aforementioned uncertainties are not problematic. Models developed for the conceptual burial site adequately represent, *for a screening-type risk analysis*, the contaminants, waste types, releases, fate and transport, exposures, and receptors for DOE buried waste sites. The decision then becomes how well the conceptual burial model describes a DOE buried waste site and how meaningful will be the results. However, more importantly, the usefulness of the conceptual burial model is like that of the risk assessment process itself, as an organizational tool to help focus attention on the critical assumptions made and the significant contributors to risk and impacts of uncertainty, which are factors critical to the decision-making process. DOE buried waste sites are sufficiently complex that any tool shown to provide useful information for these sites should find much broader applicability elsewhere.

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CHAPTER VI

A NOVEL LIFE-CYCLE RISK ANALYSIS SCREENING TOOL FOR DEPARTMENT OF ENERGY (DOE) BURIED WASTE SITES

A conceptual burial site model was defined in Chapter V to help manage the complexity and large uncertainties in estimating risks for the disposition of Department of Energy (DOE) buried waste sites. This conceptual model embodies, *for a screening-type analysis*, the contaminants, waste types, releases, fate and transport, exposures, and receptors representative of DOE buried waste sites. This chapter describes the implementation of the conceptual burial site model in a form that provides screening estimates of the exposure and accident risks and uncertainties for proposed remedial actions for comparison purposes.

Screening Risk Tool Overview

The screening conceptual burial model was implemented in the GoldSim Monte Carlo simulation software (GTG 2005b; c) employing the Radionuclide Transport (RT) Module (GTG 2005a). The GoldSim software allows critical components described in Chapter V to be captured for a screening risk analysis. The exposure and risk estimates made using the screening risk tool developed in this research are based on the best information possible. However, any such software tool may be used erroneously or correctly applied to the wrong site. Even if applied correctly, the exposure and risk results generated using this or any such software tool are subject to different interpretations.

The primary driver for exposure risks from buried wastes is the transport of contaminants from the site to receptors in the environment. Contaminant transport in GoldSim is represented as mass fluxes among media (e.g., air, water, soil, etc.). If these fluxes were known, there would be no need for modeling and exposure concentrations (and doses and risks) could be estimated directly. However, mass fluxes vary temporally and spatially, and estimating these fluxes requires modeling the features and processes (e.g., inventory, release, transport, etc.) needed to define inter-media fluxes. GoldSim allows the important features and processes to be modeled either deterministically or stochastically to analyze the impact of uncertainty on the predicted exposures and risks.

The screening risk tool uses the generic performance assessment (PA) model[108] by Tauxe (2004; 2005) as an initial foundation. The Tauxe model provides an excellent example for estimating exposure risks related to the final disposal of radioactive wastes. The Tauxe PA model (Tauxe 2004; 2005) was expanded to include new radionuclides, hazardous chemicals, fractured and surface water media, new transport pathways and receptors, and standard industrial risks. Any errors or omissions in the screening risk tool developed in this research should not be attributed to the Tauxe generic PA model.

The screening risk tool describes both arid and humid conditions representative of the prototypic sites (i.e., Idaho Site Subsurface Disposal Area and Oak Ridge Bear Creek Burial Grounds) and estimates both exposure and industrial risks for baseline conditions as well as before, during, and after remedial actions have been performed. For this research, remedial actions are categorized as either managing buried wastes in-place or retrieving wastes for treatment and ultimate disposal elsewhere.

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[108] The generic performance assessment model (Tauxe 2004; 2005) is available at http://www.neptuneandco.com/goldsim/generic/index.html (accessed March 13, 2008).

The top level of the screening risk analysis tool, denoted the Conceptual Burial Site Model (CBSM), is illustrated in Figure 30. The description of the screening risk tool is organized much like the tool itself as illustrated in Figure 31. Exposure media through which receptors could be exposed are first defined. Inventory and release properties for the waste sites are described. Transport pathways and properties are defined describing contaminant migration from source to receptors. Receptors are also defined because a source, transport to receptor, and exposure of a receptor are required for there to be risk. On the other hand, the mere presence of workers performing routine actions presents standard industrial risks. Both exposure and industrial risks tend to increase during remedial activities either through increased exposure or more hazardous operations.

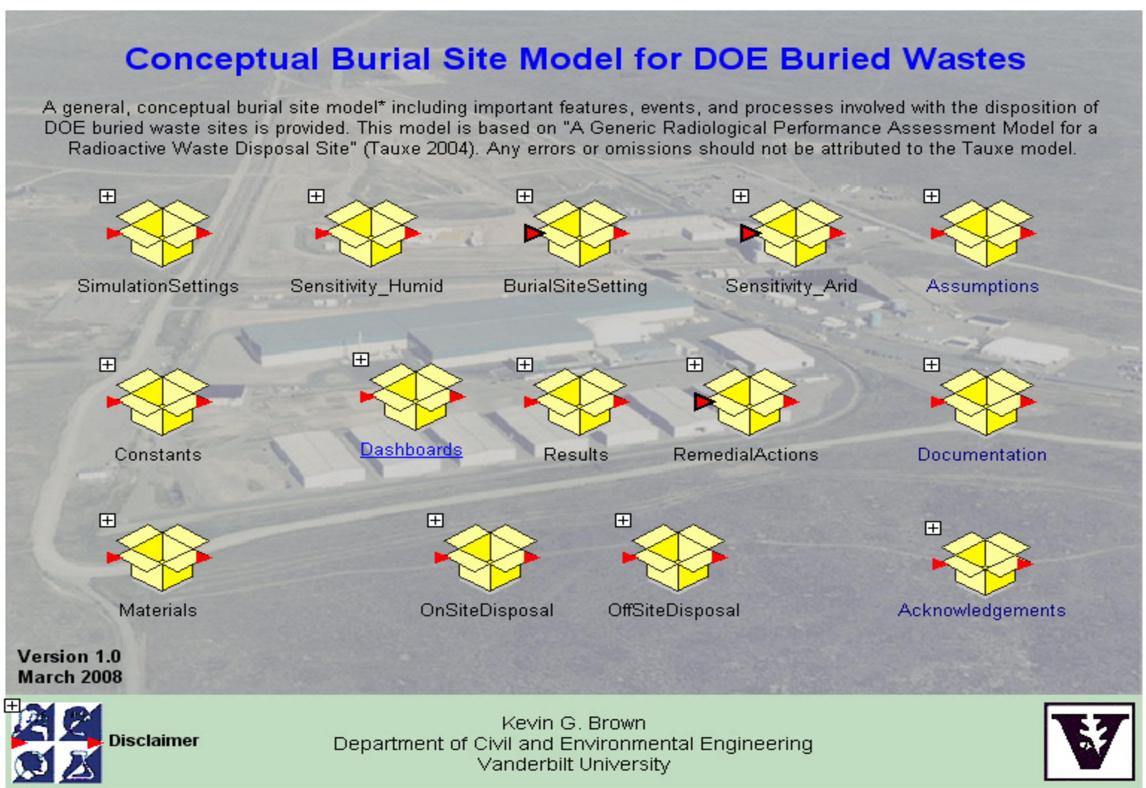


Figure 30. Screening Risk Tool as Implemented in GoldSim

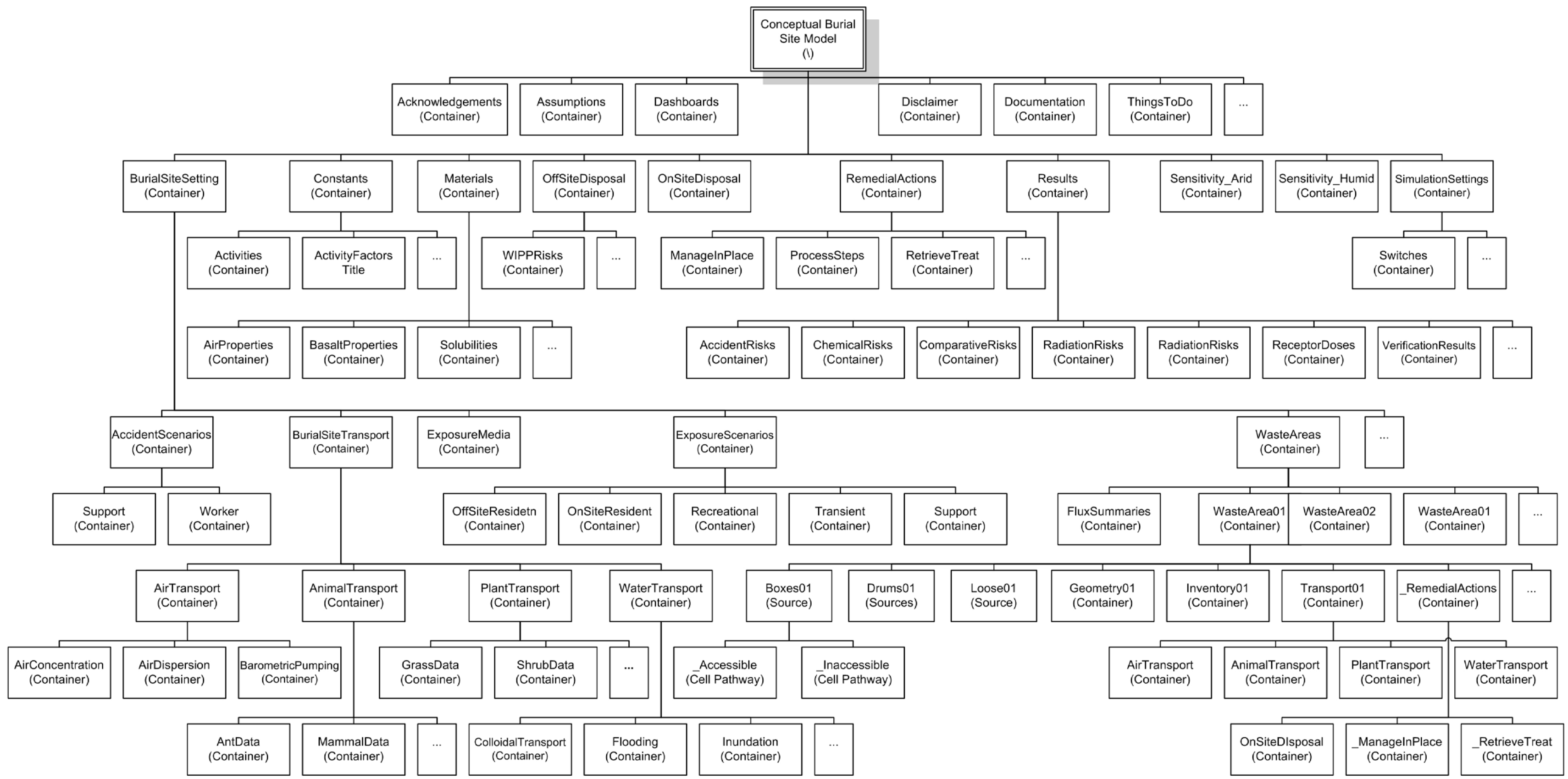


Figure 31. Example organizational structure of the Conceptual Burial Site Model in GoldSim. The Waste Area and Remedial Action containers in the lower right-hand side of the structure have some details not shown for clarity. GoldSim also provides an optional Browser View showing the containment hierarchy in tree form (GTG 2005b).

Exposure Media

The screening risk tool can be described in terms of the necessary components (i.e., exposure media, inventory, source, and release by waste type, transport pathways, and potential receptors) of the system. At a fundamental level, the burial model is defined by the media through which contaminants migrate and receptors may be exposed. These are the media that must be either managed in-place or excavated so the waste and contaminants can be retrieved for treatment and disposal elsewhere.

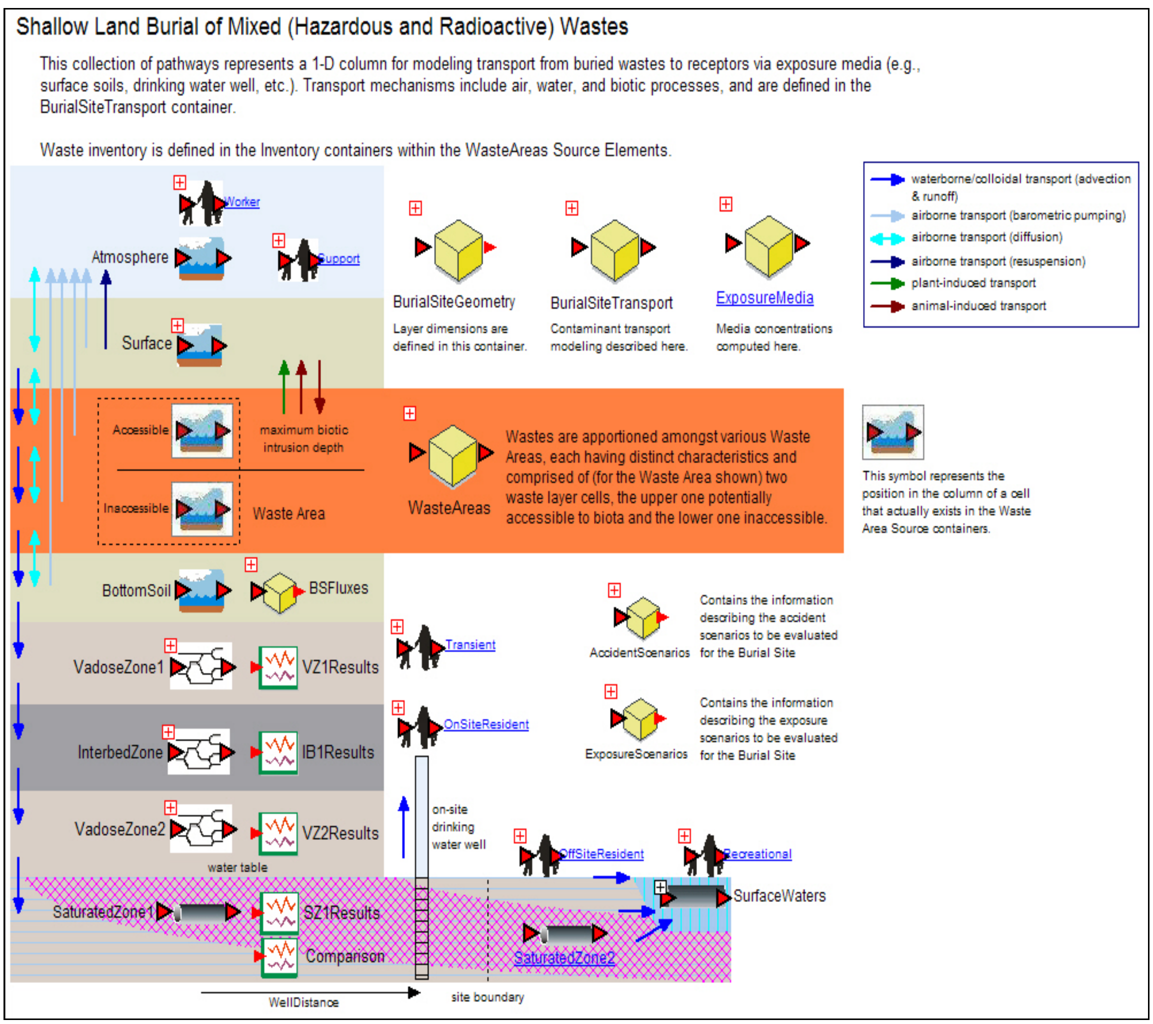


Figure 32. Burial Site Setting including Exposure Media and Receptors

The burial site setting including exposure media for the conceptual burial site model in Chapter V is illustrated in Figure 32. Media are defined so that either arid conditions (e.g., little percolation flow; deep, fractured vadose zone and long travel times; interbed regions; and primarily groundwater impacts) or humid conditions (e.g., large percolation flow, karst zone with preferential flows, short travel times, and primarily surface water impacts) can be described. The use of arid or humid conditions is controlled by settings or "switches" in the GoldSim model SimulationSettings container and dashboard as illustrated in Figure 31 and Figure 33, respectively.

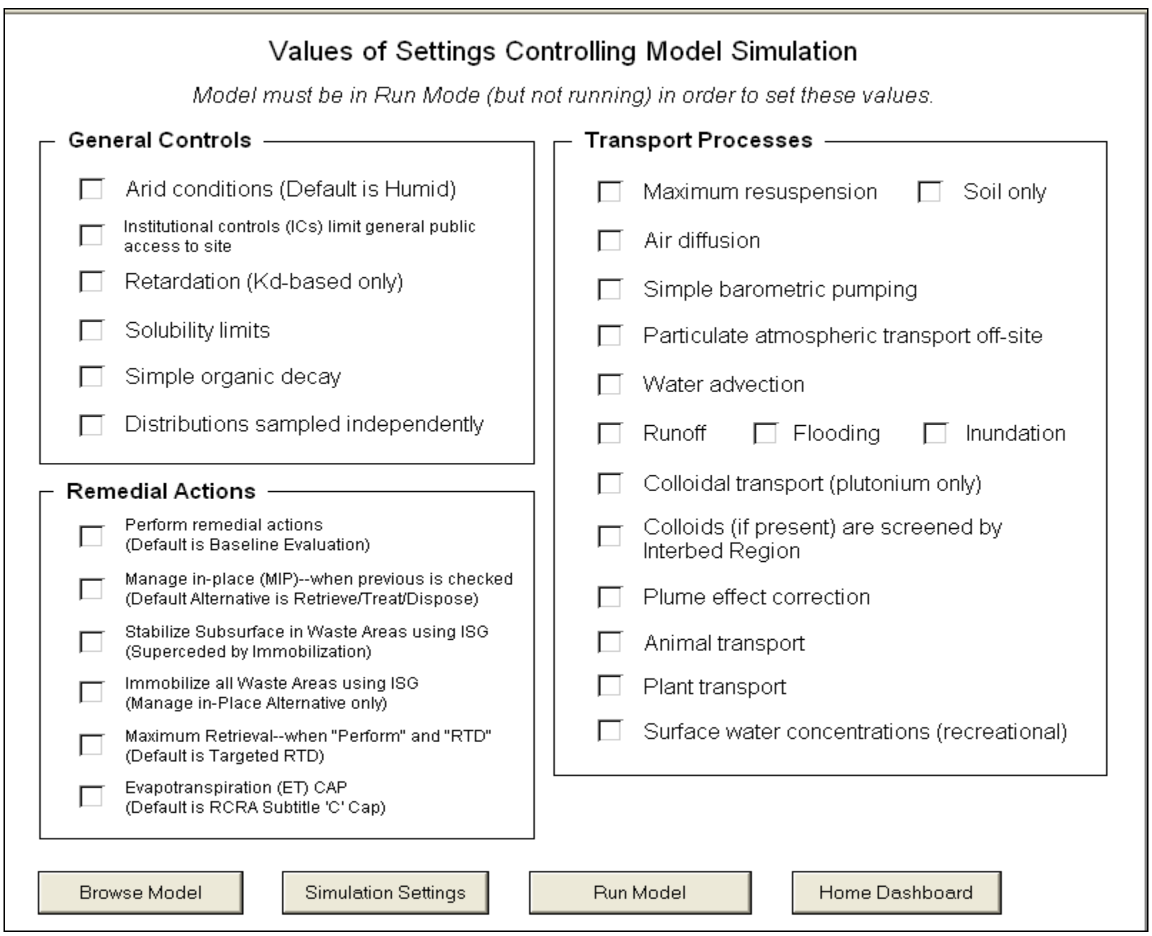


Figure 33. The Control Settings Dashboard for the Screening Risk Tool

The exposure media in Figure 32 represent transport pathways leading from sources to potential receptors. Another way to visualize the media and transport pathways is provided in Table 24, which describes the media and exposure pathways in the burial model relative to the generic PA model by Tauxe (2004; 2005). The current model is more comprehensive in terms of both exposure and industrial risks than that by Tauxe (2004; 2005) and can be used for baseline, remedial action, and post-closure conditions.

GoldSim Elements Used to Describe Exposure Media

The GoldSim elements shown in Table 24 are used to model exposure media for the conceptual burial site model. A brief description of each follows (GTG 2005a; b)

Expression—This element, similar to a cell in a spreadsheet, produces a single numerical or conditional output using a user-supplied formula.

Cell Pathway—This element models a well-mixed tank and can be used to model partitioning, solubility, and mass transport for multiple fluid media.

Pipe Pathway—This element is a fluid conduit where mass enters, advects and disperses, and exits the other end. It can contain a single fluid medium and solid, porous media, which may impact transport via porosity and sorption. Solubility constraints cannot be modeled in a pipe element.

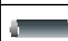
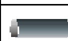
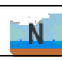
Network Pathway—This element provides a computationally-efficient solution (using a Laplace transform algorithm) to fractured flow using a series of *Pipe Pathway* elements. The number of "pipes" can be very large (e.g., 100,000) albeit grouped by a much smaller set of properties.

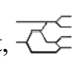
Cell Pathway Elements in Series—A series of linked *Cell Pathway* elements is used to represent a one-dimensional pathway (e.g., stream or river). Longitudinal dispersion is controlled by the number of *Cell Pathway* elements used[109].

[109] Dispersivity is not a physical property of the medium; this scale-dependent parameter is instead a way to describe flow heterogeneities within the pathway (GTG 2005a). A series of *Cell Pathway* elements, which provides an equivalently dispersive signal, is needed when the error from using a *Pipe Pathway* element is too large to be tolerated (GTG 2005a). This error was discovered when attempting to use a *Pipe Pathway* to model the surface water pathways for the conceptual burial site model.

The bases for selecting these GoldSim elements to describe the necessary media to predict exposure and risk will be described in the sections to follow.

Table 24. Media and Pathways for Two Performance Assessment Models

	Conceptual Burial Site Model[a]	Tauxe (2005) Generic PA Model	Diffusion (Air)[b]	Diffusion (Water)	Advection (Air)	Advection and Dispersion (Air)	Barometric Pumping (Advection, Air)	Advection (Water)	Colloidal (Advection, Water)	Flooding (Advection, Water)	Inundation (Advection, Water)	Resuspension (Advection, Soil)	Runoff (Advection, Soil)	Plant-Induced Transport (Soil)	Animal-Induced Transport (Soil)
Off-Site Atmosphere					↑	↑									
Atmospheric Layer			↓		↑s	↑	↑					↑			
Surface Soil/Cap Layers[c]			↕				↑	↓		↓		↑	↓	↑	↓
Accessible Waste Area			↕				↑	↓	↓	↓	↓			↑	↑
Inaccessible Waste Area			↕				↑	↓	↓	↓	↓				
Bottom Soil			↑				↑	↓	↓	↓	↓				
Upper Vadose Zone								↓	↓						
InterBed Region								↓	↓						
Lower Vadose Zone								↓							
Local Saturated Zone								↓		↓	↓				
Off-Site Saturated Zone								↓s							
Local Surface Water								↓		↓	↓		↓		
Off-Site Surface Water								↓s							

a. The GoldSim implementation includes:  *Expression*,  *Cell* element,  *Fracture Network*,  *Pipe* element, and  multiple *Cell* elements in series to represent plug flow (GTG 2005a).

b. Arrows (i.e., ↑ or ↓) indicate the direction of transport. Shaded areas indicate new transport mechanisms versus those in the model by Tauxe (2005). Diffusion of contaminants in water is not implemented because transport via the water pathway is likely dominated by advection. An 's' indicates transport to a sink or from a source.

c. The Generic PA model used a single cap comprised of four layers (implemented as *Cell Pathway* elements) (Tauxe 2005). The current model can use 1) a single soil layer to represent baseline conditions or 2) an evapotranspiration (ET) or RCRA Subtitle 'C' type cap (Mattson et al. 2004).

The Atmospheric Pathway

An important pathway for potential exposure to contaminants from buried waste sites is the atmosphere, especially the air above the burial site. A "box" model is used via a GoldSim *Cell Pathway* element to represent the air in contact with the burial site where the height of the "box" is the average mixing height for the area as illustrated in Figure 34. Although mixing heights vary both with time of day and season, the average mixing height is used to provide an approximation of contaminant concentrations reasonable for *screening-level analyses* of exposure risks to those in the vicinity of the burial site (Cohen and Cooter 2002). The contaminant concentrations in the atmospheric layer, which are assumed uniform, are controlled by contaminant fluxes into the layer from the burial site (assuming no import fluxes from external areas) and the advective export rate (inversely proportional to the average wind speed) at which material is removed from the layer as well as any decay or ingrowth processes.

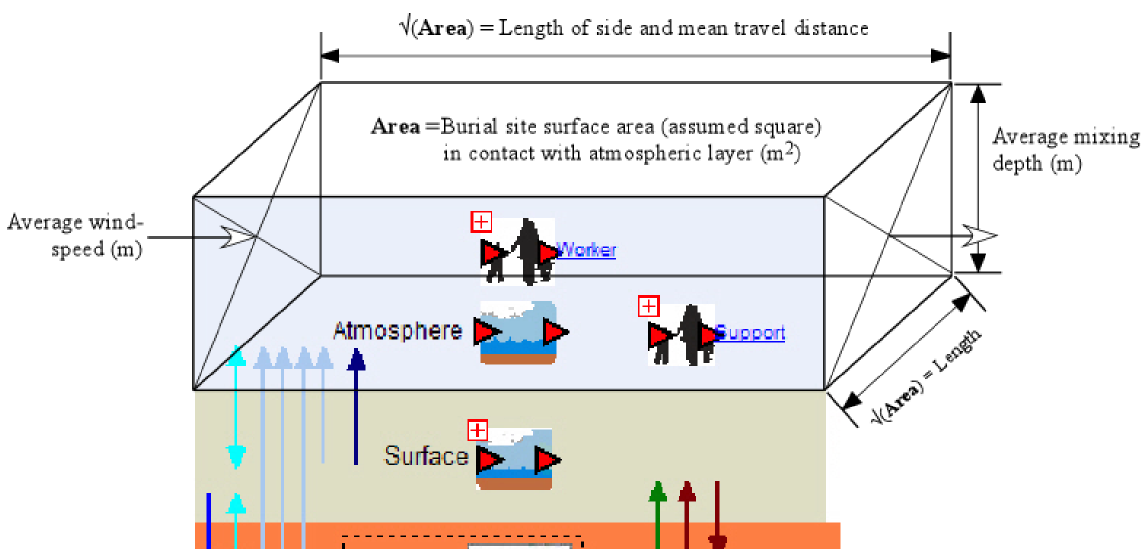


Figure 34. Atmospheric "Box" Model as Implemented using a GoldSim *Cell Pathway*

The parameters controlling the magnitude of predicted exposure concentrations (and thus doses and risks) for the "box" model in Figure 34 are described briefly in Table 25. Because the GoldSim model can be used to predict both point-value exposure risks as well as the distribution of exposure risks due to uncertainties in the parameters used for prediction, both point-value parameters and distributions (when applicable)—for arid and humid conditions—are provided in Table 25.

Table 25. Parameters Describing the Atmospheric "Box" Model in Figure 34

		Point-Value		**Probabilistic**		
Parameter	**Conditions**	**Value**	**Reference**	**Distribution**[a]	**Reference**	**Comments**
Total waste volume	Arid	3.56×10^6 m^3	(Holdren et al. 2006)	D(3.56×10^6 m^3)	(Holdren et al. 2006)	There are no data describing uncertainties in the waste volume. It is assumed that the uncertainty would be small relative to other uncertainties (i.e., mixing depth) and the predicted risks insensitive.
	Humid	1.90×10^5 m^3	(SAIC 1996a)	D(1.90×10^5 m^3)	(SAIC 1996a)	
Average waste thickness	Arid	10 m	(Holdren et al. 2006)	D(10 m)	(Holdren et al. 2006)	No information was found for uncertainty in waste thickness. It is assumed the uncertainty would be small relative to others (i.e., mixing depth) and the predicted risks insensitive.
	Humid	19.5 ft[b] (5.94 m)	(SAIC 1996a)	D(5.94 m)	(SAIC 1996a)	
Average mixing depth	Arid	2 m	(Ho et al. 2005; Yu et al. 1993)	LN(1880 m, 1892 m)[c]	(Clawson et al. 1989)	The deterministic value, equal to the approximate height of a person, was used in Tauxe (2004) and should be smaller (providing higher predicted concentrations and risks) than that from meteorological data.
	Humid	2 m	(Ho et al. 2005; Yu et al. 1993)	LN(768 m, 773 m)	(ORNL 2006)	
Average wind speed	Arid	7.1 mi/hr (3.17 m/s)	(Clawson et al. 1989)	N(7.1 mi/hr, 0.283 mi/hr)[d]	(Clawson et al. 1989)	Uncertainty in wind speed is included because it controls export of contaminants from the atmosphere above the site.
	Humid	3.4 mi/hr (1.52 m/s)	(ORNL 2006)	N(3.4 mi/hr, 0.283 mi/hr)[d]	(ORNL 2006)	

a. The distributions used in the GoldSim model include: *Discrete* D(point value), *LogNormal* LN(arithmetic mean, standard deviation), and *Normal* N(mean, standard deviation).

b. The value used is the average of the trench depths (14 ft to 25 ft) given in Table 3.3 of SAIC (1996a).

c. The standard deviation provided for the humid conditions (ORNL 2006) implies that the distribution is not Gaussian; therefore, a lognormal distribution is used. For lack of better information, the same relative standard deviation (i.e., 1880*(773/768) = 1892 m^3) is used for arid conditions.

d. The standard deviation, 0.283 mi/hr (0.126 m/s), is obtained from the two values (7.1 and 7.5 mi/hr) provided for the Idaho Site for the same year (Clawson et al. 1989) and may underestimate the uncertainty. For lack of better information, this standard deviation is used for both sites.

The "box" model in Figure 34 can provide a reasonable first approximation of the concentrations (and exposure doses and risks) in the area above the buried waste site for a screening-level analysis. However, there may be important receptors who are either outside the "box" described by the model or require more accurate risk predictions. A convenient and oft-utilized model that can be employed for these receptors is the Gaussian plume model (Lamarsh 1983).

The Gaussian plume model is diffusion-based with a constant emission source that incorporates empirical dispersion coefficients that are functions of wind velocity, atmospheric stability, and distance from the source. The model, which is based on the assumption that there is also no contaminant deposition, can provide reasonably accurate concentration predictions up to 10,000 m from the source (Lamarsh 1983). Use of the Gaussian plume model for the CBSM is illustrated in Figure 35 (where the off-site receptor as shown in Figure 32 has been moved for the purpose of illustration).

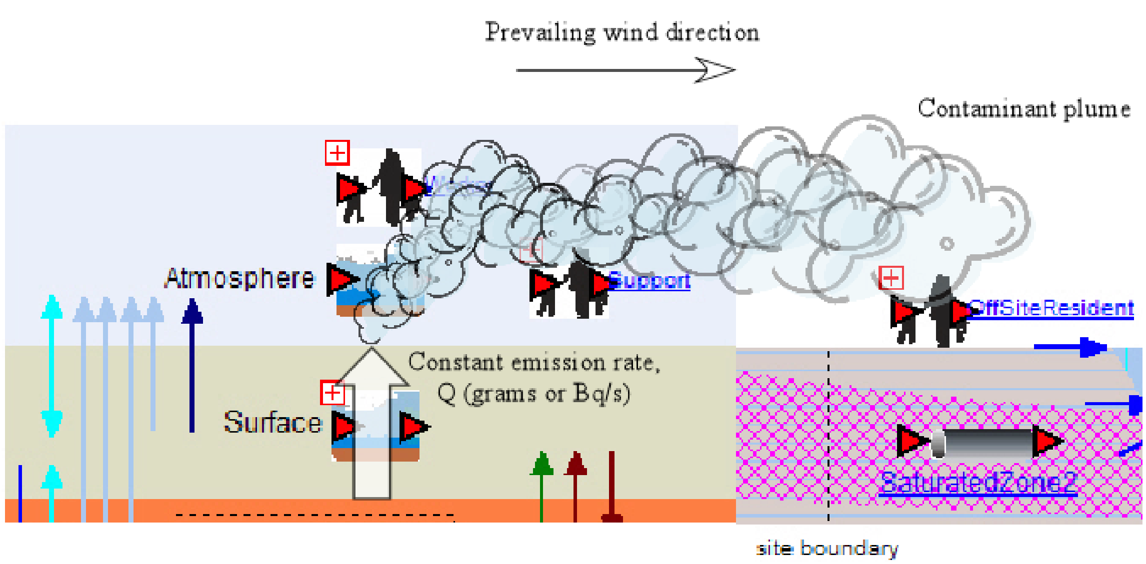


Figure 35. Gaussian Plume Model as Implemented in the Conceptual Burial Site Model

The effluent concentration, χ, provided by the Gaussian plume model is a maximum along the centerline of the plume. For a ground-level release, the maximum concentration, χ_{max}, is given by (Lamarsh 1983):

$$\chi_{max} = \frac{Q}{\pi v \sigma_y \sigma_z} \qquad [7]$$

where Q is the emission rate (grams or Bq/s), ν the average wind speed (m/s), and σ_y and σ_z the horizontal and vertical dispersion coefficients (both in meters), respectively. Often both sides of Equation 7 are divided by the emission rate, Q, to provide "dilution factors", i.e., values of (χ_{max}/Q) that are independent of the emission rate. Software is available[110] to generate tables of dilution factors that can be used (knowing the direction and distance to the source) to evaluate (by multiplying by the emission rate) the maximum contaminant concentration for the given scenario.

Although tables of dilution factors can be derived for the scenarios considered in the screening risk tool, the process is simplified further by using accepted values of dilution factors based upon the distance and direction to the receptor in question as illustrated in Table 26. The emission source term is the combined emissions from diffusion of contaminants from the surface soil (Tauxe 2004) and barometric pumping from the contaminated layers below the surface[111].

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[110] One example of software that provides dilution factors is the CAP-88 program available from the U.S. EPA at http://www.epa.gov/radiation/assessment/CAP88/index.html (accessed March 13, 2008). A table of (χ_{max}/Q) values can be exported from the CAP-88 program for use in different scenarios.

[111] The diffusion pathway is described in the section entitled *Intermedia Diffusion via the Vapor Phase* on p. 311 and barometric pumping is described in *Advection in the Water and Atmospheric Phases including Barometric Pumping* on p. 314.

Table 26. Parameters Describing the Gaussian Plume Model in Figure 35

		Point-Value		Probabilistic		
Parameter	**Conditions**	**Value**	**Reference**	**Distribution**[a]	**Reference**	**Comments**
Noninvolved Worker Location	Arid	100 m S	(USDOE 1997)	D(100 m)	(USDOE 1997)	Distances are determined from dilution factors and are not stochastic.
	Humid	100 m SW	(USDOE 1997)	D(100 m)	(USDOE 1997)	
Noninvolved Worker (χ_{max}/Q)[b]	Arid	5.8×10^{-3} s/m^3	(USDOE 1997)	LN(3.3×10^{-3} s/m^3, 1.3×10^{-3} s/m^3)[c]	(NRC 2004; USDOE 1997)	The deterministic value corresponds to the upper 95% value from the distribution.
	Humid	5.8×10^{-3} s/m^3	(USDOE 1997)	LN(3.3×10^{-3} s/m^3, 1.3×10^{-3} s/m^3)[c]		
Maximally Exposed Individual[d] **Location**	Arid	4000 m ENE	(USDOE 1997)	D(4000 m)	(USDOE 1997)	These distances based on maximum exposures and are not stochastic.
	Humid	720 m NNW	(USDOE 1997)	D(720 m)	(USDOE 1997)	
Maximally Exposed Individual[d] **(χ_{max}/Q)**[b]	Arid	5.1×10^{-5} s/m^3	(USDOE 1997)	LN(2.9×10^{-5} s/m^3, 1.2×10^{-3} s/m^3)[c]	(NRC 2004; USDOE 1997)	The deterministic value corresponds to the upper 95% value from the distribution.
	Humid	2.8×10^{-3} s/m^3	(USDOE 1997)	LN(1.6×10^{-3} s/m^3, 6.4×10^{-4} s/m^3)[c]		

a. The distributions used in the GoldSim model include: *Discrete* D(point value) and *LogNormal* LN(arithmetic mean, standard deviation).

b. These dilution factors (namely "E/Q") are referred to in USDOE (1997).

c. The relative standard deviation ($1.873 \times 10^{-6}/4.605 \times 10^{-6} = 0.4$) used to define the lognormal distributions is computed from data in NRC (2004).

d. The maximally-exposed individual (MEI) is a "hypothetical member of the public who is exposed to a release of radioactive or chemically hazardous material in such a way… that the individual will likely receive the maximum dose from such a release" (USDOE 2002).

Two models, a "box" and a Gaussian plume model, were developed for predicting exposure concentrations and risks to workers and the general public for the atmospheric pathway. Because both the general public and workers may reside outside the limits defined by the "box" model illustrated in Figure 34, the plume model was implemented to allow prediction of exposure concentrations and risk for these other important receptors. However, if the support worker (or "Noninvolved Worker" in Table 26) resides within the limits of the "box" model, then there is an opportunity to assess uncertainties in both the conceptual and mathematical models used to describe the atmospheric pathway. By definition, the direct worker resides in the limits defined by the "box" model in Figure 34.

Other than the transport mechanisms that define the emission rates into the atmospheric layer, two important properties of this layer remain to be described. These are the (non-air) media within the layer that represent water vapor and resuspended soil particles that impact both contaminant partitioning and net transport. The determination of the concentration of suspended particulates in the atmosphere is described below[112].

The air temperature is required to determine the water vapor concentration in the *Cell Pathway* element used to model the atmosphere above the buried waste site. For example, the saturated vapor density, V_D (g/m^3), can be related to the air temperature, T(K), using (Federer and Lash 1978; Murray 1967):

$$V_D = 216.7\left(\frac{V_P}{T}\right) \quad \text{where} \quad V_P = 6.1078\exp\left(\frac{17.27(T-273.15)}{T-35.86}\right) \qquad [8]$$

The saturated vapor density is multiplied by the average relative humidity to obtain the water vapor concentration in the *Cell Pathway* element in Figure 34.

The average air temperature for arid conditions is 278.8K (42.1°F) (Clawson et al. 1989), and that for humid conditions is 287.6K (14.4°C) (ORNL 2006). The average relative humidity values for arid and humid conditions are 50% (Clawson et al. 1989) and 72% (BJC 1999), respectively. Temperature data for the Idaho and Oak Ridge sites indicate that there may be little variation (perhaps less than 1%) in *average* temperatures. No relevant data could be found describing relative humidity variations. To estimate the impacts of variation in temperature and humidity on exposure concentrations and risks, Gaussian distributions centered at the average values with an arbitrary 5% uncertainty are used. Point-value calculations use average temperature and relative humidity values.

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[112] Please refer to *Additional Advective Transport Mechanisms including Colloidal Transport, Runoff, and Resuspension* on p. 323 and Table 31 for details on resuspension.

Top Soil and Surface Barriers

Under normal circumstances, the primary pathways of transport of contaminants to the atmosphere originate in the surface soil layer (which would become the top soil layer if the burial site is capped). The GoldSim representation of the soil and cap layers in contact with the atmosphere is illustrated in Figure 36. For baseline conditions, advective and diffusive fluxes from the SurfaceSoil *Cell Pathway* element (Original Site) are active. When capping the site, either an idealized evapotranspiration (ET) or RCRA Subtitle 'C' cover (Mattson et al. 2004) is selected and the fluxes from the pertinent TopSoil *Cell Pathway* element become active replacing those from the SurfaceSoil element.

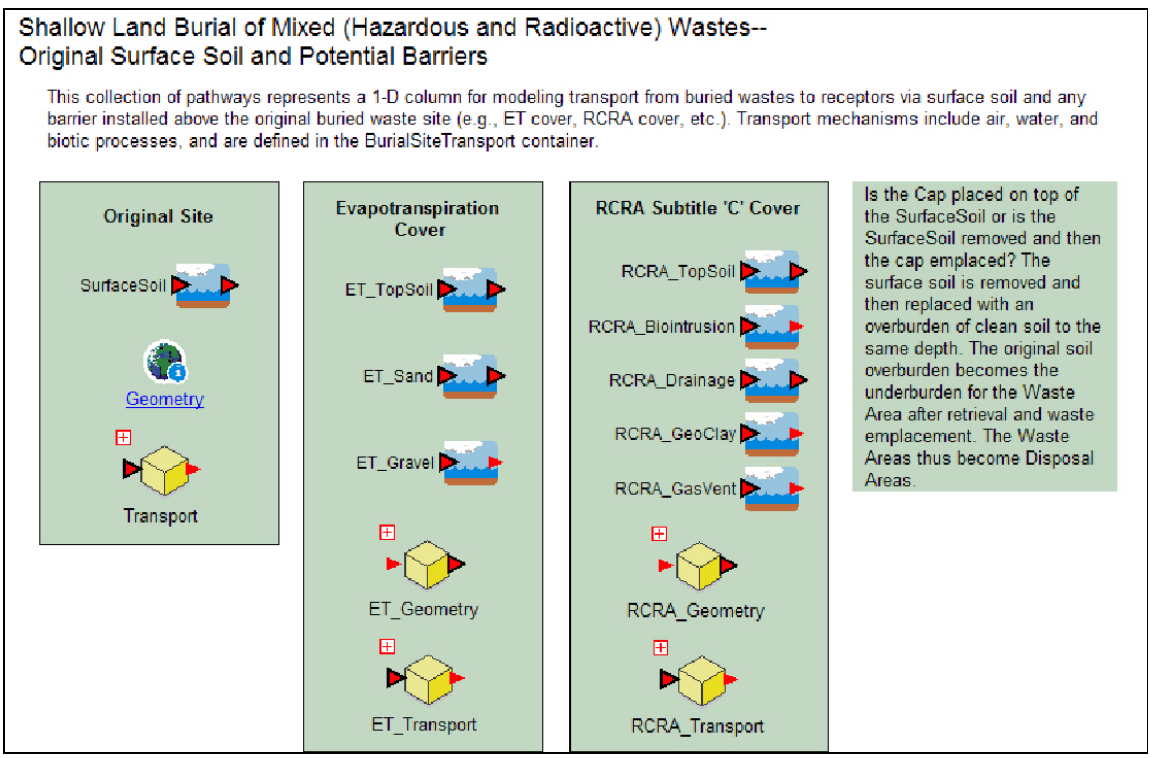

Figure 36. GoldSim representation of the Surface Layers for the CBSM including either an Evapotranspiration or RCRA Subtitle 'C' Cover.

Cross-sectional views of the types of covers that can be modeled in the CBSM are provided in Figure 37 (Mattson et al. 2004). When comparing Figure 36 to Figure 37, it is apparent that idealized representations of the covers were modeled in GoldSim. For example, the five layers in the evapotranspiration (ET) cover are modeled using three *Cell Pathway* elements or "boxes" (similar to the atmospheric "box" in Figure 34). In the GoldSim model, the top soil layer is 140 cm deep. The middle sand layer remains 30 cm deep as illustrated in Figure 37. The bottom layer of the ET cover is 90 cm of gravel.

The five layers of the RCRA cover remain the GoldSim model; however, the geomembrane is not directly represented in the model, instead the water flux from the drainage to clay layer is assumed to be zero unless the cap has failed[113]. The cover groupings and properties are based on the layer descriptions in (Mattson et al. 2004).

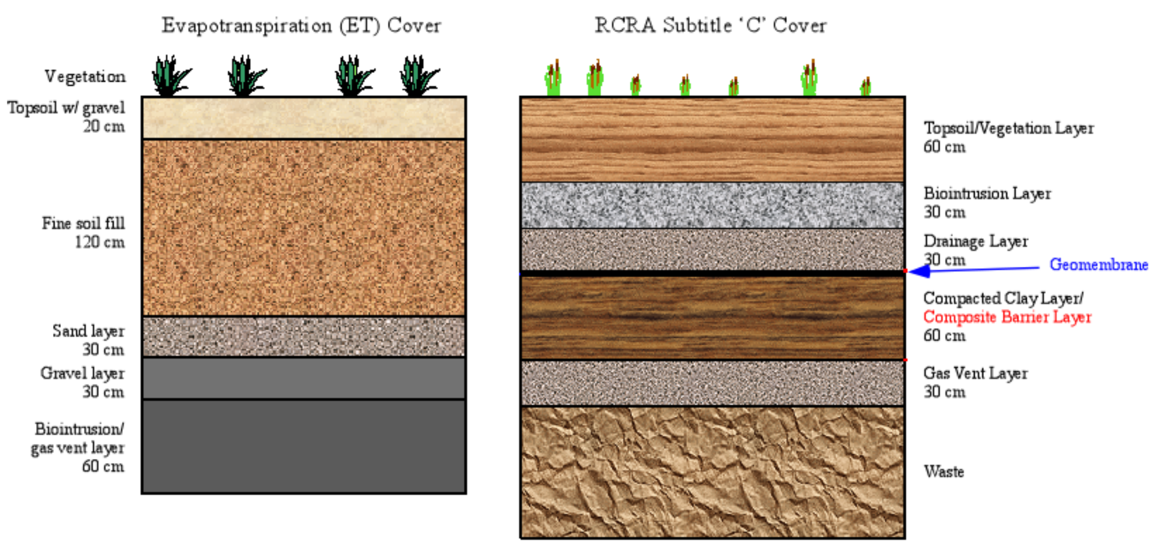

Figure 37. Cross-sectional Views of the Cover Types that can be Modeled in the CBSM (After Mattson et al. (2004))

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[113] Cap failure is modeled as a simple Poisson process with a mean time between failures based on cap type Please refer to *Advection in the Water and Atmospheric Phases including Barometric Pumping* on p. 314 and the parameters in Table 28 for more details.

The dimensions of the GoldSim *Cell Pathway* elements or "boxes" used to model the surface soil and cover layers are considered fixed over time unlike the mixing depth for the atmospheric "box" model in Figure 34. The uncertainty in the average depth of the soil and cover layers will likely be insignificant relative to other uncertainties in system, especially those in the partitioning and transport properties (e.g., diffusivities, soil-water partition coefficients, water flux, etc.). An example of how soil (including the surface and bottom soil) and cover layers are implemented in the model is provided in Figure 38.

The key properties used to define the soil and cover *Cell Pathway* elements are provided in Table 27. For the purposes of this research, these parameters are considered non-stochastic because their uncertainties are likely insignificant relative to those in the corresponding transport properties. The other key parameter needed to define each soil and cover layer is the surface area. The surface area, which is assumed to be fixed, is defined by the waste volume and thickness parameters defined in Table 25. The properties (i.e., density, porosity, moisture content, partition coefficients, etc.) of the porous media comprising the soil and cover layers are defined in Appendix C.

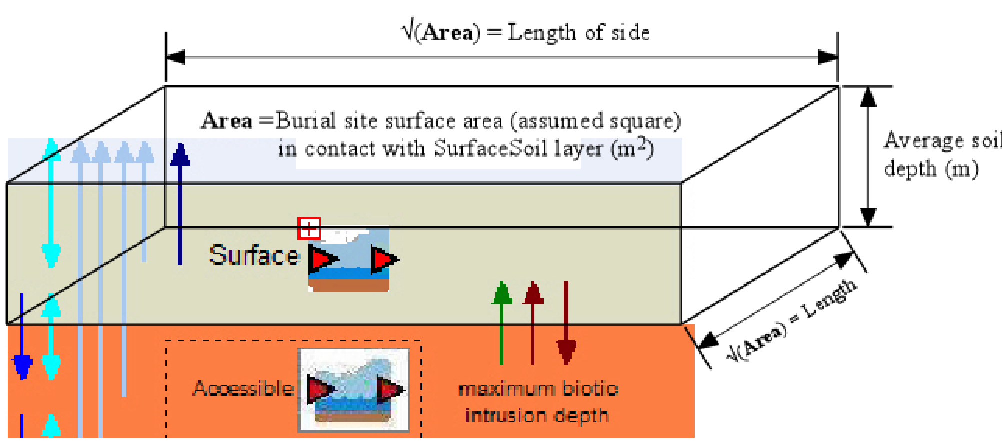

Figure 38. Soil or Cover "Box" Model as Implemented using a GoldSim *Cell Pathway*

Table 27. Key Soil and Cover Layer Properties used in the CBSM

	Layer	Parameter	Conditions	Value	Reference[a]	Comment
Baseline	Surface Soil	Average depth[b]	Arid	3 ft (0.9 m)	(Holdren et al. 2006)	Minimum depth is 3 ft for pits and 1.5 ft for trenches.
			Humid	0.5 m	(Buck et al. 1997)	The top soil extends down 0.5 m below the ground surface.
		Porous medium	Both	Organic Soil	Appendix C	The same material is used for all *soil* layers.
Evapotranspiration (ET) Cover	Top Soil[c]	Average depth[b]	Arid	1.4 m	(Mattson et al. 2004)	This combines the top two layers of the ET cover in Figure 37.
		Porous medium	Arid	Organic Soil	Appendix C	The same material is used for all *soil* layers.
	Sand	Average depth[b]	Arid	0.3 m	(Mattson et al. 2004)	This is the sand layer for the ET cover in Figure 37.
		Porous medium	Arid	Sand	Appendix C	The same material is used for all *sand* layers.
	Gravel	Average depth[b]	Arid	0.9 m	(Mattson et al. 2004)	This combines the bottom two ET layers in Figure 37.
		Porous medium	Arid	Gravel	Appendix C	The same material is used for all *gravel* layers.
RCRA Subtitle 'C' Cover	Top Soil	Average depth[b]	Both[d]	0.6 m	(Mattson et al. 2004)	This layer is the top layer of the RCRA cover in Figure 37.
		Porous medium	Both[d]	Organic Soil	Appendix C	The same material is used for all *soil* layers.
	Biointrusion	Average depth[b]	Both[d]	0.3 m	(Mattson et al. 2004)	This is biointrusion layer of the RCRA cover in Figure 37.
		Porous medium	Both[d]	Gravel	Appendix C	The same material is used for all *gravel* layers.
	Drainage	Average depth[b]	Both[d]	0.3 m	(Mattson et al. 2004)	This is drainage layer of the RCRA cover in Figure 37.
		Porous medium	Both[d]	Organic Soil	Appendix C	The same material is used for all *soil* layers.
	Clay[e]	Average depth[b]	Both[d]	0.6 m	(Mattson et al. 2004)	This is the combined geomembrane/clay layers.
		Porous medium	Both[d]	Clay	Appendix C	The same material is used for all *clay* layers.
	Gas Vent	Average depth[b]	Both[d]	0.3 m	(Mattson et al. 2004)	This is gas vent layer of the RCRA cover in Figure 37.
		Porous medium	Both[d]	Organic Soil	Appendix C	The same material is used for all *soil* layers.

a. The properties (e.g., density, porosity, moisture content, partition coefficients, etc.) needed to define material for use in GoldSim Solid element are defined in Appendix C.

b. The surface area of the soil and cover layers is assumed to be the same as that for the atmospheric "box" model in Table 25.

c. Evapotranspiration (ET) covers may be applicable in arid or semi-arid conditions.

d. The depths and materials used for the RCRA Subtitle 'C' covers are dictated by regulation and not based upon the installation location. Some media properties (e.g., moisture content, porosity, etc.) will change based on location as illustrated in Appendix C.

e. The geomembrane layer is not represented using a Cell element in the model; instead the water flux from the drainage to clay layer is assumed to be zero unless the cap has failed.

Waste Areas, Inventories, and Source Terms

DOE buried waste sites are often large and complex areas containing diverse types of wastes and contaminants buried in a variety of forms over many decades. Contaminants have often migrated into the environment surrounding the original burial sites and, therefore, remedial action is often required. The prototype sites, the Subsurface Disposal Area (SDA) and the Bear Creek Burial Grounds (BCBG), selected for this study bracket the types of wastes and conditions expected across the DOE Complex.

As illustrated in Appendix D, inventory information generated for the CERCLA remedial investigations for the SDA (Becker et al. 1998) and BCBG (SAIC 1996a) was used to generate a list of 237 isotopes and 45 nonradioactive compounds, respectively, that should be modeled using a screening risk analysis to identify contaminants of potential concern (COPC). The COPC lists generated by DOE personnel as a result of their remedial investigations are not the appropriate starting points for the analysis in this research; the original inventories must be examined based on risk to generate the appropriate COPC list. COPC lists generated from the screening risks analysis using the CBSM can then be compared to the lists generated from the site remedial investigations.

Diverse radioactive and hazardous wastes were originally buried in various areas (e.g., trenches, pits, etc.) throughout the SDA and BCBG sites. Disposal areas, denoted "source areas" in SDA remedial investigations (Holdren et al. 2006; Holdren et al. 2002), can be grouped by their similarities—the most important of which is location from a retrieval perspective. As described in Appendix D, the source areas for both sites were grouped into Waste Areas based on their potential for retrieval. The aggregation to Waste Areas simplifies modeling considerably without introducing significant inaccuracy in the resulting predictions (especially when compared to the uncertainties in the transport and other parameters used in the model).

From a contaminant release perspective—apart from the mere presence of a contaminant in the burial site—whether or not the contaminant is in a container (e.g., drum, box, etc.) and/or bound in a matrix (e.g., resin, glass, metal, etc.) are the most important characteristics of buried wastes. Three mass release mechanisms are modeled based on the types of waste forms in the SDA and BCBG (Anderson and Becker 2006):

- *Surface wash*—an equilibrium partitioning model for wastes with surface contamination that could be removed via washing,
- *Dissolution*—a model in which the waste matrix undergoes dissolution for those contaminants encapsulated in a matrix not allowing diffusion, and
- *Diffusion*—a model for those contaminants in waste forms (e.g., sludges) that can diffuse to the surface.

Different waste forms may undergo different release mechanisms, which must be implemented in the screening risk tool to provide an appropriate source term for exposure modeling. The implementations of the release mechanisms are discussed in Appendix E.

To represent the important waste and contaminant characteristics for source-term modeling, contaminants of interest (i.e., radioactive and hazardous contaminants and their radioactive and hazardous progeny) were partitioned according to source area (i.e., where the waste was buried), containment (i.e., whether loose or buried in a drum or box), and waste form (i.e., in glass, resin, metal, etc.). Appendix D provides the inventory partitioning of the contaminants of potential interest for modeling purposes for the SDA and BCBG maximum retrieval cases[114].

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[114] The corresponding information for the targeted retrieval cases is found in the GoldSim model in the \BurialSiteSetting\WasteAreas\ container (in Figure 31) by Waste Area, container, and conditions.

Two primary waste areas are defined for the burial site: one containing the wastes likely to be managed in-place and the other containing those wastes likely to be targeted for retrieval[115]. Contaminant inventories are allocated by waste form among *Source* elements representing possible containerization (i.e., drum, box, or loose). The surface wash, dissolution, and diffusion release models are then used to control contaminant fluxes from *Source* to *Cell Pathway* elements representing the contaminated waste soil. Each *Source* element contains two *Cell Pathway* elements representing layers within the Waste Area, the upper one potentially accessible to biota and the lower one inaccessible—it is into these cells that contaminants are released. The structure of the source term for release modeling is described in Figure 39.

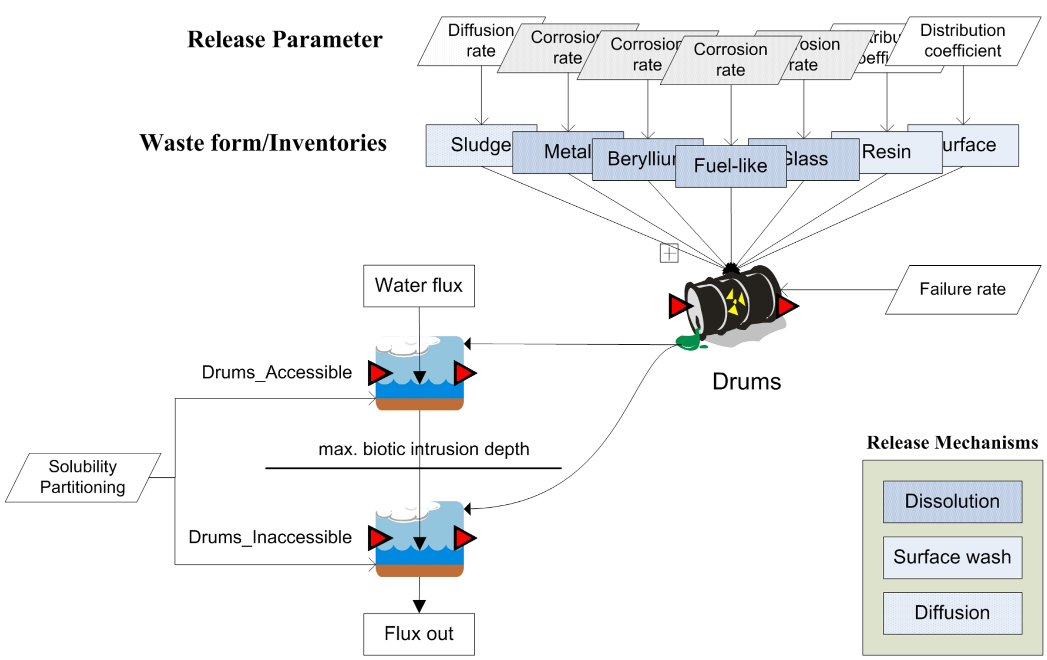

Figure 39. Conceptual Model, Release Mechanisms, and Parameters for Contaminant Releases from Different Buried Waste Forms

[115] An additional Waste Area was defined for disposal in humid conditions where excavated wastes cannot be returned to the original site because it may be impacted by inundation or shallow groundwater flow.

Vadose and Interbed Zones

The SDA resides in an arid to semiarid climate above a deep, primarily fractured basalt vadose zone below which a sole source aquifer flows that is used by many residents of Southwest Idaho. The vadose zone underlying the SDA is layered with thin sedimentary interbeds. The interbeds tend to retard the vertical movement of water and thus may play an important role in contaminant transport to the aquifer, which is the primary concern for most long-term contaminant risks to the general public. On the other hand, large portions of the BCBG reside much of the time in groundwater and almost all impacts will be via the surface water pathway. The subsurface underlying the BCBG is karst (resulting in preferential flow[116]); however, this feature will not likely significantly impact results because the contaminants and hazards with the most prominent and immediate impacts are transmitted via shallow and surface water flows. The vadose zone is designed to capture the relevant features of the SDA vadose zone because the features of the BCBG vadose zone do not significantly impact contaminant transport and risk.

The vadose zones for the SDA and BCBG are heterogeneous and complex subsurface regions comprised of multiple layers with multiple, complex interactions with the saturated zone and surface waters. The vadose zone underlying the SDA is comprised of as many as ten fractured basalt flow groups and seven major sedimentary interbeds, primarily unconsolidated sediment and volcanic materials (Anderson and Lewis 1989; Holdren et al. 2006). Representing all layers in a transport model is unlikely to be worth the effort because the benefit of resulting model realism would be more than offset by uncertainties in the parameters needed to describe the layers and their properties.

[116] Therefore, the areas underlying both the SDA and BCBG experience preferential flows. Other sites (e.g., Savannah River Site) do not have vadose zones with preferential flow regimes. Such zones are much easier to model than fractured and preferential zones and can easily be substituted for those in model.

A simplified representation of the subsurface region for a prototype site is warranted. For example, a simplified representation including four basalt flow regions and three primary sedimentary interbeds were used in the most recent SDA remedial investigation (Holdren et al. 2006). Contaminant fate and transport in the region underlying the SDA were modeled using a version of the three-dimensional TETRAD multiphase, multicomponent simulator (Vinsome and Shook 1993) specially modified for the SDA use (Holdren et al. 2006; Magnuson and Sondrup 2006). The TETRAD code can model contaminant movement in a coupled fracture-matrix system using dual-permeability capabilities; however, this capability was employed only for VOCs and C-14 transport predictions (Holdren et al. 2006; Magnuson and Sondrup 2006).

In general, flow though fractured basalt was considered during the Idaho Site remedial investigation to be adequately described by an equivalent low-porosity, high-permeability porous medium. Interbeds were characterized by a low-porosity, low-permeability surface feature, either resulting from a sediment layer or infilling of the fracture network (Magnuson and Sondrup 2006). The movement of water and dissolved-phase contaminants can be significantly retarded by these features, which can also result in ponding of water above the interbed surface. However, a sophisticated three-dimensional model like TETRAD, which requires extensive and site-specific data often unavailable at the formative stages of the remedial investigation process, does not appear appropriate for a screening risk analysis and is be used here[117].

For the BCBG remedial investigation, surface water pathways rightly take center stage when considering the major sources of potential exposure risk although there is

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[117] In fact the initial screening risk analysis for the SDA (Becker et al. 1998) was performed using GWSCREEN (Rood 1994). Transport in the unsaturated zone is described by a simple plug flow model and that in the saturated zone using a semi-analytical solution to the advection dispersion equation.

significant DNAPL contamination in the vadose and saturated zones beneath the burial grounds (SAIC 1996a; b; e). The SESOIL, ODAST, and CRAFLUSH codes were used to model contaminant transport from the buried wastes through the subsurface (SAIC 1996d). In general, SESOIL was used to model organic contaminants[118] and ODAST for inorganic compounds and radionuclides[119]. CRAFLUSH was used to model the lateral movement of contaminated groundwater to Bear Creek[120]. However, only the SESOIL model is currently available and in use (as part of the SEVIEW suite of modeling codes). Because the features used in the three aforementioned models are available in GoldSim with added probabilistic features, the GoldSim program is used directly for the BCBG without calling any external programs for vadose modeling.

Although the models used in the initial remedial investigation screening risk assessments will not be used for this research, the concepts used in these investigations including one-dimensional transport and minimum site-specific information are employed with stochastic techniques to represent uncertainty. To begin modeling, a simplified representation of the subsurface region will be required. Depictions of the SDA and BCBG subsurface regions reasonable for a screening-level analysis are those used to develop the WIPP Disposal-Phase Environmental Impact Statement (EIS) (Buck et al. 1997; USDOE 1997); these descriptions of the vadose and saturated zones as illustrated in Figure 40 are adopted for this research.

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[118] SEasonal SOIL (SESOIL) is a one-dimensional compartment model used to simulate contaminant movement through the vadose zone (SAIC 1996d).

[119] One-Dimensional Analytical Solute Transport (ODAST) is an analytical solution that computes the normalized concentrations of a given constituent in a uniform flow field from a source having a constant or varying concentration in the initial layer (Javandel et al. 1984; SAIC 1996d).

[120] CRAFLUSH is an analytical transport model that can be used to simulate the flushing of a contaminated matrix with a system of parallel fractures (SAIC 1996d).

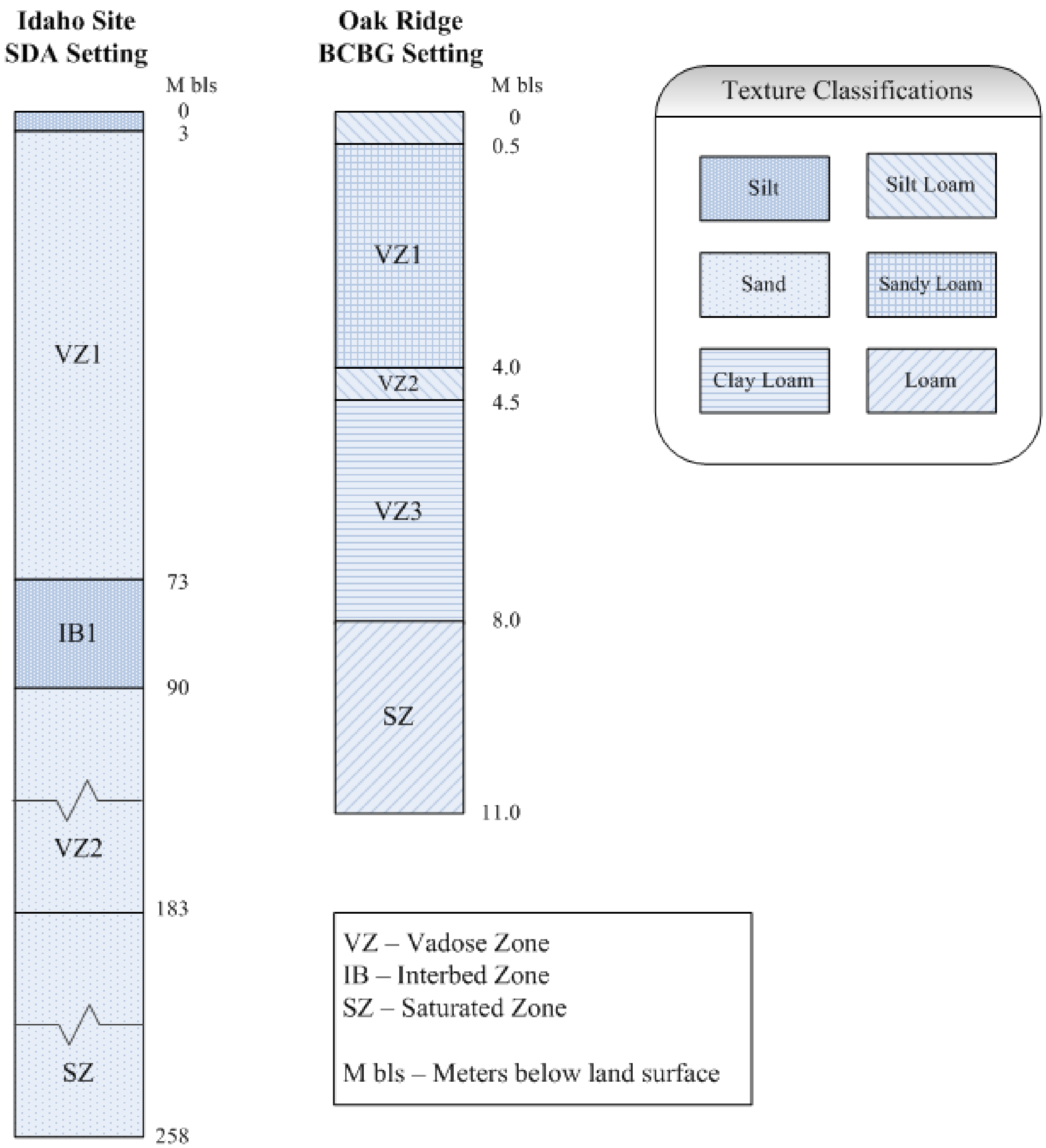

Figure 40. Stratigraphic Representations (Not to Scale) for the Subsurface Disposal Area (SDA) and Bear Creek Burial Grounds (BCBG) (Buck et al. 1997)

For a one-dimensional modeling approach, the vertical dimensions and types of material comprising the vadose and saturated zones are crucial to estimating the transport of contaminants through the subsurface. Based on this conceptualization, both sites are modeled using two regions separated by a middle region. In the SDA, the middle layer is a sedimentary interbed, which can significantly retard the downward movement of water and contaminants through the vadose zone to the aquifer. Because the vadose zone beneath the SDA is so extensive and the saturated zone beneath is a sole-source aquifer, the extent and timing of contamination in the aquifer is critical to predicting risks to the general public. An accurate representation of the vadose and saturated zones beneath the BCBG is not important because most of the water that passes through the burial site enters the surface water, where exposure risks are most prevalent.

The description of the subsurface region in the GoldSim screening risk model is illustrated in Figure 41. The surface layer texture is used to describe the top soil layer above the buried waste areas with a depth of topsoil that may differ (e.g., the minimum soil cover for SDA pits was 1 m and for trenches was 0.5 m (Holdren et al. 2006)). For convenience, the downward contaminant fluxes from all wastes areas will enter a single, thin (≤ 1 m) bottom soil compartment, which is present in the SDA pits and trenches[121], and then to the vadose zone. Although wastes lie below ground level, the dimensions of the waste layers are assumed to be the same as those in Figure 40[122]. For the SDA, two vadose zone layers are divided by one sedimentary interbed layer with dimensions given in Figure 40. For the BCBG, the middle layer represents the thin silt loam layer shown in Figure 40. Unlike those used to develop the WIPP EIS (Buck et al. 1997), the vadose layers are modeled as fractured zones using the GoldSim *Network Pathway* element.

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[121] Because it is unknown whether or not a soil layer was added to the BCBG areas before the wastes were buried, the bottom soil properties will be the same as those for the BCBG waste layer.

[122] This assumption should not significantly impact the SDA transport model because of the great depth of the vadose zone. Because the majority of the water passing through the waste flows to the saturated zone and surface water, this assumption will not significantly impact exposure predictions.

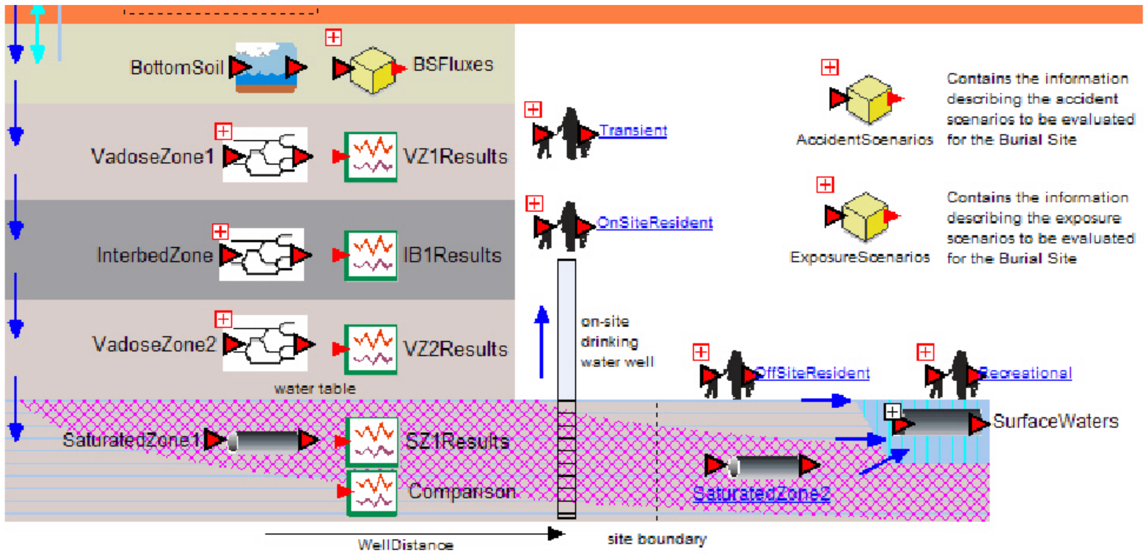

Figure 41. GoldSim Representation of the Vadose and Saturated Zones and Surface Water of the Conceptual Site Burial Model

The fracture characteristics for the SDA and BCBG vadose zones are not well-known because such regions are notoriously difficult to characterize. However, because of the potential large impact of fractured or preferential flow on transport and thus exposure and risk modeling, this capability was added to the model even though the model is intended for screening-level analysis. Fractured flow is modeled in GoldSim using the *Network Pathway* element, which is an efficient way to represent a very large and complex network of pipes that can be used to simulate flow through a fractured medium (GTG 2005a). The Network Pathway element requires specification of a fracture network, which identifies the pipes in the network, how the pipes are connected, and the dimensions and flow rates of the pipes. An example of a fracture network layout is illustrated in Figure 42 (GTG 2005a).

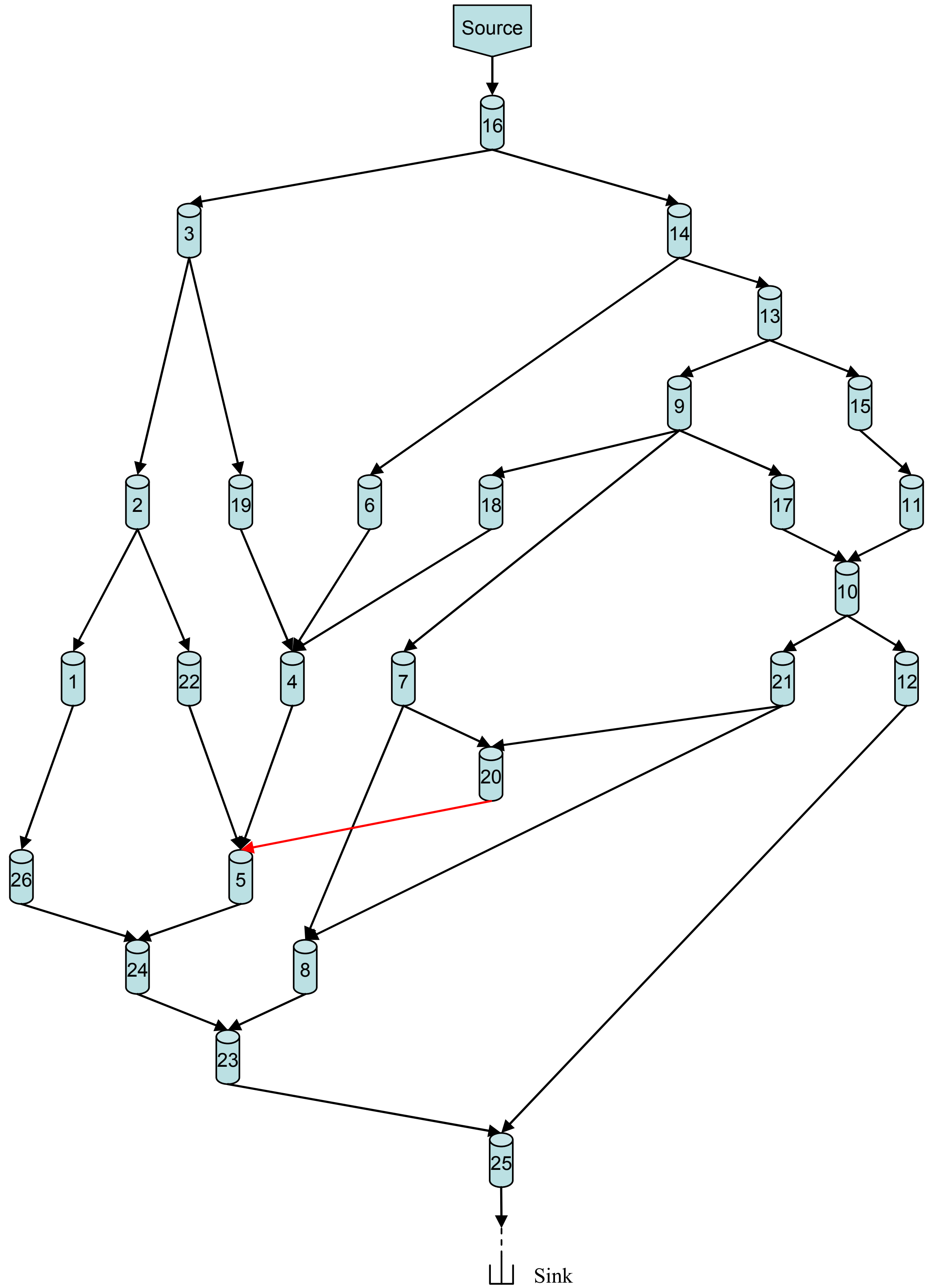

Figure 42. Example GoldSim *Fracture Network* (GTG 2005a) Represented as a Series of 26 Pipes that also Serves as the Basis for Fractured Flow Modeling in this Research. The red arrow indicates that this line crosses over the other.

Because insufficient information exists to represent the site-specific fractured media and flows for the SDA and BCBG[123], the decision was made to modify the GoldSim example network (illustrated in Figure 42) to represent what is known about the subsurface regions to be modeled. The modification represents a calibration process to represent the general characteristics of flow through the fractured zones based on site-specific information without attempting to capture every conceivable nuance in the fractured flow. This procedure will be concentrated on the SDA because of the extent of the vadose zone and because BCBG fractured flow waste will not be critical to exposure and risk modeling because the primary pathway for both water and contaminates will be short-circuited to the saturated zone and surface water pathways.

For most contaminants, fractured flow in the vadose zone underlying the SDA is modeled in the SDA remedial investigation as moving through a non-absorbing, anisotropic porous medium with a low effective porosity and high permeability (Magnuson and Sondrup 2006)[124]. To define an equivalent GoldSim *Network Pathway* element, the properties (e.g., length, flow rate, etc.) of the 26 pipes comprising the pathway were systematically varied until the output characteristics of the two were similar. The equivalent *Network Pathway* elements and the manner in which the properties of these elements are defined for the SDA and BCBG vadose zone regions are described in Appendix F.

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[123] The fracture information used in GoldSim would typically be generated using a discrete fracture generation and flow simulation code (GTG 2005a), which is not available. The solution of providing the fractured flow mechanism using a known fracture network calibrated to the subsurface flow conditions was decided to be the best approach for this screening risk model.

[124] Transport for VOCs and C-14 was modeled as moving through a coupled fracture-matrix system using dual-permeability capabilities (Holdren et al. 2006; Magnuson and Sondrup 2006).

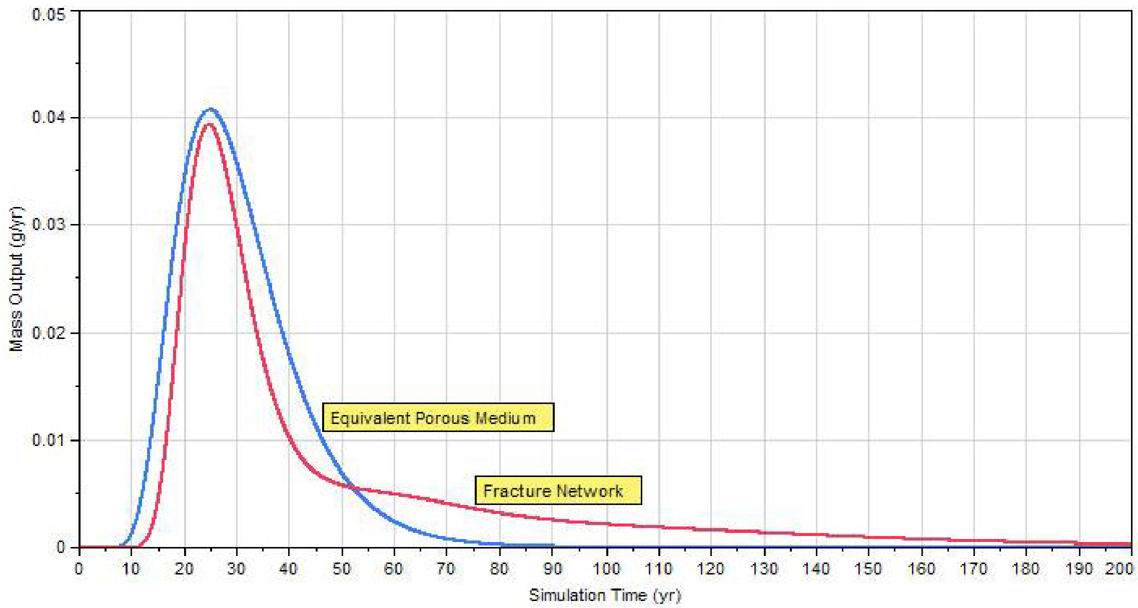

Figure 43. Relationship between the Fracture Flow Representation and Equivalent Porous Medium for SDA Vadose Zone 1 for a Unit Mass Input of an Unretarded, Conservative Tracer (73 m, expected conditions)

Saturated Zones

After migrating through the vadose zone, or in the case of the BCBG perhaps directly from the waste area, contaminants move with the water to the saturated zone. The saturated zones for the SDA and BCBG are described in Figure 40. As illustrated in Figure 44, two saturated zones (namely SaturatedZone1 and SaturatedZone2) were modeled to address potential receptors—both on-site and off-site—that might be impacted by contaminated groundwater. For both the SDA and BCBG sites, on-site use of groundwater is restricted and the restriction continues until the Institutional Control (IC) period has passed and the general public has access to the land above the waste area and can drill a drinking water well[125].

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[125] For the purpose of this study, the well is not drilled through the waste area because a surface barrier will be in-place providing much easier drilling in another location.

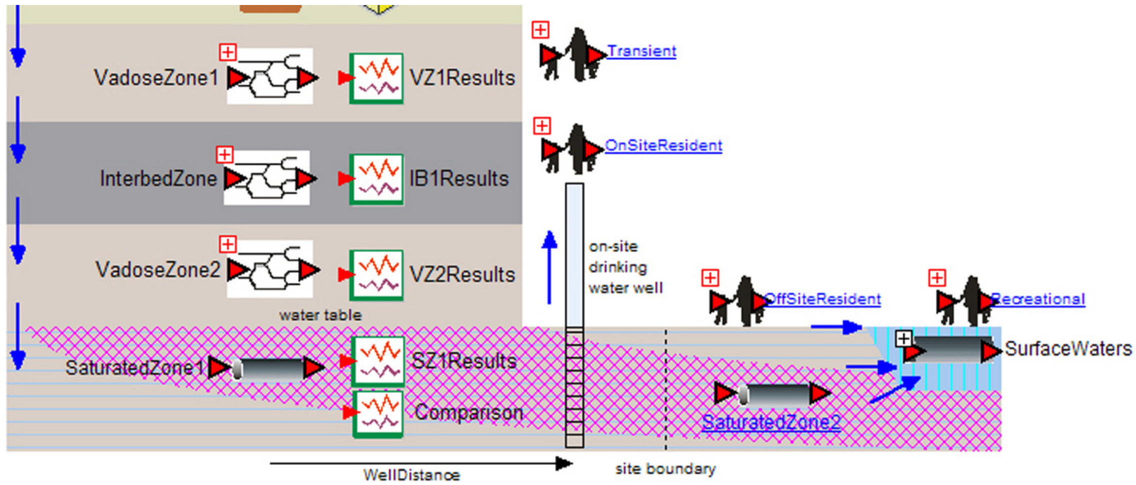

Figure 44. The Vadose and Saturated Zones and Receptors as Implemented in the GoldSim Model

The impact of contaminated groundwater to off-site receptors is via the transport of contaminants to surface water, which is used for recreation, fishing, etc. Although not considered a scenario for this research, the drinking water well can be "moved" to the site boundary to estimate the potential exposure and impact to the off-site general public. Instead the maximum impact to potential receptors is considered by "turning off" ICs and estimating risks to on-site receptors from contaminated groundwater.

Pipe Pathway elements are used to model the saturated zones as illustrated in Figure 45. The first zone, SaturatedZone1 (SZ1), models drinking water contaminant concentrations to future on-site receptors. Effects of contaminants entering SZ1 are assumed local and the source zone for the impacted region has the same dimensions as the vadose zone above. The average linear horizontal velocity for the aquifer, assumed to be fractured basalt with an effective porosity of 0.06 and dispersivity of 9 m, underlying the SDA ranges between 1 and 24 m/yr with an area-weighted average value of 9.3 m/yr (Holdren et al. 2006)[126].

Two contaminant concentrations can be predicted for water leaving SZ1 (and entering SaturatedZone2). The normal concentration provided by the *Pipe Pathway* element is the average concentration exiting the element and disregards any spatial variation in concentration (perpendicular to the flow direction). Alternatively, the GoldSim *plume* function can be used to correct the spatially-averaged concentration returned from the *Pipe Pathway* element based on the location of the well relative to the flow path (GTG 2005a). The corrected value will typically be lower so using the normal average concentration will maximize exposure concentration and exposure risk.

The second saturated zone is primarily used to estimate surface water effects for the contaminants released from the Oak Ridge BCBG. There are no permanent surface water features near the Idaho Site SDA (Holdren et al. 2006); the aquifer is instead bounded by the Snake River and the Yellowstone basin, which are both far (i.e., more than 10 km) from the SDA—and too far to be reasonably modeled. The BCBG is between approximately 180 m and 500 m from the Bear Creek (SAIC 1996a), which is the nearest surface water to the BCBG and the primary pathway for contaminant migration from the burial site to off-site receptors. The majority (approximately 95%) of groundwater flow passing through the BCBG is assumed to enter the surface water pathway at the Bear Creek (SAIC 1996a)

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[126] The flow in the aquifer under the SDA is primarily horizontal (i.e., 300 times that of the vertical flows) (Holdren et al. 2006). The focus will be on the saturated zone parameters for the SDA because the BCBG saturated zones are not a major pathway for contaminant migration. The corresponding parameter values for the BCBG saturated zones are found in the GoldSim model.

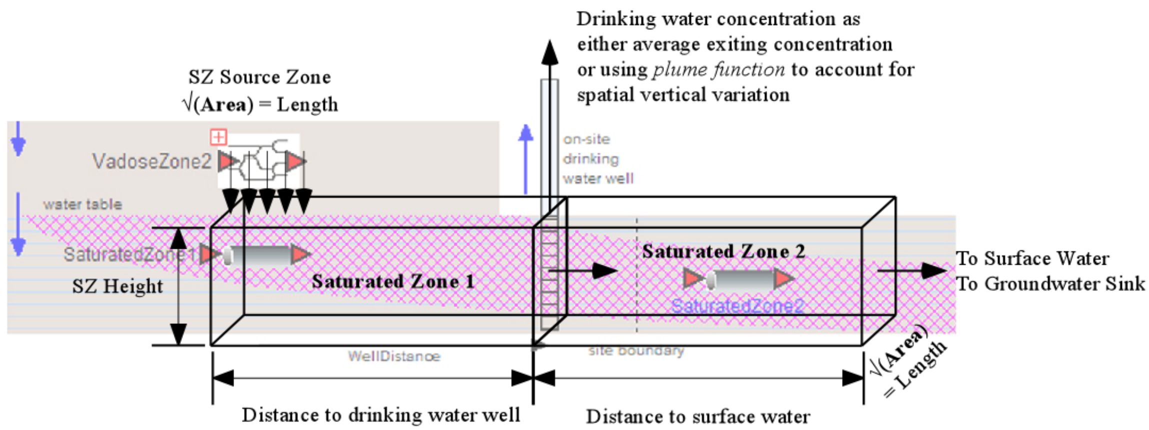

Figure 45. GoldSim Implementation of the Saturated Zones using Pipe Elements

Surface Water

Water-borne hazardous and radioactive contaminants from the BCBG enter the surface water pathway via Bear Creek, which is entirely contained inside the boundaries of the Oak Ridge Reservation. A convenient starting point for the BCBG surface water pathway analysis is Bear Creek Marker (BCM) 9.47[127] where more than 99% of available water passes either as surface or groundwater (SAIC 1996a).

As illustrated in Figure 46, Bear Creek enters the East Fork of Poplar Creek (at BCM 0.0), and East Fork feeds Poplar Creek approximately 2 km further downstream. Poplar Creek is a tributary of the Clinch River entering the river near the East Tennessee Technology Park. Thus the nearest *off-site* surface water receptor location would be the Clinch River near the East Tennessee Technology Park.

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[127] The marker number represents the distance from the specified location to the point (BCM 0.0) at which the Bear Creek flows into the East Fork of Poplar Creek.

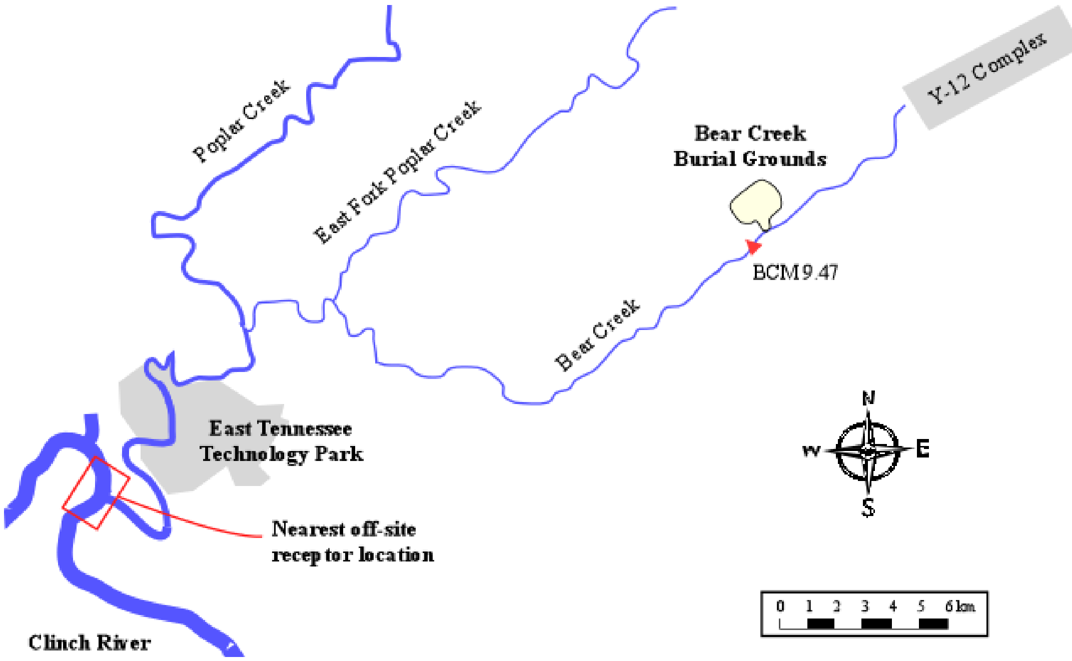

Figure 46. Nearest Off-Site Receptor Location for the Surface Water Pathway for Water-borne Contaminants from the Bear Creek Burial Grounds (BCBG) (Adapted from Figure 1 and Figure 10 in ATSDR (2006))

The surface water pathway for contaminant migration from the BCBG to the Clinch River can be implemented in many ways; two possible GoldSim implementations are illustrated in Figure 47[128]. The first implementation (i.e., the top model in Figure 47) employs one *Pipe Pathway* element for the combined Bear Creek and Lower East Fork Poplar Creek and another for the Poplar Creek. However, the *Pipe Pathway* element can return inaccurate results if properties vary temporally or if the transit time through the pipe is significantly less than one timestep (GTG 2005a). Either or both of these conditions are true for the BCBG, and the *Pipe*-based model was found to be inadequate for the conditions pertaining to BCBG contaminant migration via the surface pathway.

[128] As shown in Figure 47, the Clinch River is modeled as a single reach fed from Poplar Creek. The length of the reach is assumed to be 305 m (1,000 ft) from Figure 2 in ATSDR (2006). The average cross-sectional area of 632 m^2 (6,800 ft^2) was taken from an example problem in Jacobs (1968). These dimensions provide a total reach volume of 1.93×10^5 m^3 (6.8×10^6 ft^3).

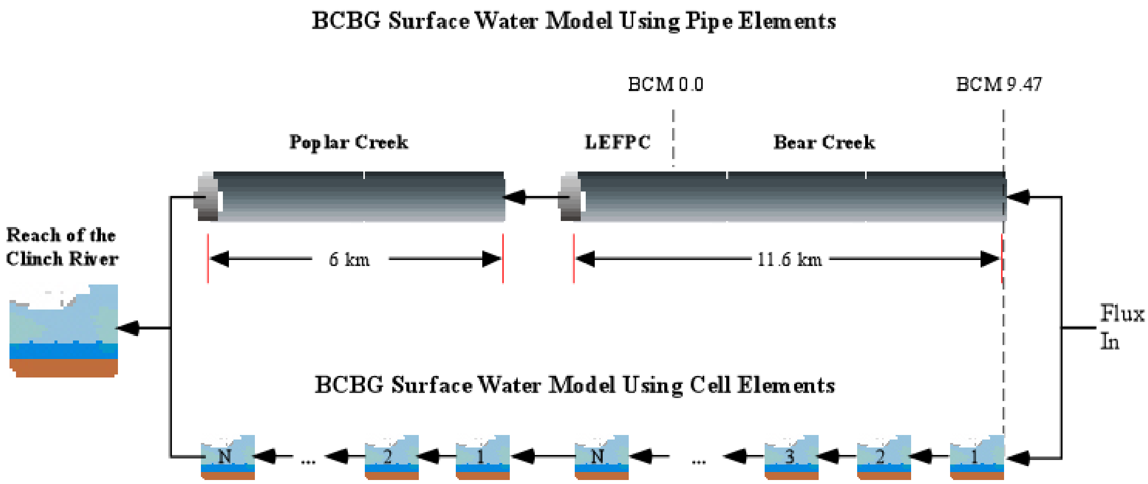

Figure 47. Possible Implementations of the BCBG Surface Water Conceptual Model Using Either GoldSim Pipe (Top Model) or Cell (Bottom Model) Elements (LEFPC = Lower East Fork Poplar Creek)

When GoldSim *Pipe Pathway* elements provide inaccurate results, an alternative modeling technique is to represent the pathway using a series of linked *Cell Pathway* elements (GTG 2005a). Although using a series of mixing cells to represent a creek or stream may appear to be overly simplistic, the dispersivity that is, for example, applied in the *Pipe Pathway* element essentially accounts for flow heterogeneities in the flow field and is not a fundamental parameter of the medium in the pipe (GTG 2005a). A series of mixing cells merely provides an alternative that can provide the desired contaminant dispersion. For example, it can be shown that the number, *N*, of mixing cells needed to represent a desired degree of dispersion (described by the longitudinal dispersivity, α_L, in m) for a distance *L* (in m) is given by the approximate relationship[129]:

$$N \cong \frac{L}{2\alpha_L}. \qquad [9]$$

[129] A detailed explanation of how this approximate relationship was obtained is provided on pp. 114-115 (Chapter 4) of the GoldSim Contaminant Transport Module User's Guide (GTG 2005a).

Typically, both mechanical dispersion (represented by α_L) and molecular diffusion play roles in the spreading of contaminants. However, for the conditions expected in the surface waters near the BCBG, mechanical dispersion tends to be the dominant process. Under these conditions, the longitudinal dispersion coefficient, k_x (m^2/s), is approximately equal to ($U\alpha_L$) where the average velocity, U, is in m/s. Thus, Equation 9 for the number of mixing vessels can be expressed as:

$$N \cong \frac{L}{2\alpha_L} \cong \frac{L}{2\left(\frac{k_x}{U}\right)} \cong \frac{LU}{2k_x}. \qquad [10]$$

For a river, the longitudinal dispersion coefficient can be approximated from the river flow velocity, U (m/s), river width, B (m), and river depth, d (m) (NCRP 1996a):

$$k_x \cong \frac{UB^2}{3d} \qquad [11]$$

where the river width and depth can be expressed as power-law functions of the average annual flow rate, Q (m^3/s) (NCRP 1996a):

$$\begin{aligned} B &= aQ^b \\ d &= eQ^f \end{aligned} \qquad [12]$$

and a, b, e, and f are constant values estimated to be 10, 0.46, 0.19, and 0.4, respectively, from graphs provided in NCRP (1996a). The river velocity can be obtained from:

$$U = \frac{Q}{Bd}. \qquad [13]$$

Thus given a flow rate, width, depth, and velocity, the dispersion coefficient can be estimated using Equations 10 through 13. For a given river length, L (m), the number, N

of identical mixing cells needed to represent the desired dispersion is estimated using Equation 9. For example, the average annual flow rates for Bear Creek, East Fork Poplar Creek, and the Clinch River are 0.11, 1.46, and 132 m^3/s, respectively (USDOE 2000). For the flow rates and lengths (from Figure 47), the numbers of Cell elements required for Bear Creek and Poplar Creek (using the East Fork rate) are 104 and 14, respectively. The Clinch River is modeled as a single reach fed from the Poplar Creek.

However, the number of mixing cells needed to model the surface water pathway is only applicable to the flowrate, Q, for which it was obtained. If the flowrate changes, the dispersion will also change and a different number of vessels would be needed. In the GoldSim implementation, the number of vessels is fixed. The impact of this change is predictable and tolerable and certainly more desirable than the potential large mass losses incurred when using *Pipe Pathway* elements to model the surface water pathway.

The uncertainty introduced by approximating the desired dispersion using a series of mixing cells is a reasonable modeling technique for a screening-level analysis given the limitations in the *Pipe Pathway* element. However, the uncertainty introduced by the use of a series of mixing cells is not the only one impacting the surface water pathway. Two assumptions are made that produce exposure and risk results that are intentionally biased high. The first assumption is that no contaminants settle out during the trip from the BCBG to the Clinch River. The second assumption is that any volatile contaminants in the surface water remain all the way to the Clinch River.

Transport Pathways

As shown in Table 24, various transport pathways must be modeled to represent migration of contaminants from the burial site through exposure media to potential receptors. The important contaminant transport pathways that must be modeled include advection, diffusion, and animal- and plant-induced transport. For the atmospheric pathway, advection and dispersion of contaminants and barometric pumping are important transport mechanisms. For water-borne contaminants, flooding and inundation may play an important part in migration; whereas, colloidal transport may be significant for high-profile radionuclides (e.g., plutonium isotopes). Contaminants may be transported via the bulk movement of soil either via the atmospheric pathway (i.e., resuspension) or the surface water pathway (i.e., runoff). These important transport pathways and how they are implemented in GoldSim are now described.

Contaminant Partitioning and Solubility Constraints

When entering a GoldSim element, contaminants are instantaneously and completely mixed throughout the various solid and fluid media in the element. Media are assumed to be at equilibrium and contaminant partitioning is based on partition coefficients and the masses of the media present. Partition coefficients for the various media available in the model are described in Appendix C.

Solubility constraints for contaminants in the fluid media (i.e., water) can only be imposed in GoldSim *Cell Pathway* elements. Solubility limits are entered either as constants or as results of GoldSim *Stochastic* elements. The dissolved concentration for the contaminant in the medium cannot exceed the solubility limit. If isotopes of the same element are included, the solubility limit is "shared" among all such isotopes. Therefore, because many isotopes will be simulated in the model, solubility limits are initially defined in terms of mol/m^3 and converted to the appropriate mass/volume basis in the

GoldSim model as illustrated in Appendix C (GTG 2005a). The user has the ability to run the simulation without imposing solubility constraints on the contaminants in the system.

Intermedia Diffusion via the Vapor Phase

Once entering a pathway, contaminants may migrate via numerous mechanisms, the primary of which are advection and diffusion. Diffusion of contaminants through a fluid medium (e.g., pore vapor, atmosphere, etc.) as well as inter-media diffusion across the interface between adjacent fluid media (e.g., air-water) can be directly simulated in GoldSim (GTG 2005a). The diffusion coefficients for fluid media through which a contaminant may diffuse are defined in the screening risk tool. The primary pathway for diffusion is via air or vapor phase; these diffusion coefficients are defined in Appendix C.

To define the diffusive mass link, the two media are specified and the properties of the diffusive link are defined including the geometry of the link and whether either or both media are porous. The geometry of the diffusive link is described by the characteristic diffusion length, λ, for both media and the area over which diffusion occurs. The manner in which these parameters are defined for the porous media in the model is best described graphically as illustrated in Figure 48.

The diffusion length is the distance from the center of the medium to the interface. The distance, however, is not the straight-line distance because tortuosity must be taken into account as shown. The diffusive area is just that area containing the fluid—in this case, air. Unlike other transport processes that can be modeled in GoldSim, diffusion is a process driven by the concentration gradient, and thus contaminants can be transported in either direction.

The conceptual model for diffusion illustrated in Figure 48 pertains to solid, porous media in Table 24. These media include the air-borne diffusion processes between the surface soil and cap layers, waste layers, and bottom soil layer. As also illustrated in Table 24, the model represented in Figure 48 omits the diffusion of contaminants between the surface soil (or the top cap layer if a barrier is installed) and the atmosphere. The diffusion of contaminants from the surface layer to the atmosphere is simulated by defining the thickness (stochastically) of the stagnant boundary layer of air through which diffusion occurs and above which the contaminant concentration is zero (Jury et al. 1990; Jury et al. 1983). The distribution of possible boundary layer thicknesses is assumed uniform with values from 0.001 to 1 m (Ho et al. 2005; Jury et al. 1983) where the upper 95% value (of 0.2 m) is used for the deterministic case[130].

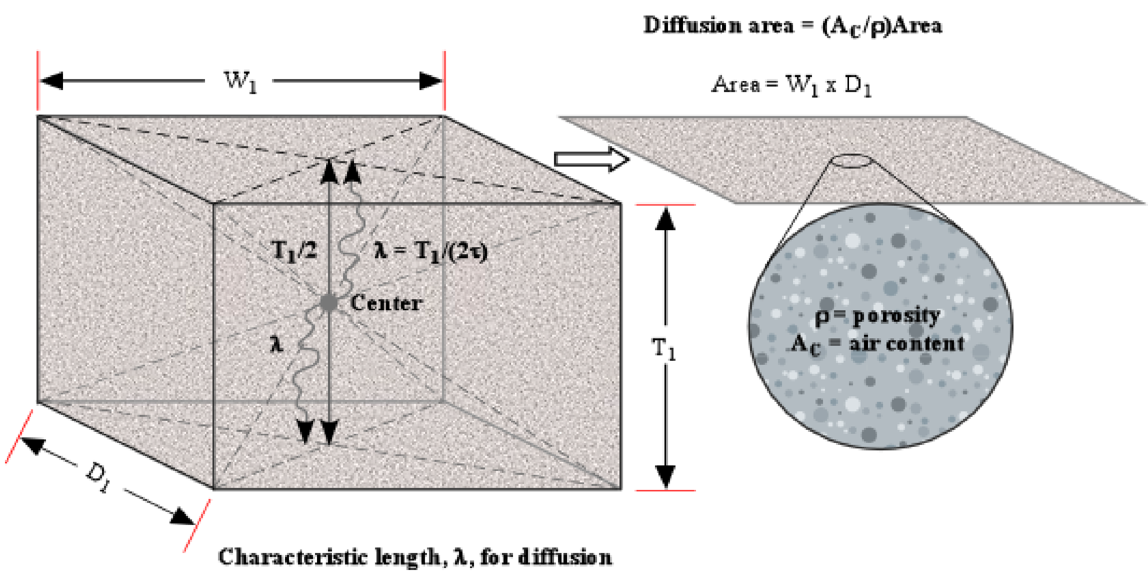

Figure 48. Defining the characteristic length and area for contaminant diffusion through a porous media.

[130] Tauxe (2004) uses a fixed value of 0.1 m in his generic performance assessment model for the boundary layer based on an unspecified reference.

Diffusion via the Water Phase and across Fluid Phase Boundaries

In the model, infiltrating water enters the top surface layer (either the surface soil or the top cap layer if a barrier is in-place) and then percolates through the Waste Areas, vadose zone, and finally down to the saturated zone. For the BCBG, infiltrating water may divert to the surface water or groundwater pathways from the surface or waste layers. Contaminant transport via the water pathway is assumed to be dominated by advection, and thus contaminant diffusion through the water phase is neglected (Tauxe 2005). When available, diffusion coefficients for contaminants in water have been included so that waterborne diffusion can be added either for completeness or if the assumption of advection-dominated transport through the water phase is discovered to be untenable.

Diffusion of contaminants across fluid interfaces (e.g., air-water) is not simulated in the screening risk model. Instead, contaminants are instantaneously and completely mixed throughout the various solid and fluid media present in an element. Media are assumed to be at equilibrium and contaminant partitioning is based on partition coefficients and the masses of the media present. Diffusion of contaminants from, for example, the pore water in the surface soil to the atmosphere or from the surface water elements to the atmosphere are neglected. In the former case, waterborne transport is assumed to be advection-dominated. In the latter case, ignoring the transport of contaminants from the surface water to the atmosphere intentionally biases high the exposures and risks related to the surface water pathway.

Advection in the Water and Atmospheric Phases including Barometric Pumping

In the GoldSim model, fluid and soil advection processes (e.g., erosion and entrainment of soil in runoff) are simulated. To define an advective flux, the flow rate of the medium must be specified; the contaminant flux is the contaminant concentration in the medium times the flux. Typically, the fluid will be either air or water; however, advective fluxes can also be used to transport contaminated soil via resuspension in air or water. Colloidal transport (discussed below) is modeled using advective mass fluxes where the contaminant concentration is that sorbed onto the solid being transported.

The primary pathway for both the release of contaminants and their transport to potential receptors is the water percolating through the waste areas. The resulting contaminated water will eventually migrate to groundwater (where it might be accessed via potable water wells and ingested or used for washing) or to surface water where it might contaminate fish or other aquatic creatures (that, again, might be caught and ingested) or directly impact recreational users of the surface water. The conceptual advective water flow model is described in Figure 49.

Three sources of water (i.e., percolation, flooding, and inundation) can facilitate the release and transport of contaminants from the buried waste sites. A fourth flow shown on Figure 49, overland flow, is assumed to transport *suspended contaminated soil* to the surface water. Whereas, inundation flow through the buried waste site is only a function of subsurface flow and transport properties, percolation and flooding influxes are functions of whether or not conditions are right for flooding, the site has been capped, and the state of the cap as shown in Figure 50.

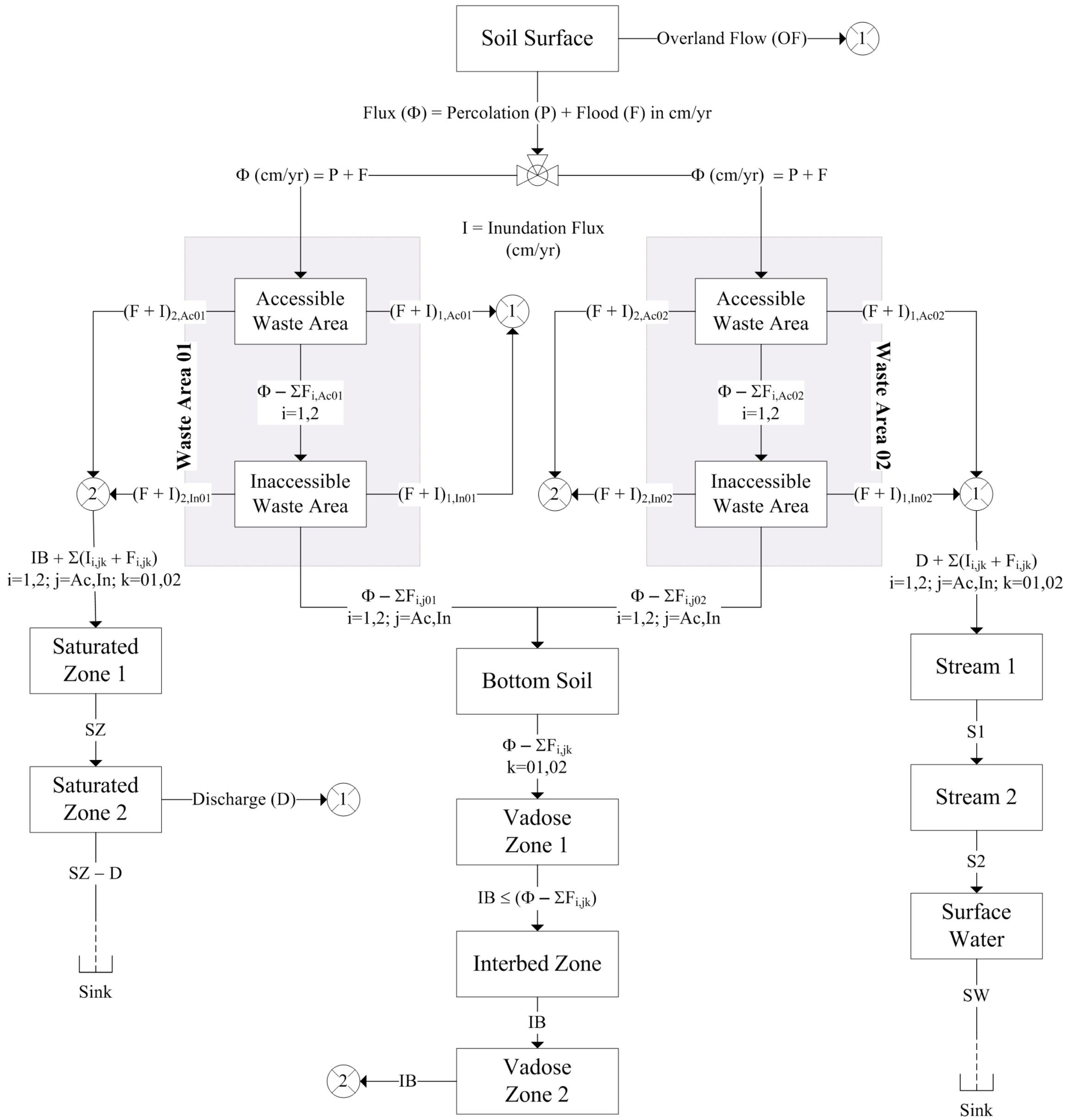

Figure 49. Advective Flow Model for the Conceptual Burial Site Model describing the Idaho Site SDA and Oak Ridge BCBG (Assuming Two Waste Areas)

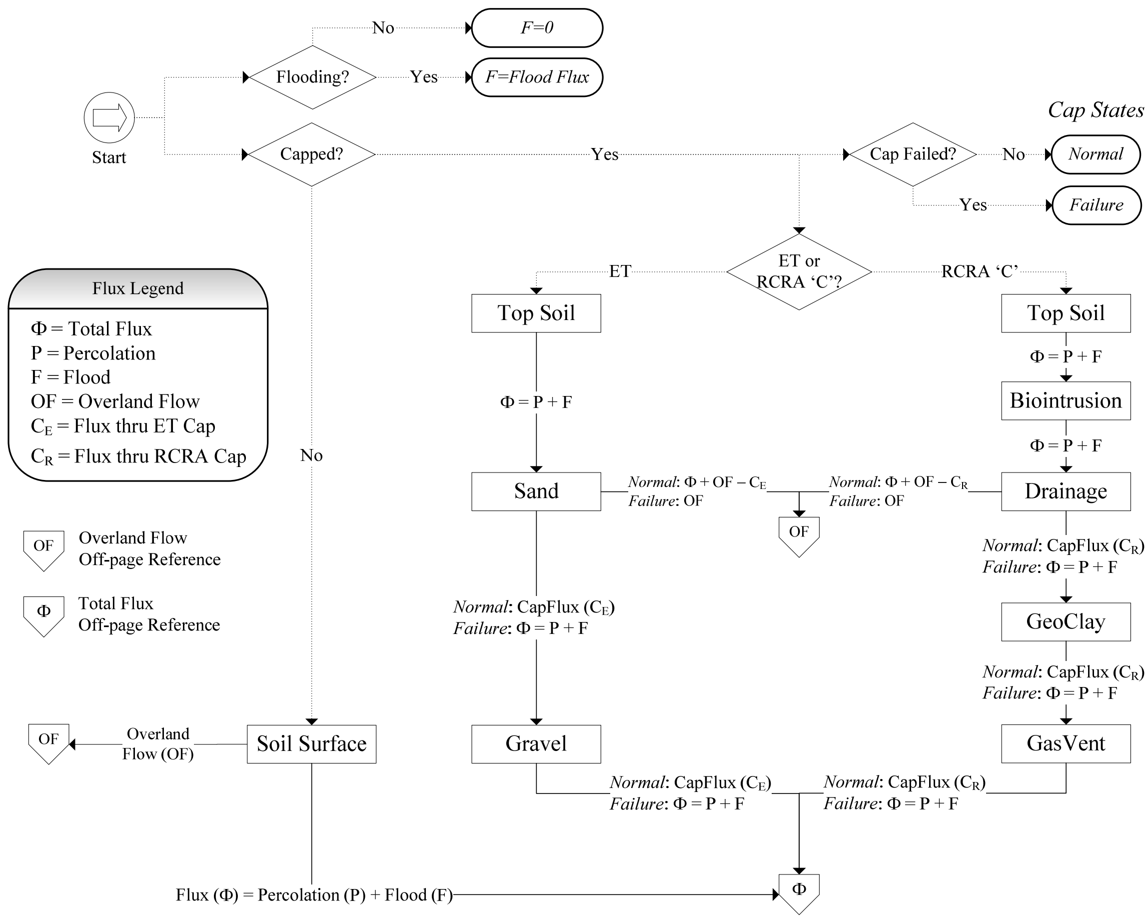

Figure 50. The Surface Soil Advective Flow Model for the Conceptual Burial Site Model for the Idaho Site SDA and Oak Ridge BCBG

Cap failure is modeled as a simple Poisson process with a mean time between failures based on cap type as shown in Table 28. The diagnosis of cap failure is assumed to be delayed by as much as the 5-year CERCLA review period. Once failure has been recognized, the cap is repaired assuming that repair activities take between one-fourth and twice as long as the initial barrier installation. Diagnosis and repair activities only take place during the Institutional Control period (e.g., 100 years after closure). While the cap has failed or is under repair, the flux of water through the cap is assumed to be that prior to capping but returns to the design or nominal flux after repairs are completed.

Table 28. Parameters for Modeling the Advective Flows for the Near Surface Layers as illustrated in Figure 49 and Figure 50

		Probabilistic		Point-Value		
Parameter	**Conditions**	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Percolation, P	Arid	LN2(1.43, 3.27)	(USDOE 1997)	10 cm/yr	(Holdren et al. 2006)	Mean values provided in reference. Large variation assumed to cover flooding.
	Humid	LN3(42.2, 10%)	Judgment	46 cm/yr	(USDOE 1997)	
Flooding, F (frequency)	Arid	P(3/55)	Judgment	(3/55) yr^{-1}	(Holdren et al. 2006)	Three parameters are used to model flooding; flow goes to the vadose zone. Three floods were used for statistics. For humid conditions, this is included in percolation.
Flooding, F (duration)	Arid	LN3(10, 20%)	Judgment	7 d	(Holdren et al. 2006)	
Flooding, F (volume)	Arid	LN(2.9 $x10^4$, 1.7 $x10^4$)	(Vigil 1988)	$6x10^4$ m^3	(Vigil 1988)	
Overland Flow, OF	Arid	D(0.0)	Judgment	0.0 cm/yr	(Holdren et al. 2006)	Overland flow only applies to humid conditions. Units are cm *of soil*/yr, and no net erosion is assumed.
	Humid	U(0.004[c], 0.012)	Judgment	0.012 cm/yr	(TDEC 2006) (USDOE 1997)	
ET Cap Flux, C_E	Both	D(1 cm/yr)[d]	(Mattson et al. 2004)	1 cm/yr	(Mattson et al. 2004)	Proposed performance requirement for SDA.
ET Cap Failure	Both	P(1/10 yr^{-1})	Judgment	1/10 yr^{-1}	Judgment	Most ET caps have been around for ~ 10 years.
RCRA Cap Flux, C_R	Both	LN2(1 cm/yr, 1.5)[e]	(Mattson et al. 2004)	1.9 cm/yr	(Mattson et al. 2004)	Flux equivalent to cap hydraulic conductivity.
RCRA Cap Failure	Both	P(1/30 yr^{-1})	(Mattson et al. 2004)	1/30 yr^{-1}	(Mattson et al. 2004)	30 year design life.
Time to Diagnose	Both	T(30 d, 1 yr, 5 yr)	Judgment	4 yr	Judgment	Quarterly analyses to 5-yr CERCLA review
Inundation, I	Humid, WA01	Not applicable	(SAIC 1996d)	N/A	(SAIC 1996d)	Only applies to humid conditions. WA01 is dry.
Inundation, I (duration)	Humid, WA02	LN(6 months, 1 month)	(SAIC 1996d) Judgment	240 d	Judgment	WA02 inundated at least during wet season. Expected GW flow of 5% and SW by difference. Assume 50/50 split possible.
Inundation, I (vol. fraction)	Humid, WA02	T(0.5,0.75,1.0)[f] T(0.75,0.95,1.0)	Judgment	0.92 0.975	Judgment	
Inundation, I (GW flow)	Humid, WA02	T(0,0.05,0.5)	(SAIC 1996d)	0.04	(SAIC 1996d)	
Inundation, I (duration)	Humid, WA03	LN(50 d, 10 d)	(SAIC 1996d) Judgment	68 d	(SAIC 1996d)	WA03 inundated during storm events. Flows are defined in the same manner as for WA02.
Inundation, I (flow fraction, vol. fraction)	Humid, WA03	Same as for WA02	(SAIC 1996d)	Same	(SAIC 1996d)	

a. The distributions used in the GoldSim model include: *Discrete* D(point value), *LogNormal* including LN(mean, stdev), LN2(gmean, gstd), and LN3(mean, % relative stdev), *Poisson* P(rate), *Triangular* T(minimum, most likely, maximum), and *Uniform* U(minimum, maximum) where mean and stdev are the arithmetic mean and standard deviation, and gmean and gstd their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Units are provided in this column.

c. The value of 957 lb/ac/yr was converted to 0.004 cm of soil/yr using the soil density (TDEC 2006).

d. It is assumed there is no uncertainty in this value because it is a performance requirement.

e. Unlike the ET flux, the RCRA cap installation is prescriptive and thus the flux is handled stochastically using the design value as the RCRA mean value.

f. The upper distribution is for the accessible (i.e., upper waste layer) and the lower distribution is for the inaccessible waste layer.

Flooding of the buried waste site is a function of both endogenous and exogenous factors. External conditions (i.e., a large snow melt in conjunction with a large rainfall event) caused flooding in the SDA on three occasions (Holdren et al. 2006; Vigil 1988). Flooding for arid conditions is simulated using three parameters (i.e., frequency, duration, and volume) based on the known SDA floods as shown in Table 28. During flooding events for the SDA, the waste pore space is assumed to be completely filled with water.

Flooding in the BCBG is a frequent event; however, frequent events are difficult to model in GoldSim without making the simulation inefficient. Instead flooding impacts for the BCBG are represented by treating percolation flow and Waste Area moisture content stochastically (i.e., including the average increased flood flow and moisture content in the percolation flow and waste moisture content).

Whereas, potential impacts of flooding must only be explicitly implemented for the SDA[131], impacts of inundation on exposure and risk must only be explicitly simulated for the BCBG Waste Areas. BCBG Waste Area 01 is dry with no inundation flow. Waste Area 02 is inundated at least during the wet season and perhaps as much as perennially and will be considered 6 months in length beginning every January 1 for convenience. Waste Area 03 is either subject to bathtubbing or inundated intermittently during storm events, which will be translated to an expected 50 days per year. During the inundation period, the saturated zone flow (*SZ*) moves through the accessible and inaccessible waste layers in proportion to their thicknesses. The presence of a cap has no impact on inundation flows. The parameters and distributions for modeling flows through the upper layers of the burial site model are provided in Table 28.

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[131] The GoldSim implementation has been configured so that potential impacts of flooding on the humid site (i.e., BCBG) can be explicitly simulated, if desired, and the necessary data become available. Similarly, inundation impacts on the arid site can be modeled if the site potentially has such impacts.

The flows described by the parameters in Table 28 translate into fluid and contaminant fluxes from the surface and Waste Areas and bottom soil layer to the vadose and saturated zones and surface water as shown in Figure 49. The continued transport of contaminants through these media to potential receptors is controlled by the advective flow parameters defined in Table 29. For example, the vadose zone flow (*VZ*) leaving the Bottom Soil is the nominal flux through the vadose zone; however, the interbed region for arid conditions may retard the flow, which would result in pooling above this layer.

The flow leaving the interbed and lower vadose zone enters the upper saturated zone (with flow *SZ*); the upper zone is used to predict drinking water exposure to contaminants transported to the aquifer. All water from the upper saturated zone is fed to the lower saturated zone and is eventually discharged to the surface water or goes to the groundwater sink (maintained to close the overall material balance). The amount of water discharged (*D*) to the surface water versus that going to the groundwater sink is a fraction of the total incoming flow as illustrated in Table 29.

For baseline conditions, the primary pathways for contaminant transport to potential receptors are by advection of contaminated water through the various media. Once contaminants have been released into the environment, they can move via diffusion through the vapor phase or advection to the atmosphere. Transport through the atmospheric pathway was previously described when the atmospheric medium was described because this medium, unlike the others, is not defined by physical boundaries (like the subsurface layers which have physical delineations) but instead by a theoretical stagnant boundary layer for diffusion and a mixing layer defining the vertical dimension of the layer for advection in the atmospheric layer as shown in Figure 34 and Figure 35.

Table 29. Parameters for Modeling the Advective Flows for the Vadose and Saturated Zones and Surface Water

		Probabilistic		Point-Value		
Parameter	**Conditions**	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Interbed Zone, IB_{max}	Arid	LU(0.2,213)	(Hubbell et al. 2004)	150 cm/yr	(Hubbell et al. 2004)	This is the maximum IB flux. Actual flow is the minimum of that exiting the Bottom Soil (*VZ*) and IB_{max}. Retarding flow expected for arid conditions.
	Humid	D(*VZ*)	Judgment	*VZ*	Judgment	
Saturated Zone, *SZ*	Arid	LN3(300,20%)	(Holdren et al. 2006)	194 m^3/yr	Judgment	Based on aquifer velocities for SDA and 10-month water flux for BCBG[c]. Large uncertainties are assumed.
	Humid	LN3(64800,20%)	(SAIC 1996d)	45,900 m^3/yr	Judgment	
Discharge, *D*	Arid	N(50%,5%)	(Holdren et al. 2006)	58%	Judgment	SRPA discharges into Snake River and 96% of BCBG water discharges into surface water[d].
	Humid	T(90%,96%,100%)	(SAIC 1996a)	99%	Judgment	
Surface Water, *S1*	Humid	D(0.11 m^3/s)[e]	(USDOE 2000)	0.11 m^3/s	Judgment	Surface water impacts for humid conditions only. Bear Creek flow rate.
Surface Water, *S2*	Humid	D(1.46 m^3/s)[e]	(USDOE 2000)	1.46 m^3/s	Judgment	Assumed same as that for East Fork Poplar Creek.
Surface Water, *SW*	Humid	LN3(132,20%)	(USDOE 2000)	94 m^3/s	Judgment	Clinch River average flow downstream side of Melton Hill Dam.

a. The distributions used in the GoldSim model include: *Discrete* D(point value), *LogNormal* LN3(mean, % relative stdev), *LogUniform* LU(low, high), *Normal* N(mean, stdev), and *Triangular* T(low, most likely, high) where mean is arithmetic mean and stdev is standard deviation.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. The area-weighted average (9.33 m/yr) of linear aquifer velocity for the SDA converted to flow rate using the cross-sectional aquifer area in the direction of flow shown in Figure 45. For the BCBG, the 10-month water flux at BCM 9.47 (Figure 46) was measured at 54,000 m^3 (SAIC 1996d) and was converted to the annual flow shown in the table.

d. The Snake River Plain Aquifer (SRPA) discharges into the Snake River and Yellowstone Basin and a 50-50 split is assumed without additional information. Approximately 96% of the water passing through the BCBG waste areas enters the surface water at BCM 9.47 (Figure 46).

e. Although there are uncertainties in the stream flow rates, the number of GoldSim *Cell Pathway* elements needed to model the stream pathways is fixed for the simulation (based on the flow rate and other parameters as illustrated above) and thus the flow rates must also be fixed.

The final advective transport process implemented in the model is barometric pumping. Weather patterns cause cyclical variations in barometric pressure above the buried waste site; as the barometric pressure decreases, gases can be drawn from the waste site to the atmosphere above the site (Nilson 1991). When the barometric pressure increases, uncontaminated air is forced into the waste areas. The net effect of these cycles may be the transport of volatile contaminants and radioactive gases to the atmosphere. In some cases, the effects of barometric pumping may be two orders of magnitude more than that due to molecular diffusion (Nilson 1991). A very simple model for estimating the maximum net impact of barometric pumping on contaminant transport is developed.

A useful measure of the barometric pumping transport process is the overall efficiency[132], η, from which it is possible to estimate the fractional amount ($\Delta M_c/M_0$) of a contaminant exiting each cycle (Nilson 1991):

$$\frac{\Delta M_c}{M_0} \approx 2\eta\left(\frac{\Delta p}{p_0}\right) \qquad [14]$$

where ($\Delta p/p_0$) is the fractional change in barometric pressure and M_0 the total mass of contaminant in the medium. The typical fracture spacing for both the fractured basalt underlying the SDA and the subsurface in the Bear Creek Valley is approximately 2 m (Magnuson 1995; SAIC 1996d; Unger et al. 2004), which translates into a fractional efficiency of $\eta \leq 0.1$ (Nilson 1991). Efficiencies of less than 10^{-3} are rapidly approached as the fracture spacing increases or decreases from 2 m (Nilson 1991).

An extreme weather cycle may produce a pressure variation of ($\Delta p/p_0$) = 2/30 with a typical fractional pressure change of 1/100 expected over the period of a few days (Nilson 1991). Periods between 1 and 70 days have been discovered in the literature for various conditions (Auer et al. 1996; Dresel and Waichler 2004; Hubbell et al. 2004; Nilson 1991; Parker 2003). If there are N such cycles during a given year, the total fraction of contaminant transported annually to the surface can be estimated using:

[132] The overall efficiency is a function of molecular diffusivity, bulk pneumatic diffusivity, and fracture spacing (Nilson 1991).

$$\left.\frac{\Delta M_c}{M_0}\right|_N \approx 1-\left[1-(2\eta)\left(\frac{\Delta p}{p_0}\right)\right]^N . \qquad [15]$$

Equation 15 accounts for the removal during previous cycles assuming the contaminants are immediately redistributed throughout the medium.

The N barometric pressure cycles during a year are represented as a continuous advective transport of the equivalent vapor mass. The parameters used to simulate the barometric pumping of volatile contaminants and gaseous radionuclides are provided in Table 30. As an example, the deterministic case presented in Table 30 translates into a fractional transfer of just more than 30% of the volatile contaminants and radioactive gases in the affected subsurface. In the GoldSim model, this transfer is implemented as an advective transfer of the same 30% per year of the pore vapor in the affected media.

Table 30. Parameters for Modeling Barometric Pumping

		Probabilistic		Point-Value		
Parameter	**Conditions**	**Distribution[a]**	**Reference**	**Value[b]**	**Reference**	**Comment**
Efficiency, η	Both	LN(0.05,0.02)	(Nilson 1991) Judgment	0.085	Judgment	Distribution appears to be unimodal.
Pressure change, $\Delta p/p_0$	Both	T(0,1/100,2/30)	(Nilson 1991) Judgment	0.053	Judgment	Triangular distribution used for lack of information.
Period	Both	T(1,20,70)	Various[c]	9.1 day	Judgment	The deterministic period corresponds to 40 cycles/yr.

a. The distributions used in the GoldSim model include: *LogNormal* LN(mean, stdev) and *Triangular* T(low, most likely, high) where mean is arithmetic mean and stdev is standard deviation.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. Periods between 1 and 70 days have been discovered in the literature for various sets of conditions (Auer et al. 1996; Dresel and Waichler 2004; Hubbell et al. 2004; Nilson 1991; Parker 2003). A period of 20 days was assumed for the SDA (Dresel and Waichler 2004), and this value is used as the most likely period (i.e., the mode) for both sites for lack of information.

Additional Advective Transport Mechanisms including Colloidal Transport, Runoff, and Resuspension

Facilitated transport of certain radionuclides (e.g., plutonium isotopes) upon association with colloids and other particles that may be transported with water can be an important transport mechanism (Honeyman 1999; Kersting et al. 1999; Nimmo et al. 2004). Facilitated transport is implemented using a Solid element (namely Colloid) that is suspended in and transported with the aqueous phase by advection (GTG 2005a). The amounts of the impacted radionuclides that are associated with the colloid are controlled by defining colloid-water partition coefficients as described in Appendix C. Preliminary research indicated the importance of the assumptions related to the presence and persistence of colloids on the predicted risks to potential receptors from the SDA.

As shown in Figure 50, another mechanism by which contaminants can be transported by their association with solids suspended in the aqueous phase is by overland flow or runoff. Runoff is characterized by the rate of soil transported per year to the surface water; however, it is assumed that there is no net soil loss (i.e., transported soil is replaced by clean soil). Suspended soil is transported through the surface water by defining sediment concentrations that correspond to the runoff rate.

Contaminated soil can be transported not only by runoff but also by resuspension of soil into the atmosphere above the buried waste site. Resuspension is modeled by determining the annual mass particulate loading in air and then by assuming that resuspension of contaminated soil accounts for some fraction of the loading[133]. The (Particulate Matter) PM-10 values are used to represent the particulate mass loading in the air. The PM-2.5 values are also reasonable candidates for representing the particulate

[133] For this research, the fraction will be assumed to be unity for lack of information but can be varied in the model to determine the sensitivity of the exposure and risk results to this parameter.

loading in the air (especially because the EPA revoked the annual PM-10 standard (ID-DEQ 2006)); however, the PM-10 values are larger and will maximize the exposure via the resuspension mechanism. Both sets of values are provided in Table 31, which defines the parameter values needed to simulate the resuspension pathway.

Table 31. Parameters for Modeling Contaminated Soil Resuspension

		Probabilistic		Point-Value		
Parameter	**Conditions**	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
PM-2.5	Arid	LN(7.2,1.3)	EPA AirData[c]	9.5 μg/m^3	Judgment	Based on 5 years of monitoring data from Bonneville County, ID.
	Humid	LN(15,1.9)	EPA AirData[c]	18 μg/m^3	Judgment	Based on 8 years of monitoring data from Roane County, TN.
	Limit	---	---	15 μg/m^3	EPA AirData[c]	Current EPA Air Quality Standard
PM-10	Arid	LN(24, 2.1)	EPA AirData[c]	28 μg/m^3	Judgment	Based on 5 years of monitoring data from Bonneville County, ID.
	Humid	LN(24, 2.2)	EPA AirData[c]	28 μg/m^3	Judgment	Based on 8 years of monitoring data from Roane County, TN.
	Limit	---	---	50 μg/m^3	EPA AirData[c]	This EPA standard has been revoked (ID-DEQ 2006).
Soil fraction	Both	D(1.0)	Judgment	1.0	Judgment	Particulate loading all from surface soil.
Maximum rate[d]	Both	LT(10^{-5},10^{-4}, 10^{-3})	(Tauxe 2004)	5x10^{-4} yr^{-1}	(Tauxe 2004)	Fractional average annual loss of radionuclides surface layer.

a. The distributions used in the GoldSim model include: *Discrete* D(point value), *LogNormal* LN(mean, stdev), and *LogTriangular* LT(low, most likely, high) where mean is arithmetic mean and stdev is standard deviation.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. These data are obtained from the U.S. EPA site "AirData - Monitor Values Report - Criteria Air Pollutants" available at http://www.epa.gov/oar/data/index.html (accessed March 13, 2008).

d. The maximum resuspension method from Tauxe (2004) in which a fixed annual rate of radionuclides is directly transported to the atmosphere above the burial site is also implemented in the model.

Plant-Induced Transport

The transport mechanisms described to this point are based upon physical processes including advection and diffusion. However, biological processes may also transport contaminants from the buried waste site or contaminated soils to potential receptors. The potential significance of biological transport on exposure and risk is highly site-dependent; therefore, the biological transport mechanisms implemented in this screening model are designed to prove the concept and provide an idea of the potential impact more than to accurately describe biological transport for all possible plant and animal species near buried waste sites.

Plant-induced transport of contaminants proceeds by contaminants being absorbed into the roots and then redistributed to the aboveground tissues of the plant (Kennedy and Strenge 1992; Tauxe 2004). During senescence the aboveground plant parts (and contaminants) are incorporated into the surface soils, and roots are incorporated into soils at their respective depths (Tauxe 2004). The method developed for the decommissioning of facilities licensed by the U.S. Nuclear Regulatory Commission (NRC) is used to model contaminant transport from soils to plants (Kennedy and Strenge 1992).

Two pathways for the plant-induced transport of contaminants are modeled: root uptake and resuspension to plant surfaces (Kennedy and Strenge 1992). The dominant pathway for a given element is represented using a soil-to-plant concentration factor[134], C_R (Bq/kg dry plant weight per Bq/kg soil), which converts from a soil concentration to an equivalent concentration in each of four food crops (i.e., leafy vegetables, other

[134] The concentration factors provided in NUREG/CR-5512, Vol. 1 (Kennedy and Strenge 1992) are considered geometric means with a large geometric standard deviation of 5.7 (Sheppard and Evenden 1997). These factors were updated in NUREG/CR-5512, Vol.3 (Beyeler et al. 1999). The distributions describing the concentration ratios are described in Table 41.

vegetables, fruits, and grain). Because the types of plants likely to grow (or to be grown) above the buried waste site may or may not include these four types of crops, the maximum of the four factors for each element is assumed to describe the impact of the biomass growing above the waste site.

The biomass growing above the waste site helps determine the amount of radionuclides that are transported to the surface soil. If there were no biomass, then there would be no plant-induced transport. Given the biomass, the ratio of the aboveground (or "shoots") to belowground (or "roots") parts can be defined as well as the maximum depth the roots are likely to travel. The cumulative fraction, Y, of roots above a given depth, Z (cm), can be obtained from the following asymptotic relationship (Jackson et al. 1996):

$$Y = 1 - \beta^{Z} \qquad [16]$$

where β is the extinction coefficient obtained from a regression analysis and is a function of both the type of plant and the conditions in which it is growing. Only grasses and shrubs are considered in this research; other plants can be added if deemed important. The parameters used to simulate plant-induced transport are provided in Table 32.

The parameters in Table 32 and the soil-to-plant concentration factors (Kennedy and Strenge 1992) provided in the model are used to estimate the contaminant masses fixed by the shoots. The fixation of contaminants is modeled as a direct annual transfer from the accessible waste layer to the surface. If a cap is installed, transport of contaminants to the surface via plant is only assumed to occur when the cap has failed or is under repair. The accuracy of the plant-induced transport of contaminants can be improved my incorporating site-specific information for the plants that may grow on the site.

Table 32. Parameters for Modeling Plant-Induced Transport

<table>
<tr><th></th><th></th><th></th><th colspan="2">Probabilistic</th><th colspan="2">Point-Value</th><th></th></tr>
<tr><th></th><th>Parameter</th><th>Conditions</th><th>Distribution[a]</th><th>Reference</th><th>Value[b]</th><th>Reference</th><th>Comment</th></tr>
<tr><td rowspan="9">Grasses</td><td rowspan="2">Total Biomass</td><td>Arid</td><td>LN(6.2,9.8)</td><td>(Hessing et al. 1996; Hooten and Myles 2006)</td><td>17 kg/ha/yr</td><td>Judgment</td><td>Used Yucca Mountain (YM) data for lack of better information.</td></tr>
<tr><td>Humid</td><td>LN3(13000, 48%)[c]</td><td>(Hui and Jackson 2006)</td><td>24,600 kg/ha/yr</td><td>Judgment</td><td>Average grassland data for multiple sites.</td></tr>
<tr><td rowspan="2">Root/Shoot ratio</td><td>Arid</td><td>LN3(1.53, 38%)[d]</td><td>(Hooten and Myles 2006)</td><td>2.62</td><td>Judgment</td><td>Used YM data and %RSD for shrubs.</td></tr>
<tr><td>Humid</td><td>LN(0.71, 19.8%)[c]</td><td>(Hui and Jackson 2006)</td><td>0.96</td><td>Judgment</td><td>Average grassland data for multiple sites.</td></tr>
<tr><td rowspan="2">Rooting depth, Z_max</td><td>Arid</td><td>LN3(140, 20%)[e]</td><td>(Foxx et al. 1984; Magnuson 1993)</td><td>190 cm</td><td>Judgment</td><td>Mean based on SDA data.</td></tr>
<tr><td>Humid</td><td>LN(85,5)</td><td>(Schenk and Jackson 2002)</td><td>93 cm</td><td>Judgment</td><td>Estimated from figure for prairie grasses.</td></tr>
<tr><td>Extinction parameter, β</td><td>Both</td><td>T(0.93,0.952, 0.972)</td><td>(Jackson et al. 1996)</td><td>0.97</td><td>Judgment</td><td>Estimated from Fig. 5 in Jackson et al. (1996)</td></tr>
<tr><td rowspan="7">Shrubs/Trees</td><td rowspan="2">Total Biomass</td><td>Arid</td><td>LN(115,115)</td><td>(Hessing et al. 1996; Hooten and Myles 2006)</td><td>289 kg/ha/yr</td><td>Judgment</td><td>Used YM data for lack of better information.</td></tr>
<tr><td>Humid</td><td>LN(13000, 3600)[f]</td><td>(Busing 2005; Zheng et al. 2004)</td><td>19,550 kg/ha/yr</td><td>Judgment</td><td>From ANPP and root to shoot distributions.</td></tr>
<tr><td rowspan="2">Root/Shoot ratio</td><td>Arid</td><td>LN(0.91,0.35)[d]</td><td>(Hooten and Myles 2006)</td><td>1.6</td><td>Judgment</td><td>Used YM data for lack of better information.</td></tr>
<tr><td>Humid</td><td>LN(0.42,0.18)[f]</td><td>(Ovington 1957)</td><td>0.75</td><td>Judgment</td><td>From tree root to shoot data.</td></tr>
<tr><td rowspan="2">Rooting depth, Z_max</td><td>Arid</td><td>LN3(457, 20%)</td><td>(Foxx et al. 1984)</td><td>617 cm</td><td>Judgment</td><td>Some tree data included.</td></tr>
<tr><td>Humid</td><td>LN(120,10)</td><td>(Schenk and Jackson 2002)</td><td>140 cm</td><td>Judgment</td><td>Extrapolated 95% rooting depth.</td></tr>
<tr><td>Extinction parameter, β</td><td>Both</td><td>T(0.96,0.978, 0.98)</td><td>(Jackson et al. 1996)</td><td>0.98</td><td>Judgment</td><td>Estimated from Fig. 5 in Jackson et al. (1996)</td></tr>
</table>

a. The distributions used in the GoldSim model include: *LogNormal* including LN(mean, stdev) and LN3(mean, % relative stdev) and *Triangular* T(low, most likely, high) where mean is arithmetic mean and stdev is standard deviation.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. This distribution is based on the annual averages for aboveground and belowground net primary production for 12 different grassland sites around the world (Hui and Jackson 2006).

d. A mean root to shoot ratio of 0.91 with a standard deviation of 0.35 is a % relative standard deviation (RSD) of 38%. For lack of data, this % RSD was used for grasses (Hooten and Myles 2006).

e. Foxx et al. (1984) suggest maximum root depths of between 76 and 396 cm for grasses native to the US. A depth of 140 cm was used to simulate moisture movement in barriers proposed for the SDA (Magnuson 1993).

f. The annual aboveground net primary production (ANPP) with mean and standard deviation of 9200 and 2200 kg/ha/yr, respectively, was provided for a temperate forest in the Great Smoky Mountains, Tennessee, USA (Busing 2005; Zheng et al. 2004). A root to shoot ratio, ϕ, for trees with mean and standard deviation of 0.42 and 0.18, respectively, was provided in Ovington (1957). These distributions were resampled using the relationship: Total Biomass = $(1 + \phi)$ ANPP to provide the distribution parameters given in the table. Any resemblance of this distribution to that for grasses is due to happenstance.

Animal-Induced Transport

Animal-induced transport is implemented in a fashion similar to that for plant-induced transport. Like plant-induced transport, the actions of burrowing animals as they excavate soil may result in soil and contaminant movement from the buried waste site to the surface. However, unlike plant roots, burrows created by animals may collapse thus moving contamination downward from the surface (Tauxe 2004). Contaminant transport via both burrow excavation and collapse are modeled in this research.

Two types of burrowing animals are represented in the model: ants and mammals. Site-specific information on the types and parameters describing burrowing animals can be used to increase the accuracy of the exposure and risk predictions. For example, burrowing animal information for only the SDA was found and is used for both sites. Because water is very difficult to come by at an arid site like the SDA (versus a humid site like the BCBG), it is appropriate to focus on the animal transport mechanism for the SDA and similar arid or semi-arid sites. For burrowing animals, the annual rate of burrow construction, *BR*, (a.k.a., the annual rate that soil is excavated) is given by:

$$\mathrm{BR(m^3 ha^{-1} yr^{-1}) = V_{Nest}(m^3) \times Density_{Colony}(ha^{-1}) \times Turnover(yr^{-1})} \qquad [17]$$

where the nest volume, density, and turnover rate are provided in Table 33.

The burrow construction rate, obtained from Equation 17, is multiplied by the relevant surface area and the fraction of the burrow in the Waste Area to provide the volumetric contaminant flux from each layer to the surface. The fraction, Y_a, of the burrow in the waste area can be obtained using a simplified gamma function providing the fraction of the burrows above a given depth, Z (Tauxe 2001; 2004):

Table 33. Parameters for Modeling Animal-Induced Transport

		Probabilistic		Point-Value		
	Parameter	Distribution[a]	Reference	Value[b]	Reference	Comment
Ant	Burrow volume	LN(1.77,1.01)	(Fitzner et al. 1979)	3.68 L	Judgment	Hanford 300 Area Study for Harvester ants (100-yr scenario). Burrow volume and maximum depth have a correlation coefficient of 0.75.
Ant	Maximum depth, Z_{max}	LN(2.28,0.415)	(Fitzner et al. 1979)	3.0 m	Judgment	
Ant	New Burrows	LN3(0.1,100%)	(Holdren et al. 2006)	0.28 yr^{-1}	Judgment	A value of 0.1 yr^{-1} provided and references are inconsistent in Holdren et al. (2006).
Ant	Colony Density	LN3(36,10%)	(Holdren et al. 2006)	42 ha^{-1}	Judgment	Only a value of 36 ha^{-1} is provided in Holdren et al. (2006).
Ant	Curvature, β	T(2.4,3.4,5.7)[c]	(Holdren et al. 2006)	4.1	(Tauxe 2004)	Ant data from Holdren, et al. (2006) were fit to Equation 18 from Tauxe (2004).
Mammals	Burrow volume	*A*: LN3(440,57%) *H*: LN3(120,57%)	(Holdren et al. 2006)	*A*: 910 L *H*: 250 L	Judgment	Sum over all mammals (100-yr scenario) for arid and excludes badger for humid. %RSD from ant data.
Mammals	Maximum depth, Z_{max}	*A*: LN3(3.0,18%) *H*: LN3(3.0,18%)[d]	(Sullivan 1996) (Holdren et al. 2006)	*A*: 4.0 m *H*: 4.0 m	Judgment	Assume maximum depth for badger for arid and rabbit for humid. %RSD from ant data.
Mammals	New Burrows	*A*: LN(2.4,10%) *H*: LN(0.76,10%)	(Holdren et al. 2006)	*A*: 2.8 yr^{-1} *H*: 0.89 yr^{-1}	Judgment	Volume-weighted sum over all mammals for the 100-yr scenario for arid and excludes badger for humid.
Mammals	Colony Density	*A*: LN3(7.0,10%) *H*: LN(17,10%)	(Holdren et al. 2006)	*A*: 8.1 ha^{-1} *H*: 20 ha^{-1}	Judgment	Volume-weighted sum over all mammals for the 100-yr scenario for arid and excluding badger for humid.
Mammals	Curvature, β	*A*: T(2.75,4.5,6.7)[c] *H*: T(2.38,3.7,5.5)[c]	(Holdren et al. 2006)	*A*: 5.0 *H*: 4.0	(Tauxe 2004)	Badger data from Holdren, et al. (2006) were fit to Equation 18 from Tauxe (2004) for arid and rabbit data for humid.

a. The distributions used in the GoldSim model include: *LogNormal* including LN(mean, stdev) and LN3(mean, % relative stdev), and *Triangular* T(low, most likely, high) where mean is arithmetic mean and stdev is standard deviation. "*A*:" represents arid conditions and "*H*:" represents humid. If nothing specified, distribution applies to both sets of conditions.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. The fraction of burrow volume at depth is tabulated in Holdren et al. (2006). As indicated in Tauxe (2004), the value of the curvature parameter, β, was varied so that the tabulated values were spanned.

d. The average rooting depth of 150 cm was provided in Holdren et al. (2006). No maximum rooting depth was provided for rabbits at the Idaho Site so the average value was doubled to 300 cm for this study.

$$Y_a = \left(1 - \frac{Z}{Z_{max}}\right)^{\beta - 1} \qquad [18]$$

where Z_{max} and β are the relevant parameters from Table 33. Equations 17 and 18 provide the basis for estimating the transport of contaminants to the surface in the soil excavated by burrowing animals. In the GoldSim model, this transport is modeled as a direct annual transfer of contaminants from the impacted layer to the surface.

The transport of contaminants to the surface with the soil excavated by burrowing animals is not the only transport that may result from these animals. The burrows created by the animals may collapse resulting in a downward movement of soil and contaminants. For each layer, the rate of burrow construction, *BR*, computed from Equation 17 multiplied by the surface area and burrow fraction is assumed to be the collapse volume. The cumulative collapse volume for each layer (i.e., from all higher layers) provides the basis for estimating downward contaminant transport. This transport is modeled as an annual transfer of contaminants from each impacted layer downward to the next.

Potential Receptor Scenarios and Exposure

A variety of representative receptors are used in the model to characterize exposures, doses, and risks to potential receptors. These receptors are meant to represent reasonable exposure scenarios and not worst-case scenarios. Specific receptors will be selected to represent general receptor categories instead of aggregating exposures or risks over populations that cannot be represented accurately into the future. If receptors are present, then their reasonable maximum or expected exposure and risks are estimated using the model. The model can be used to simulate either a single desired point-value estimate (e.g., best, 95%-percentile, etc.) or a probabilistic case. The general properties of the receptors defined for this study are provided in Table 34.

Table 34. Summary of Receptor Scenarios for the Conceptual Burial Site Model

	Receptor	Conditions	Distance from Source	Reference	Inhalation[a]	Immersion[a]	External[a]	Ingestion[a,b]	Dermal[a]	ICs Apply[c]	Comments
General Public	On-Site Resident	Both	0 m[b]	(USDOE 1997; 1999)	I_g,I_p, I_v	R_a	R_s	F_s,F_p, F_a,F_{dw}	D_s, D_{sh}	√	Primary residential receptor is subject to all pathways.
	Transient or Scavenger	Both	0 m	(USDOE 1997)	I_g,I_p, I_v	R_a	R_s	F_s	D_s	√	Secondary receptor applies to transient behavior.
	Off-Site Resident	Arid	4000 m[d]	(USDOE 1997)	I_g,I_p, I_v	R_a					This residential scenario is not impacted by ICs and is the minimal case. On-site resident can be used for selected off-site scenarios.
		Humid	720 m[d]	(USDOE 1997)	I_g,I_p, I_v	R_a					
	Recreational User	Arid	4000 m[d]	(USDOE 1997)	I_g,I_p, I_v	R_a					No surface water (and thus exposure) is assumed.
		Humid	720 m[d]	(USDOE 1997)	I_g,I_p, I_v	R_a, R_{sw}	R_{sr}	F_{sw},F_f	D_{sw}		This is primary scenario for humid conditions.
Workers	Direct Worker[e]	Both	0 m	(USDOE 1997)	I_g,I_p, I_v	R_a	R_s	F_s	D_s		This scenario is of primary importance for workers.
	Support Worker[e]	Both	100 m	(USDOE 1997)	I_g,I_p, I_v	R_a					Workers located at the area of maximum concentration.

a. Key to exposure pathways: gas inhalation (I_g), particulate/dust inhalation (I_p), VOC inhalation (I_v), external irradiation from immersion in air (R_a) or surface water (R_{sw}), external irradiation from soil (R_s) or shoreline (R_{sr}), soil ingestion (F_s), plant ingestion (F_p), animal ingestion (F_a), fresh fish ingestion (F_f), drinking water ingestion (F_{dw}), surface water ingestion (F_{sw}), dermal contact from showering (D_{sh}), and dermal contact with soil (D_s) or surface water (D_{sw}).

b. For this research, all drinking water well ingestion is assumed to be at the standard 100 m regulatory boundary (NRC 1994; USDOE 1999) regardless of the physical distance of the receptor to the source.

c. IC denotes Institutional Controls. A checkmark ('√') in this column indicates that Institutional Controls (ICs) impacts the exposure and risk calculation for this receptor.

d. These receptors are located at a distance representing the maximum exposure via the air pathway (Freeze and Cherry 1979; USDOE 1997).

Numerous parameters are needed to estimate the exposures for the receptors in this research. Site-specific information is desired when available. However, it is rare that site-specific information is available for key parameters let alone for all parameters that might significantly impact exposure. Furthermore, both uncertainty and variability information[135] may be needed for critical parameters to accurately represent errors in the exposure estimates.

[135] Uncertainty denotes a lack of information; whereas, variability describes true heterogeneity.

A number of simplifying assumptions are made to develop a simple-as-possible screening risk analysis model. Variability is managed by defining a set of six specific receptors that represent the inter-individual variation in receptor exposure. Furthermore, the distributions used to estimate exposures for each receptor are treated as uncertainty distributions (although they often represent both uncertainty and variability). This treatment will inflate the uncertainty obtained from a probabilistic analysis or result in an excessively high bounding exposure with unknown uncertainty for a point-value analysis. These results are considered reasonable for a screening type analysis.

The best distribution information available for the prototypic sites will be used in the screening analysis; however, any such distribution information should be replaced with site-specific information when possible. Changes are only made to distributions from the literature if a more reasonable distribution form is apparent (e.g., log-normal substituted for Gaussian) or if the desired distribution (e.g., three-parameter gamma distribution converted to Weibull) cannot be implemented in GoldSim. For those parameters without probability distributions in the literature, the author has used professional judgment (based on more than 20 years of experience in the area) to fit distributions to available information for the specified receptors.

On-Site Residential Scenario

The primary scenario describing potential *long-term* impacts to the general public is represented by the on-site resident (farmer), who would live, grow crops, and raise animals for personal use above the buried waste site. The on-site resident is exposed to contaminants via all media (i.e., air, soil, and water) and multiple routes (i.e., inhalation, external exposure to contaminated soil, immersion in contaminated air, ingestion of food and drinking water, and dermal contact via showering). This scenario is designed intentionally to represent reasonable, maximum impacts to hypothetical on-site residents from contaminants released into the environment from the burial site. It is very unlikely that actual future residents would be exposed to higher concentrations than those estimated using the screening model. Finally, on-site residents will be living in the area[136] (and exposed to site contaminants) only after Institutional Controls (ICs) are no longer in effect. The IC period is assumed to be 100 years (Holdren et al. 2006; SAIC 1996a).

Potential exposures to the on-site resident can be described by the relationships provided in Table 35. Focusing on the exposure relationships as is done in Table 35 appears unusual from the manner in which dose or risk is typically computed for exposure to radionuclides or hazardous chemicals (Kennedy and Strenge 1992; NCRP 1996a; USEPA 1989). The exposure part of the dose or risk calculation is frequently multiplied by the exposure-to-dose or exposure-to-risk conversion factor.

As described in Chapter II, a point-value calculation involves evaluating the *exposure* for a given scenario (e.g., worst-case, reasonable maximum, expected, etc.) and then multiplying the exposure (e.g., obtained the relationships like those in Table 35) by a benchmark dose or risk conversion coefficient set by regulatory fiat. These conversion factors are protective of the most sensitive populations and thus provide a bounding—not expected—value of the dose or risk corresponding to a given exposure.

[136] The user of the screening model has the ability to "turn off" Institutional Controls (ICs) so that impacts from the exposures to contaminants from the buried waste site can be estimated also during the IC period.

Table 35. Exposure Relationships for On-Site Resident Scenario
(Beyeler et al. 1999; NCRP 1996a)

Pathway	Exposure Relationship	Media Conc.[a]
Inhalation of Gases	$X_i^{inh}\left[\frac{m^3}{yr}\right] = \sum_j t_j V_j = (\underline{V} \bullet \underline{t})$; $i \in$ gases, $j \in$ activities t = time spent [hr/yr] by activity ($\underline{t}$ is the vector of times spent) V = breathing rate [m^3/yr] by activity ($\underline{V}$ is the vector of rates)	C_i^{air}
Inhalation of Particulates	$X_i^{pinh}\left[\frac{m^3}{yr}\right] = \sum_j t_j V_j PL_j^{public}$; $i \in$ particulates PL^{public} = particulate loading correction factor [dim'less] by activity	C_i^{air}
Immersion in Radioactive Cloud	$X_i^{air,imm}\left[\frac{hr}{yr}\right] = \sum_j t_j$; $i \in$ radionuclides	C_i^{air}
Direct Irradiation ("Groundshine")	$X_i^{soil,ext}\left[\frac{kg}{m^3}\right] = \rho_s \sum_j ShF_j t_j = \rho_s (\underline{ShF} \bullet \underline{t})$; $i \in$ radionuclides ρ_s = soil density [kg/m3] ShF = shielding factor [dim'less] by activity ($\underline{ShF}$ is the vector of factors)	C_i^{soil}
Ingestion of Drinking Water	$X_i^{dw}\left[\frac{m^3}{yr}\right] = IR^{dw} \sum_j t_j$; $i \in$ chemicals IR^{dw} = drinking water ingestion rate [m^3/day]	C_i^{dw}
Ingestion of Contaminated Soil	$X_i^{soil}\left[\frac{kg}{yr}\right] = IR^{soil} \sum_j t_j$; $i \in$ chemicals IR^{soil} = inadvertent soil ingestion rate [mg/day]	C_i^{soil}
Ingestion of Plants grown On-Site	$X_i^{plant}\left[\frac{kg}{yr}\right] = f_{local}^{plant} \sum_p \left(CR_{p,i}^{plant} + ML_{p,i}^{plant}\right)\left(IR_p^{plant} WD_p^{plant}\right)$; $i \in$ radionuclides, $p \in$ plant parts f_{local}^{plant} = fraction of plants consumed grown on-site [dim'less] CR^{plant} = concentration ratio [dim'less] by (radionuclide, part) ML^{plant} = mass loading on plants [dim'less] by (radionuclide, part) IR^{plant} = plant consumption rate [kg/yr] by plant part WD^{plant} = wet-to-dry mass ratio [dim'less] by plant part	C_i^{soil}
Ingestion of Animal Products (including Beef, Milk, Poultry, and Eggs) raised On-Site (Continued below)	$X_i^a\left[\frac{kg}{yr}\right] = f_{local}^{animal} IR_a^{animal} TF_{a,i}^{animal} IR_{a,i}^{soil}$ $IR_{a,i}^{soil} = \sum_f \left(CR_{f,i}^{feeds} + ML_{f,i}^{feeds}\right)\left(C_f^{feeds} IR_f^{feeds} + FR_a^{soil}\right)$; $i \in$ radionuclides, $a \in$ animal products, $f \in$ animal feeds f_{local}^{animal} = fraction of animal products grown on-site [dim'less] IR^{animal} = consumption rate [kg/yr] by animal product TF^{animal} = product transfer factor [d/kg] by (radionuclide, product)	C_i^{soil}

Table 35, Continued

Pathway	Exposure Relationship	Media Conc.[a]
Ingestion of Animal Products raised On-Site (Continued from above)	$IR_{a,i}^{soil} = \sum_f \left(CR_{f,i}^{feeds} + ML_{f,i}^{feeds}\right)\left(C_f^{feeds} IR_f^{feeds} + FR_a^{soil}\right)$ $FR_a^{soil} = f_a^{soil} C_a^{forage} IR_a^{forage}$; $i \in$ radionuclides, $a \in$ animal products, $f \in$ animal feeds CR^{feeds} = concentration ratio [dim'less] by (radionuclide, feed) ML^{feeds} = mass loading on feeds [dim'less] by (radionuclide, feed) C^{feeds} = fraction contaminated [dim'less] by product IR^{feeds} = feed consumption rate [kg/yr] by animal feed f^{soil} = soil ingestion rate per mass forage [dim'less] for animal C^{forage} = fraction of forage contaminated [dim'less] for animal IR^{forage} = forage consumption rate [kg/yr] for animal	
Dermal Contact with Soil	$X_i^{soil,derm} = SA \times ABS_i \sum_j EV_j AF_j t_j$; $i \in$ chemicals SA = skin surface area available for contact [cm^2/event] ABS = absorption fraction [dim'less] per chemical EV = event frequency [event/day] by activity AF = adherence factor [mg/cm^2] by activity	C_i^{soil}
Dermal Contact with Water (e.g., bathing, showering, etc.)	$X_i^{water,derm} = \left(\frac{DA_{event}}{C_w}\right)_i \times SA \times EF \times \sum_j t_j$; $i \in$ chemicals EF = exposure frequency (time spent) per day [min/day] C_w = concentration of chemical in water [mg/L] For *inorganic* chemicals ($i \in$ inorganic chemicals): $\left(\frac{DA_{event}}{C_w}\right)_i = K_{p,i} t_{event}$ K_p = permeability constant [cm/hr] by chemical $t_{event} \equiv$ EF [min/day] For *organic* chemicals ($i \in$ organic chemicals): $\left(\frac{DA_{event}}{C_w}\right)_i = 2FA_i K_{p,i} \sqrt{\frac{6\tau_{event,i} t_{event}}{\pi}}$ when $t_{event} \le t^*$ $= FA_i K_{p,i} \left[\frac{t_{event}}{1+B_i} + 2\tau_{event,i}\left(\frac{1+3B_i+3B_i^2}{(1+B_i)^2}\right)\right]$ otherwise FA, τ_{event}, t*, and B are chemical-specific parameters related to molecular weight and permeability coefficient, K_p, defined in USEPA (2004) and are provided in Attachment A.	C_i^{dw}

a. This column represents the media concentration by which the exposure result is multiplied to obtain the exposure to the receptor. When the constituent is a radionuclide, the concentration in the medium (C_i^m) will be in terms of Bq per m^3 for air, Bq per kg for soil, and Bq per m^3 for drinking water. For hazardous chemicals, the mass of the contaminant (in mg) is substituted for the activity (in Bq) in the exposure media concentrations.

The probability distributions (and their point-value counterparts) needed to estimate exposures to the on-site resident using the relationships in Table 35 are defined below after describing other receptors. As will be seen, there is considerable overlap in both the exposure relationships and how distributions are defined for the receptors. This overlap will be exploited to simplify and consolidate the presentation of the relevant sets of parameter and distribution descriptions after the remaining receptors and exposure relationships have been briefly described.

Transient or Scavenger (Intruder) Scenario

The transient or scavenger scenario describes potential impacts to members of the general public who do not live in the area above the buried waste site but would be transiting the area. The transient or scavenger would be exposed to contaminants via the inhalation of contaminated air, external radiation (air and ground), and the inadvertent ingestion of and dermal contact with contaminated soil. This scenario is designed to represent reasonable, maximum impacts to scavengers or intruders from contaminants from the burial site. It is unlikely that actual receptors would be exposed to higher concentrations than those predicted. The transient or scavenger scenario can be distinguished from the intruder by whether or not ICs limit exposure.

Potential exposures for this scenario are described by a subset of the relationships in Table 35 for the on-site resident. Because it is assumed that the transient or scavenger will neither drink contaminated site water nor ingest plants grown or animal products raised on the site, only the inhalation, immersion, external irradiation, and soil ingestion pathways apply. The primary differences in the results of the exposure relationships are the factors representing the time spent on-site by activity described below.

Recreational User Scenario

While the recreational user does not reside on the site, this person would be exposed to site contaminants via many pathways including some not applicable to the on-site resident or transient. New pathways are needed for the recreational user to estimate exposures to surface water contaminants including immersion in contaminated water, external exposure to the contaminated shoreline, ingestion of freshly caught fish and surface water, and dermal contact to the surface water. Potential exposures to the recreational user are described by the inhalation and air immersion relationships provided in Table 36.

The recreational scenario is designed intentionally to represent the reasonable, maximum impacts to a hypothetical off-site recreational user of surface waters contaminated by site wastes. It is very unlikely that actual recreational users would be exposed to higher concentrations than those calculated by the model. Recreational users are not impacted by the 100-year IC period defined for the prototype sites (Holdren et al. 2006; SAIC 1996a).

Off-Site Residential Scenario

Unlike the on-site residential scenario, which was designed to estimate maximum impacts to the general public, the off-site resident represents the *minimum* exposure and risk to the general public from contaminants released from the buried waste site. The off-site resident uses water from a supply other than the aquifer impacted by the site and eats food grown at areas not contaminated by site wastes. Furthermore, because the model incorporates the assumption that there is no deposition of contaminants to off-site soil, there is no exposure to contaminants in the soil.

Table 36. Exposure Relationships for the Recreational User Scenario (Beyeler et al. 1999; NCRP 1996a)

Pathway	Exposure Relationship	Media Conc.[a]
Inhalation of Gases	See Table 35	$C_i^{air,bound}$
Inhalation of Particulates	See Table 35	$C_i^{air,bound}$
Immersion in Radioactive Cloud	See Table 35	$C_i^{air,bound}$
Direct Irradiation from Shoreline	$X_i^{shore,ext}[m] = F_s \times U_{sh} \times F_w \times F_{d,i}$; $i \in$ radionuclides F_s = shoreline deposition velocity [m/d] U_{sh} = annual usage factor for shoreline activities[b] [d] F_w = shore-width correction factor [dim'less] $F_{d,i}$ = sorption-adjustment factor [dim'less] by radionuclide	C_i^{sw}
Ingestion of Fresh Fish	$X_i^{fish}\left[\frac{m^3}{yr}\right] = IR^{fish} \times K_i \times BF_i \quad \text{where} \quad K_i \equiv \frac{\lambda_b}{\lambda_i + \lambda_b};$ $i \in$ chemicals IR^{fish} = intake rate [kg/yr] of fish caught in surface water BF_i = bioaccumulation factor [m^3/kg] for fresh caught fish[c] K_i = term [dim'less] for radionuclide-specific BF, when needed λ_b = biological decay constant [1/d] = $\ln(2)/t_b$ λ_i = radiological decay constant [1/d] = $\ln(2)/t_i$ t_b = biological half-life [d] t_i = radiological half-life [d]	C_i^{sw}
Ingestion of Surface Water	$X_i^{sw}\left[\frac{m^3}{yr}\right] = IR^{sw} \sum_j t_j$; $i \in$ chemicals, $j \in$ activities IR^{sw} = inadvertent surface water ingestion rate [m^3/hr] t = time spent [hr/yr] by activity	C_i^{sw}
Immersion in Surface Water	$X_i^{sw,imm}\left[\frac{hr}{yr}\right] = t_{swim}$; $i \in$ radionuclides t_{swim} = time spent swimming in surface water per year [hr/yr]	C_i^{sw}
Dermal Contact while swimming	See Table 35	C_i^{sw}

a. This column represents the media concentration by which the exposure result is multiplied to obtain the exposure to the receptor. When the constituent is a radionuclide, the concentration in the medium (C_i^m) will be in terms of Bq per m^3 for air, per kg for soil, and per m^3 for drinking water. For hazardous chemicals, the mass of the contaminant (in mg) is substituted for the activity (in Bq) in the exposure media concentrations. The concentration, $C_i^{air,bound}$, of the contaminant in the air at the boundary where the maximally exposed individual is located is obtained by multiplying the contaminant flux from the burial site into the atmosphere by the appropriate dilution factor (χ/Q) obtained from Table 26.

b. The shoreline usage factor is currently assumed independent of the activities times defined for the recreational user.

c. The intake rate of fresh fish caught in the contaminated surface water is currently assumed independent of the activities times defined for the recreational user.

The off-site resident is exposed to contaminants from inhalation of and immersion in contaminated air. Potential exposures to the off-site resident under these conditions are described by the inhalation and air immersion relationships in Table 36. Like the recreational scenario, the exposure of off-site residents to contaminants is not impacted by ICs at the site. Although alternative scenarios may be defined, exposures to the general public are sufficiently bracketed by the scenarios defined here for a screening type analysis and captures enough of the essential exposure and risk elements from the buried waste sites for a comparison of remedial options.

Direct Worker Scenario

In addition to the four scenarios defined for the general public, scenarios will also be needed to describe exposures and corresponding impacts to workers, whose exposures will be different than those of the general public primarily due to the workers' proximity to the wastes. The direct worker is assumed to spend the majority of his or her time in the area directly above the buried waste site and is exposed to contaminants via inhalation of gaseous and particulate matter, immersion in contaminated air, and external exposure to and dermal contact with contaminated soil. It is assumed that direct workers do not ingest any site water or contaminated food but may inadvertently ingest contaminated soil[137].

The exposure relationships and media concentrations for the direct worker are the same as those for the transient or scavenger scenarios[138] in Table 35. The direct worker scenario is designed intentionally to represent the reasonable, maximum impacts to a

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[137] The model is set up so that workers (both direct and support) can drink from the contaminated aquifer; however, the water ingestion rate is currently set to zero and can be changed by the user.

[138] It is likely that the particulate loading correction for workers performimed heavy activities (or "gardening" for the general public scenarios) may have a different value.

hypothetical worker who works in the area above the buried waste site. It is very unlikely that actual workers will be exposed to higher concentrations than those predicted by the model. Finally, direct workers will only be in the contaminated area (and thus exposed) during the operational phase of the site. For convenience, it is assumed that the operational phase extends through the institutional control (IC) period.

Support Worker Scenario

The support worker spends the majority of his or her time outside the area above the buried waste and is exposed to contaminants only via inhalation of gaseous and particulate matter and immersion in contaminated air. Support workers do not ingest any site water or contaminated soil. The exposure relationships for the support worker are the same as those for the transient or scavenger scenarios (in Table 35); however, the contaminant concentration in the air is determined by multiplying the contaminant flux from the burial site by the appropriate dilution factor (χ/Q) from Table 26. It is unlikely that support workers who do not come into direct contact with the wastes will be exposed to higher concentrations than those predicted by the model. Finally, support workers will only be exposed during the operational phase of the site. For convenience, it is again assumed that the operational phase extends through the institutional control period.

Parameters Describing the Receptor Scenarios Used in the Model

Four human activities are defined, each with its own breathing rate, particulate mass loading factor, and amount of time performing the activity during a year. The activities are: resting (including sleeping), sedentary (including sitting awake), light activity (including light exercise), and heavy activity and exercise (including gardening)

(ICRP 1994; Tauxe 2004). For simplicity, resting and sedentary activities take place inside and the other activities take place outside. Adult male values are used in this research because the focus is on relative risks and risk trade-offs and the comparative results would likely be similar if a different age group were instead used. Only the time spent during the year for each activity is scenario-dependent. The scenario-independent parameters (i.e., breathing rates, particulate mass loadings, and shielding factors) needed to model inhalation and external exposure are provided in Table 37.

Particulate (or dust) loading factors allow corrections to be made for inhalation impacts as functions of location and activity. For example, the particulate loadings provided in Table 31 are long-term average values for outdoor exposure to contaminants in the air. However, if the receptor in question spends time indoors or performs activities (e.g., gardening, excavation, etc.) that would change the local mass loading, the resulting exposure could be significantly different.

Mass loadings for a number of locations and conditions are available in the literature (Colome et al. 1992; Tauxe 2004; Thatcher and Layton 1994; Yu et al. 1993); however, none are available for the sites under consideration. Furthermore, the distributions used in Tauxe (2004) are not referenced in the generic PA model or any supporting documentation. Instead of using the Tauxe distributions for mass loading, a consistent set of mass loading values were used to estimate the indoor to outdoor factor (Colome et al. 1992). Additional information, including that used to estimate the impact of resuspended indoor dust, is also available in NUREG-5512, Vol. 3 (Beyeler et al. 1999).

Table 37. Scenario-Independent Parameters for the Inhalation and External Pathways

		Probabilistic		Point-Value		
	Parameter	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Breathing Rate, V	Resting	T(0.4,0.45,0.5)[c]	(ICRP 1994; Tauxe 2004)	0.48 m^3/hr	Judgment	Resting and sleeping.
Breathing Rate, V	Sedentary	T(0.45,0.54, 0.70)[c]	(ICRP 1994; Tauxe 2004)	0.66 m^3/hr	Judgment	Sedentary activities (e.g., sitting awake).
Breathing Rate, V	Light Activity	T(0.6,1.5,1.7)[c]	(ICRP 1994; Tauxe 2004)	1.6 m^3/hr	Judgment	Light activities and light exercise.
Breathing Rate, V	Gardening (Heavy activities)	T(1.7,3.0,4.2)[c]	(ICRP 1994; Tauxe 2004)	3.8 m^3/hr	Judgment	Heavy activities (e.g., gardening).
Particulate loading correction, PL	Indoor Air, CDI[d]	LN(4.25,2.2)×10^{-4} *CDI = CDO×PF*	(Colome et al. 1992) (Beyeler et al. 1999)	8.3×10^{-4} g/m^3	Judgment	Colome et al. (1992) data from a sample of California houses and positively correlated.
Particulate loading correction, PL	Outdoor Air, CDO[d]	LN(6.1,2.7)×10^{-4} LU(1×10^{-7},1×10^{-4})	(Colome et al. 1992) (Beyeler et al. 1999)	1.1×10^{-3} g/m^3 7.1×10^{-5} g/m^3	Judgment	
Particulate loading correction, PL	Indoor to Outdoor Ratio, PF[d]	U(0.2,0.7)	(Beyeler et al. 1999)	0.67(5)	(Beyeler et al. 1999)	Alternative to using indoor/outdoor data.
Particulate loading correction, PL	Dust Loading on Floor, P_d	U(0.02,0.3)	(Beyeler et al. 1999)	0.29 g/m^2	(Beyeler et al. 1999)	Used to provide indoor resuspension exposure in addition to CDI.
Particulate loading correction, PL	Indoor Resuspension factor, RF_r	LU(1×10^{-7}, 8×10^{-5})	(Beyeler et al. 1999)	5.7×10^{-5} 1/m	(Beyeler et al. 1999)	
Particulate loading correction, PL	Gardening to Outdoor Ratio	U(1,7)[e]	(Kennedy and Strenge 1992)	6.7	Judgment	Gardening assumed 7× outdoor.
Particulate loading correction, PL	Construction to Outdoor Ratio	U(1,6)[e]	(Yu et al. 1993)	5.7(5)	Judgment	Construction activities assumed 6× outdoor.
Shielding, ShF	Indoor Shielding factor	T(0.04,0.33,0.4)	(Kennedy and Strenge 1992)	0.36	Judgment	The majority of shielding factors fall in the range provided.
Shielding, ShF	Outdoor Shielding factor	T(0.85,0.95,1.0)	(IAEA 2003)	0.98	Judgment	A factor of 0.7 may apply to gardening.

a. The distributions used in the GoldSim model include: *LogNormal* LN(mean, stdev), *LogUniform* LU(low, high), *Triangular* T(low, most likely, high), and *Uniform* (low, high) where mean is arithmetic mean and stdev is standard deviation.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided in this column.

c. The ranges for the triangular distributions were taken from Tauxe (2004) and the most likely point was taken from the values suggested in ICRP Publication 66 (ICRP 1994). The upper end of the "Light Activity" range was increased from a value of 1.3 m^3/hr to the minimum value (i.e., 1.7 m^3/hr) from the "Gardening" (a.k.a., "Heavy Activity") category to span the value provided in ICRP 66 (ICRP 1994).

d. The indoor-to-outdoor ratio method will be used in the model (Beyeler et al. 1999).

e. From information in multiple sources (Beyeler et al. 1999; Kennedy and Strenge 1992; Yu et al. 1993), short-term gardening and construction activities will produce localized, elevated loadings of up to approximately seven and six times, respectively, that of the outdoor level.

Unlike indoor activities that may produce lower exposures to suspended particulates, gardening and construction activities may produce localized, elevated particulate loadings of approximately seven and six times, respectively, that of the outdoor level (Beyeler et al. 1999; Yu et al. 1993). The indoor to outdoor ratio and the increased gardening and construction factors are used to compute "correction" factors that can be multiplied by the outdoor particulate mass loading to provide more accurate exposures. The indoor mass loading factor applies to resting and sedentary activities, the outdoor mass loading to light activities, and either the gardening or construction loading factor that applies to the gardening or heavy activities, respectively.

Two additional factors are defined in Table 37 to correct for shielding either due to building occupancy or clean soil covering the site. Although the impact of shielding will depend on the energy of the photons emitted by the site (in addition to the source dimensions and clean soil cover), a single factor is defined stochastically for each to remain in keeping with the screening nature of the analysis in this research (Kennedy and Strenge 1992). The shielding factor also depends on the type of building being occupied, for example, the shielding factor for a building may have a mode of 0.05 whereas that for a wooden house might be five times higher (IAEA 2003). However, most values fall within a range of 0.04 to 0.4, which is used in this study. The indoor shielding factor applies to resting and sedentary activities and the outdoor shielding factor applies to light and heavy (or gardening) activities.

Table 37 describes the scenario-independent distributions and deterministic values for modeling the inhalation and external pathways. However, the inhalation and external exposures are primarily functions of the time spent at the particular site and how the time is spent by activity. For ingestion pathways, the exposure rates to the contaminated media (i.e., soil and water) will also play an important role. The time spent and other relevant scenario-dependent parameters needed to model exposures to the two general public scenarios (i.e., on-site resident and transient) are provided in Table 38. Considerable information is available for the residential scenario; however, a great deal of judgment is needed to define parameters for the transient case. The parameters for the transient scenario are not definitive; however, likely differences in exposure and risk for cases involving the transient should be apparent from using the parameters in Table 38.

Table 38. Time-Spent-on-Site and Ingestion Parameters for Modeling the On-Site Resident and Transient Scenarios

		On-Site Resident		Transient or Scavenger	
	Parameter	**Distribution[a]|Value[b]**	**Reference**	**Distribution[a]|Value[b]**	**Reference**
Time Spent, *t*	Days on site	T(180,350,365)| 353 d/yr	(USEPA 2004)	LN3(12 d/yr,10%)| 14 d/yr	(USDOE 2003)
	Indoors	T(12,16.4,18)|17.3 hr/d	(USEPA 1997)	D(0)|0 hr/d	Judgment
	Resting	LN(8.7,2.3)|5.6 hr/d	(USEPA 1997)	D(0)|0 hr/d	Judgment
	Sedentary	Diff(Resting)|11.7 hr/d	Judgment	Diff(Resting)|0 hr/d	Judgment
	Showering	LN2(15 min/d,1.95)| 45 min/d	(USEPA 1997)	Not applicable	Judgment
	Outdoors	T(1,2,6)|5 hr/d	(USEPA 1997)	T(1,2,4)|3.5 hr/d	Judgment
	Light Activity	Diff(Heavy)|1.6 hr/d	Judgment	Diff(Heavy)|1.1 hr/d	Judgment
	Heavy Activity[c] (Gardening)	T(0.25,0.5,0.75)|0.67 or 3.4 hr/d	Judgment	T(0.25,0.5,0.75)|0.67 or 2.3 hr/day	Judgment
Ingestion Rate, *IR*	Drinking Water	LN2(2.1 L/d,1.435)[d]| 3.8 L/d	(Finley et al. 1994)	Not applicable	Judgment
	Surface Water	Not applicable	Judgment	Not applicable	Judgment
	Soil Ingestion	T(0,50,200)|160 mg/d	(Beyeler et al. 1999)	T(0,50,200)|160 mg/d	(Beyeler et al. 1999)

a. The distributions used in the GoldSim model include: *Discrete* D(point value), *LogNormal* including LN(mean, stdev), LN2(gmean, gstd dev), and LN3(mean, % relative stdev), and *Triangular* T(low, most likely, high) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts. Diff(*Parameter*) denotes a "unit-sum" constraint with *Parameter*.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. For want of specific information, the easiest way to manage the relative times spent between light and heavy activities is to define the fractional value for one (in this case, heavy activities providing the time spent) and then set the other (i.e., light activity) to the remaining outdoor time.

d. A log-normal distribution was determined by trial-and-error to reproduce (within reason) quantile data from Finley et al. (1994).

The parameters for the two off-site public scenarios (i.e., off-site resident and recreational user) are provided in Table 39. The parameters for time spent on-site by activity for the off-site resident are the same as those for the on-site resident. The primary difference between these receptors is that the on-site resident ingests contaminated soil and drinking well water whereas the off-site receptor is assumed to not use well water and soil is assumed uncontaminated by deposition of contaminants. The recreational user will inadvertently ingest contaminated surface water, that is, when surface water is present, and is considered to be outdoors for the entire exposure event.

Table 39. Time-Spent-on-Site and Ingestion Parameters for Modeling the Off-Site Resident and Recreational User Scenarios

		Off-Site Resident		**Recreational User**	
	Parameter	**Distribution[a]|Value[b]**	**Reference**	**Distribution[a]|Value[b]**	**Reference**
Time Spent, *t*	Days on site	T(180,350,365)|353 d/yr	(USEPA 2004)	LN(59,53)|135 d/yr	(USEPA 1997)
	Indoors	T(12,16.4,18)|17.3 hr/d	(USEPA 1997)	D(0)|0 hr/d	Judgment
	Resting	LN(8.7,2.3)|5.6 hr/d	(USEPA 1997)	D(0)|0 hr/d	Judgment
	Sedentary	Diff(Resting)|11.7 hr/d	Judgment	Diff(Resting)|0 hr/d	Judgment
	Showering	Not applicable	Judgment	Not applicable	Judgment
	Outdoors	T(1,2,6)|5 hr/d	(USEPA 1997)	T(0.5,2.6,8)|6.6 hr/d	(USEPA 1988)
	Light Activity	Diff(Heavy)|1.6 hr/d	Judgment	Diff(Heavy)|34 d/yr	Judgment
	Heavy Activity (Swimming)	T(0.25,0.5,0.75)|0.67[c] or 3.4 hr/d	Judgment	1: T(0,7,14)[d]|12 d/yr 2: T(0,2.6,5.2)[d]|4.4 hr/d	(USEPA 1988) Judgment
IR	Drinking Water	Not applicable	Judgment	Not applicable	Judgment
	Surface Water	Not applicable	Judgment	T(0,50,100)|84 mL/hr	(USEPA 1988)
	Soil Ingestion	Not applicable	Judgment	Not applicable	Judgment

a. The distributions include: *Discrete* D(point value), *LogNormal* LN(mean, stdev), and *Triangular* T(low, most likely, high) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts. Diff(*Parameter*) denotes a "unit-sum" constraint with *Parameter*.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. For want of specific information, the easiest way to manage the relative times spent between light and heavy activities is to define the fractional value for one (in this case, heavy activities providing the time spent) and then set the other (i.e., light activity) to the remaining outdoor time.

d. The only heavy activity for the recreational scenario is assumed to be swimming. Two distributions are provided to estimate the exposure frequency.

Table 40. Time-Spent-on-Site and Ingestion Parameters for Modeling the Direct and Support Worker User Scenarios

		Direct Worker		Support Worker	
	Parameter	**Distribution[a]|Value[b]**	**Reference**	**Distribution[a]|Value[b]**	**Reference**
Time Spent, *t*	Days on site	T(200,219,250)|241 d/yr	(USEPA 2004)	T(200,219,250)|241 d/yr	(USEPA 2004)
	Indoors	Diff(Outdoors)|5.1 hr/d[c]	Judgment	Diff(Outdoors)|5.1 hr/d[c]	Judgment
	Resting	T(0.05,0.10,0.25)|0.07[d] or 0.37 hr/day	Judgment	T(0.05,0.10,0.25)|0.07[d] or 0.37 hr/day	Judgment
	Sedentary	Diff(Resting)|4.7 hr/d	Judgment	Diff(Resting)|4.7 hr/d	Judgment
	Showering	Not applicable	Judgment	Not applicable	Judgment
	Outdoors	LN(1.5 hr/d,1.5)|2.9 hr/d	(USEPA 1997)	LN(1.5 hr/d,1.5)|2.9 hr/d	(USEPA 1997)
	Light Activity	Diff(Heavy)|1.7 hr/d	Judgment	Diff(Heavy)|2.3 hr/d	Judgment
	Heavy Activity (Gardening)	T(0,0.25,0.5)|0.42[d] or 1.2 hr/d	Judgment	T(0,0.1,0.25)|0.21[d] or 0.6 hr/d	Judgment
IR	Drinking Water	D(0)|0 L/d	Judgment	D(0)|0 L/d	Judgment
	Surface Water	Not applicable	Judgment	Not applicable	Judgment
	Soil Ingestion	T(50,330,480)|420 mg/d	(USEPA 2002)	T(50,100,200)|170 mg/d	(USEPA 2002)

a. The distributions include: *Discrete* D(point value), *LogNormal* LN(mean, stdev), and *Triangular* T(low, most likely, high) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts. Diff(*Parameter*) denotes a "unit-sum" constraint with *Parameter*.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. This result assumes that the average workday lasts 8 hours.

d. For want of specific information, the easiest way to manage the relative times spent between light and heavy activities is to define the fractional value for one (in this case, heavy activities providing the time spent) and then set the other (i.e., light activity) to the remaining outdoor time.

The parameters describing time spent working and the ingestion of contaminated media for the direct and support worker scenarios are provided in Table 40. The average workday for the workers is 8 hours, and these receptors are exposed to contaminants only while on-site. The types and durations of exposures for the worker scenarios are similar, with direct workers experiencing heavier activities and higher resulting soil ingestion.

Parameters for ingestion pathways are also needed. These pathways include the ingestion of plants and products (including beef, milk, poultry, and eggs) from animals raised on-site. Four types (i.e., leafy and root vegetables, fruits, and grains) of plants may be contaminated via plant-induced transport. The parameters needed to model exposures from the ingestion of plants grown on the site are described in Table 41.

Table 41. Parameters Needed to Model Exposure from Plant Ingestion

		Probabilistic		Point-Value		
	Parameter	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Consumption Rate, IR^{plant}	Leafy vegetables	LN2(12 kg/yr, 2.9)	(Beyeler et al. 1999)	66 kg/yr	(Beyeler et al. 1999)	Assumed equivalent to US EPA "exposed" vegetable category.
	Root vegetables	LN2(26 kg/yr, 2.9)	(Beyeler et al. 1999)	146 kg/yr	(Beyeler et al. 1999)	Annual consumption rate of root vegetables.
	Fruit	LN2(20 kg/yr, 3.9)	(Beyeler et al. 1999)	190 kg/yr	(Beyeler et al. 1999)	Annual consumption rate of fruits.
	Grain	LN2(8.2 kg/yr, 2.8)	(Beyeler et al. 1999)	44 kg/yr	(Beyeler et al. 1999)	Annual consumption rate of grains. Assumed zero in NUREG-5512, Vol 1. (Kennedy and Strenge 1992)
Mass Loading, *ML*	Leafy vegetables	LT(0.0011,0.1, 0.26) for each	(Beyeler et al. 1999; Kennedy and Strenge 1992)	0.16	Judgment	A constant value of 0.1 was used in NUREG-5512 for all elements and plant types (Beyeler et al. 1999; Kennedy and Strenge 1992). This information was turned into the distribution provided.
	Root vegetables					
	Fruit					
	Grain					
Wet-to-dry, *WD*	Leafy+Root vegetables	W(0.032,2.25, 0.076)	(Beyeler et al. 1999)	0.17	Judgment	A gamma distribution was proposed in Beyeler et al (1999) for vegetables and fruits.
	Fruit					
	Grain	U(0.88,0.89)	(Beyeler et al. 1999)	0.89	Judgment	A fixed value of 0.88 was suggested in the reference.
Concentration factors, *CR*	Leafy vegetables	"Leafy" values from Table 6.75 in (Beyeler et al. 1999)	(Beyeler et al. 1999; Kennedy and Strenge 1992)	Varies	Judgment	The manner in which these factors were defined in Vol. 1 (Kennedy and Strenge 1992) were simplified (i.e., placed into two instead of four categories) and updated in Vol. 3 (Beyeler et al. 1999). Distributions were assumed to be log-normal.
	Root vegetables	"Reproductive" values from Table 6.75	(Beyeler et al. 1999; Kennedy and Strenge 1992)	Varies	Judgment	
	Fruit					
	Grain					

a. The distributions used in the model include: *LogNormal* LN2(gmean, gstd dev), *LogTriangular* LT(low, mode, high), *Uniform* U(low, high), and *Weibull* W(min, slope, mean – minimum) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. It is assumed that noble gases (i.e., He, Ne, Kr, Xe, and Rn) are not taken up by plants, and no values were provided for various elements (i.e., Li, Ti, V, Tm, Yb, Lu, Pt, At, and Fr). These elements were assigned values of zero for the concentration factor. A factor of 6.92 Bq/kg dry plant weight per Bq/kg soil was taken from Tauxe (2004) for hydrogen.

The animal pathway involves the ingestion of poultry and beef products raised by the on-site resident and thus exposed to contaminants from the buried wastes; the model parameters are provided in Table 42. Poultry includes that contaminated poultry directly ingested by the on-site resident and the layer hens producing eggs that are then eaten. Cattle include both beef and dairy cattle raised on-site either directly consumed or producing milk (which is consumed).

Animals are exposed to contaminants in the soil either incidentally from the ingestion of contaminated soil during foraging activities or from ingestion of contaminated feeds grown on the site (Beyeler et al. 1999). The feeds included in the model are forage, grain, and hay. For this screening analysis, the lag times between growing and harvesting the feeds and potential storage are ignored, and the transfer from the contaminated soil to the animal is considered instantaneous (Tauxe 2004). The distributions needed to model the animal ingestion pathway are presented in Table 42.

One of the bases for the distributions provided in Table 42 was Volume 3 from NUREG/CR-5512 (Beyeler et al. 1999). Some of these distributions were given as three-parameter gamma distributions to which there is no equivalent in the GoldSim model (GTG 2005b; c). Weibull distributions were used in the model for the gamma distribution with the same minimum value, centered at the same location, and using a Weibull slope providing a similar distribution to the original. The Weibull slope was determined by trial-and-error based on a visual characterization.

Table 42. Parameters Needed to Model Exposure from Animal Ingestion

		Probabilistic		Point-Value		
	Parameter	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Average Consumption rate, IR^{animal}	Beef	LN2(29 kg/yr, 2.2)	(Beyeler et al. 1999)	105 kg/yr	(Beyeler et al. 1999)	From cattle raised on-site.
	Milk	LN2(150 kg/yr, 2.6)	(Beyeler et al. 1999)	740 kg/yr	(Beyeler et al. 1999)	Assuming a specific gravity of milk of 1.03.
	Poultry	LN2(20 kg/yr, 1.9)	(Beyeler et al. 1999)	59 kg/yr	(Beyeler et al. 1999)	From poultry raised on-site.
	Eggs	LN2(12 kg/yr, 2.3)	(Beyeler et al. 1999)	47 kg/yr	(Beyeler et al. 1999)	From poultry raised on-site.
Contaminated fraction, C^{feeds}	Beef	LT(0.5,1.0,1.0)	Judgment	0.99	Judgment	Both Vol. 1 and 3 of NUREG-5512 suggested using unity for these values (Beyeler et al. 1999; Kennedy and Strenge 1992) and made stochastic by author.
	Milk					
	Poultry					
	Eggs					
Product transfer factor, TR	Beef	T(0,*FA*,(20/19)*FA*) for each	(Beyeler et al. 1999; Kennedy and Strenge 1992)	FA	Judgment	Reference indicates that the chemical-specific factors (*FA*) represent upper-bounds and assumed the 95% values. Values provided in model.
	Milk					
	Poultry					
	Eggs					
Soil intake fraction, f^{soil}	Beef	LN3(0.02,50%)	(Beyeler et al. 1999)	0.04	Judgment	NUREG-5512 suggested using a constant value of either 0.05 (Vol. 1) or 0.02 (Vol. 3).
	Milk					
	Poultry	LN3(0.10,50%)	(Beyeler et al. 1999)	0.20	Judgment	NUREG-5512 suggested using a constant value of 0.10.
	Eggs					
Soil Mass Loading, ML	Forage	LT(0.0011,0.1, 0.26) for each	(Beyeler et al. 1999; Kennedy and Strenge 1992)	0.16	Judgment	Same chemical-specific distributions as used for plant ingestion (Table 41).
	Hay					
	Grain					
Wet to dry, WD	Forage	B(0.25,0.04,0.18,0.32)	(Beyeler et al. 1999)	0.31	(Beyeler et al. 1999)	Factors fit to distribution provided in reference. Hay and forage are equal.
	Hay					
	Grain	LN(0.90,0.014)	(Beyeler et al. 1999)	0.92	Judgment	Fit data from reference to log-normal distribution.
Concentration factors, CR	Forage	"Leafy" values from Table 6.75	(Beyeler et al. 1999; Kennedy and Strenge 1992)	Varies	Judgment	Concentration ratios assumed the same as those for leafy vegetables.
	Hay					
	Grain	"Reproductive" values from Table 6.75		Varies	Judgment	Concentration ratio distributions are those for grains in Table 41.
Dry forage rate (Beef cattle), IR	Forage	B(2.1,0.14,1.7,2.3) kg/d	(Beyeler et al. 1999)	2.3 kg/d	Judgment	The distributions are the same for both in the reference.
	Grain					
	Hay	B(4.2,0.28,3.4,4.6) kg/d	(Beyeler et al. 1999)	4.6 kg/d	Judgment	

a. The distributions used in the model include: *Beta* B(mean, stdev, min, max), *LogNormal* including LN(mean, stdev), LN2(gmean, gstd dev), and LN3(mean, % relative stdev), *LogTriangular* LT(low, mode, high), and *Triangular* T(low, most likely, high) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

Table 42, Continued

		Probabilistic		Point-Value		
	Parameter	Distribution[a]	Reference	Value[b]	Reference	Comment
Dry forage rate (Dairy cattle), *IR*	Forage	W(6.3,2.25,2.4) kg/d	(Beyeler et al. 1999)	11 kg/d	Judgment	Refit gamma distribution provided to Weibull for use in GoldSim.
	Grain	LN(1.71,0.262) kg/d	(Beyeler et al. 1999)	2.2 kg/d	Judgment	Normal distribution used in reference.
	Hay	W(5.0,2.25,1.9) kg/d	(Beyeler et al. 1999)	8.5 kg/d	Judgment	Refit gamma distribution provided to Weibull for use in GoldSim.
Dry forage rate (Poultry), *IR*	Forage	B(0.016,6.2x10^{-3}, 3.5x10^{-3},0.028) kg/d	(Beyeler et al. 1999)	0.026 kg/d	Judgment	
	Grain	B(0.049,0.019, 0.010,0.085) kg/d	(Beyeler et al. 1999)	0.078 kg/d	Judgment	
	Hay	D(0) kg/d	(Beyeler et al. 1999)	0 kg/d	Judgment	No hay in poultry feed.
Dry forage rate (Egg-layers), *IR*	Forage	B(0.019,2.7x10^{-3}, 0.012,0.022) kg/d	(Beyeler et al. 1999)	0.022 kg/d	Judgment	
	Grain	B(0.056,8.2x10^{-3}, 0.036,0.067) kg/d	(Beyeler et al. 1999)	0.066 kg/d	Judgment	
	Hay	D(0) kg/d	(Beyeler et al. 1999)	0 kg/d	Judgment	No hay in poultry feed.

a. The distributions include: *Beta* B(mean, stdev, min, max), *Discrete* D(point value), *LogNormal* LN(mean, stdev), and *Weibull* W(min, slope, mean – min) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

The final ingestion pathway that is represented in the model is that related to fish caught in the water contaminated from site wastes. Fish assimilate contaminants from the surface water that can be transferred to the recreational receptor via ingestion. The assimilation of contaminants in fish is represented using the bioaccumulation factor, *BAF* (NCRP 1996a):

$$\mathrm{BAF} = \frac{\mathrm{C_{fish}}}{\mathrm{C_{water}}}, \qquad [19]$$

which expresses the ratio of the contaminant concentration in fish to that in the surface water and may vary over several orders of magnitude.

The factors represented by Equation 19 and described in Table 43 were developed for stable elements and thus would be conservative for corresponding isotopes with relatively short radioactive half-lives and relatively long biological half-lives (NCRP 1996a). However, radioactive decay of a short-lived radionuclide can significantly reduce the concentration in the fish; this effect can be accounted for by multiplying the *BAF* from Equation 19 by the term:

$$K = \frac{\lambda_b}{\lambda_r + \lambda_b}, \quad \lambda_i = \frac{\ln(2)}{t_{½,i}} \qquad [20]$$

where $t_{½}$ is the appropriate half-life. A biological half-life of 30 days is considered appropriate for screening purposes (NCRP 1996a). The parameters used to model the exposure resulting from the ingestion of fresh fish caught in the contaminated surface water during recreational activities are described in Table 43.

The second type of exposure unique to the recreational user is that from external exposure to radionuclides deposited on the shoreline (Eckerman and Ryman 1993; NCRP 1996a). The exposure (and thus dose or risk) from external exposure to the i^{th} radionuclide deposited on the shoreline would be proportional to the following screening factor (NCRP 1996a):

$$SF_{shore,i} = F_s U_{sh} F_w F_{d,i} \qquad [21]$$

where F_s is the shoreline deposition velocity, U_{sh} the annual usage factor for shoreline activities, F_w is the shore-width correction factor, and $F_{d,i}$ is the sorption adjustment factor for the i^{th} radionuclide in consistent units. The parameters used to model the recreational exposure from external exposure to deposited radionuclides during shoreline activities are described in Table 43.

Table 43. Parameters Needed to Model Fish Ingestion and External Shoreline Exposure to the Recreational User

		Probabilistic		Point-Value		
	Parameter	**Distribution**[a]	**Reference**	**Value**[b]	**Reference**	**Comment**
Fish Ingestion	Fresh fish ingestion rate, *IR*	LN2(5.3,1.8)	(USEPA 1997)	14 kg/yr	Judgment	Those who fish and eat recreationally-caught fish
	Bioaccumulation factor, *BAF*[c]	LN2(*BAF* L/kg, 2)	(Dolislager 2006; NCRP 1996a)	Varies	Judgment	Chemical-specific factors for stable elements.
	Biological half-life, λ_b[c]	LN2(30 d, 2)	(NCRP 1996a)	94 d	Judgment	Radionuclide-specific factors in the model
External Shoreline	Time spent on Shoreline, U_{sh}	T(8, 2000, 2000) h/yr,	(NCRP 1996a)	1950 h/yr	(NCRP 1996a) Judgment	Likely very high; (Eckerman and Ryman 1993) suggests 8-12 hr/yr.
	Characteristic volume, *L*	LN2(1x10^{-3} m^3, 2)	(NCRP 1996a)	3x10^{-3} m^3	(NCRP 1996a)	Corresponding to F_s = 0.07 m/d—assumed *not* chemical-specific.
	Deposition velocity, F_s	$F_s = 100\ln(2)L$ m^{-2} d^{-1}	(NCRP 1996a)	0.22 m/d	Judgment	Computed from the characteristic volume, L.
	Shore-width factor, F_w	T(0.1, 0.2, 0.5)	(Eckerman and Ryman 1993; NCRP 1996a)	0.4	Judgment	Shore-width dose reduction factor.
	Sorption adjustment factor, F_d	D(F_d)	(NCRP 1996a)	Varies 0.1 to 10	(NCRP 1996a) Judgment	Chemical-specific factors provided in the model—assumed fixed.

a. The distributions include: *Discrete* D(point value), *LogNormal* LN2(gmean, gstd dev), and *Triangular* T(low, most likely, high) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. Bioaccumulation factors (Equation 19) are provided in the model for stable elements and organic compounds. The factors are "conservative" for those radionuclides with relatively short physical half-lives and relatively long biological half-lives although they can be corrected, if needed, using the relationship in Equation 20. In this research, a factor is computed for each radionuclide.

Additional parameters are required to model other exposure pathways that may be of importance to potential receptors defined in this study[139]. The parameters describing dermal contact with soil and water from showering (for the on-site resident) and swimming (for the recreational user) are provided in Table 44. The dermal absorbed dose for either soil or water contact is proportional to the absorbed dose per event, DA_{event}. For dermal contact with soil, the absorbed dose per event is given by (USEPA 2004):

[139] Under certain conditions, dermal exposure to contaminated water can be as significant as that from ingestion of the same water (USEPA 2004).

$$DA_{event} = (AF \times ABS_d)C_{soil} \qquad [22]$$

where AF is the adherence factor of skin to soil in mg/cm^2-event, ABS_d the unitless dermal absorption factor, and C_{soil} the concentration in the soil in mg/kg.

The relationships for dermal contact with contaminants in water can be more complicated. For dermal contact with inorganic contaminants in water, the absorbed dose per event is given by the following simple relationship similar to that for soil contact in Equation 22 (USEPA 2004):

$$DA_{event} = (K_p \times t_{event})C_w \qquad [23]$$

where K_p is the chemical-specific dermal permeability coefficient in cm/hr, t_{event} is the event duration in hr/event (assuming one event per day), and C_w is the concentration in the water in mg/cm^3.

On the other hand, the absorbed dose per event for organic compounds is estimated using either of the following relationships based on the exposure duration, t_{event} (USEPA 2004):

$$\begin{aligned} DA_{event} &= \left(2 \times FA \times K_p \times \sqrt{\frac{6 \times \tau_{event} \times t_{event}}{\pi}}\right)C_w \qquad t_{event} \le t^* \\ DA_{event} &= \left\{FA \times K_p \times \left[\frac{t_{event}}{1+B} + 2 \times \tau_{event} \times \left(\frac{1+3B+3B^2}{(1+B)^2}\right)\right]\right\}C_w \qquad \text{otherwise} \end{aligned} \qquad [24]$$

where, in this case, FA, τ_{event}, t^*, and B are chemical-specific parameters related to the molecular weight and the permeability coefficient, K_p. The permeability coefficients and relationships needed to evaluate the absorbed dose per event were taken from USEPA (2004) and are provided in the GoldSim model.

Table 44. Parameters used in Modeling Exposures via Dermal Contact

		Probabilistic		Point-Value		
	Parameter	Distribution[a]	Reference	Value[b]	Reference	Comment
Receptor	Body weight, *BW*	LN(78.1,13.5)	(Finley et al. 1994; USEPA 1997)	58.0 kg	Judgment	Adult male.
Receptor	Surface area, *SA*	LN(1.94, $3.7\text{x}10^{-3}$)	(USEPA 1997)	1.95 m^2	Judgment	Surface area per event for adult male.
Dermal Contact—Soil	Adherence factor, Indoors (*AF*)	D(0 mg/cm^2)	(USEPA 2004)	0 mg/cm^2	Judgment	Applies to resting and sedentary activities.
Dermal Contact—Soil	Adherence factor, Light activity (*AF*)	LN2(0.1 mg/cm^2, 1.95)	(USEPA 2004)	0.3 mg/cm^2	Judgment	Distribution "fit" to construction data.
Dermal Contact—Soil	Adherence factor, Heavy activity (*AF*)	LN2(0.2 mg/cm^2, 2.14)	(USEPA 2004)	0.7 mg/cm^2	Judgment	Distribution "fit" to utility worker data.
Dermal Contact—Soil	Absorption fraction, inorganic (*ABS*)	LN2($1\text{x}10^{-3}$,2) for each	(Dolislager 2006; USEPA 1992)	0.003	Judgment	Chemical-specific factors.
Dermal Contact—Soil	Absorption fraction, organic (*ABS*)	LN2(0.01,2) for each	(Dolislager 2006; USEPA 1992)	0.03	Judgment	Chemical-specific factors.
Dermal Contact—Soil	Absorption fraction, semivolatile	LN2(0.1,2) for each	(USEPA 2004)	0.3	Judgment	Chemical-specific factors.
Dermal Contact—Soil	Event frequency, indoors (*EV*)	D(0 event/d)	Judgment	0 event/d	Judgment	Applies to resting and sedentary.
Dermal Contact—Soil	Event frequency, outdoors (*EV*)	D(1 event/d)	(USEPA 2004)	1 event/d	Judgment	Applies to light and heavy activities.
Dermal–Water	Permeability, inorganic (K_p)	LN2($1\text{x}10^{-3}$,2)[c] for each	(USEPA 2004)	0.003	Judgment	Chemical-specific coefficients.
Dermal–Water	Permeability, organic (K_p)	LN2(*gmean*,2)[d] for each	(USEPA 2004) (Dolislager 2006)	Varies[d]	Judgment	Chemical-specific coefficients.
Dermal–Water	Time spent, Showering (t_{event})	See Table 38				On-site resident scenario.
Dermal–Water	Time spent, Swimming (t_{event})	See Table 39				Recreational scenario.

a. The distributions used in the GoldSim model include: *Discrete* D(point value) and *LogNormal* including LN(mean, stdev) and LN2(gmean, gstd dev) where *mean* is arithmetic mean and *stdev* is standard deviation and *gmean* and *gstd dev* are their geometric counterparts.

b. The deterministic value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value.

c. Specific permeability coefficients are provided in Exhibit 3-1 of USEPA (2004) for Cd, Cr (+3 and +6), Co, Pd, Hg (+2), Ni, K, Ag, and Zn. Specific factors are used as the geometric mean when available. All other inorganic use $1\text{x}10^{-3}$ cm/hr as the geometric mean (USEPA 2004).

d. The specific predicted permeability coefficients for many of the organic constituents are provided in Exhibit B-2 of USEPA (2004). The values for any omitted compounds are taken from the Risk Assessment Information System (Dolislager 2006). The values used are provided in the GoldSim model. See Equation 24 for the manner in which to correct the permeability constant for event duration.

Converting from Exposure to Dose and Risk

The parameters, X_i^{medium}, defined in Table 35 and Table 36 can be used to convert a concentration of the ith contaminant in a medium (e.g., soil, air, water, etc.) to dose or risk using the methods provided by the U.S. Nuclear Regulatory Commission (NRC) and National Council on Radiation Protection and Measurements (NCRP) (Beyeler et al. 1999; Kennedy and Strenge 1992; NCRP 1996a; b; Tauxe 2004). The exposure relationships (in Table 35 and Table 36) and appropriate dose or risk conversion factors (e.g., those in Federal Guidance Reports 11, 12, and 13 (Eckerman et al. 1999; Eckerman and Ryman 1993; Eckerman et al. 1988)) are used to define pathway conversion factors.

For a given scenario, pathway conversion factors are constants and are multiplied by an exposure media concentrations to provide dose or risk estimates (Tauxe 2004). Initial computation of the pathway dose or risk conversion factors allow for rapid dose or risk calculation. Computational efficiency is of increasing importance when dose or risk must be estimated for many receptors, media, time steps, and/or model realizations, which is likely the case for the screening risk model developed in this research.

Pathway Dose Conversion Factors for Radionuclides

The pathway dose conversion factors (PDCFs) used in the screening model to convert from exposure media concentrations of radionuclides to annualized total effective dose equivalents (or "dose" for short) are defined in Table 45. Like the exposure factors presented in Table 35 and Table 36, the PDCFs are computed initially for subsequent use during simulation studies. This "preprocessing" step allows large simulation studies to be executed much more efficiently.

Table 45. Pathway Dose Conversion Factor (PDCF) Summary for Radionuclides

Pathway	Pathway Dose Conversion Factor (PDCF)[a]	Media Conc.
Inhalation of Gases or Particulates	$PDCF_i^h\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right] = X_i^h\left[\frac{m^3}{yr}\right]\times DCF_i^{inh}\left[\frac{Sv}{Bq}\right]$; $i \in$ radionuclides X_i^h = exposure relationship from Table 35 for gases (h=inh) or particulates (h=pinh) DCF_i^{inh} = inhalation dose conversion factor (ICRP 68/72)	$C_i^{air}\left[\frac{Bq}{m^3}\right]$
Immersion in Radioactive Cloud	$PDCF_i^{air,imm}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right] = X_i^{air,imm}\left[\frac{hr}{yr}\right]\times DCF_i^{air,imm}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right]$ $X_i^{air,imm}$ = exposure relationship from Table 35 $DCF_i^{air,imm}$ = air immersion dose conversion factor (FGR-12)	$C_i^{air}\left[\frac{Bq}{m^3}\right]$
Immersion in Surface Water	$PDCF_i^{sw,imm}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right] = X_i^{sw,imm}\left[\frac{hr}{yr}\right]\times DCF_i^{water,imm}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right]$ $X_i^{sw,imm}$ = exposure relationship from Table 36 $DCF_i^{water,imm}$ = water immersion dose conversion factor (FGR-12)	$C_i^{sw}\left[\frac{Bq}{m^3}\right]$
Direct Irradiation from Ground	$PDCF_i^{soil,ext}\left[\frac{Sv\text{ - }kg}{Bq\text{ - }yr}\right] = X_i^{soil,ext}\left[\frac{kg}{m^3}\right]\times DCF_i^{soil,ext}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right]$ $X_i^{soil,ext}$ = exposure relationship from Table 35 $DCF_i^{soil,ext}$ = air immersion DCF for infinite depth (ICRP 72)	$C_i^{soil}\left[\frac{Bq}{kg}\right]$
Direct Irradiation from Shore	$PDCF_i^{shore,ext}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right] = X_i^{shore,ext}[m]\times DCF_i^{ground}\left[\frac{Sv\text{ - }m^2}{Bq\text{ - }yr}\right]$ $X_i^{soil,ext}$ = exposure relationship from Table 36 DCF_i^{ground} = "ground-shine" dose conversion factor (FGR-12)	$C_i^{sw}\left[\frac{Bq}{m^3}\right]$
Ingestion of Drinking or Surface Water or Fish	$PDCF_i^{w}\left[\frac{Sv\text{ - }m^3}{Bq\text{ - }yr}\right] = X_i^{w}\left[\frac{m^3}{yr}\right]\times DCF_i^{ing}\left[\frac{Sv}{Bq}\right]$ X_i^{w} = relationship from Table 35 for drinking water (w=dw) or Table 36 for surface water (w=sw) or fish (w=fish) DCF_i^{ing} = ingestion dose conversion factor for adult (ICRP 68/72)	$C_i^{w}\left[\frac{Bq}{m^3}\right]$
Ingestion of Soil, Plants, or Animal products	$PDCF_i^{k}\left[\frac{Sv\text{ - }kg}{Bq\text{ - }yr}\right] = X_i^{k}\left[\frac{kg}{yr}\right]\times DCF_i^{ing}\left[\frac{Sv}{Bq}\right]$ X_i^{k} = exposure relationship from Table 35 for soil (k=soil), plant (k=plant), or animal product (k=a)	$C_i^{soil}\left[\frac{Bq}{kg}\right]$

a. The ICRP Publication 72 dose conversion factors are used for the general public and ICRP-68 factors are used for workers (ICRP 1995; 1996). These factors were taken from the *RadToolBox* by Eckerman (2003). The air immersion dose conversion factors as a function of depth were taken from Federal Guidance Report 12 (Eckerman and Ryman 1993). When multiple factors are provided for a given radionuclide (e.g., for different lung clearance classes), the maximum factor for the radionuclide is used for screening purposes to simplify implementation. All dose conversion factors are provided in the screening risk model.

The PCDFs used in the screening risk model are represented by the factors in Table 45. However, the annual dose estimated using the PCDF is only one of several metrics that can be related to the potential exposure to contaminants and, furthermore, pertains only to radionuclide exposure. Other metrics are available that convert exposures to risks including latent cancer fatalities (LCF) for carcinogens, morbidity (i.e., cancer incidence), mortality (i.e., cancer-induced fatalities), etc. Many chemicals also have non-carcinogenic impacts that are typically described using a hazard quotient (or index) that represents the ratio of the exposure concentration to a reference dose (Crump et al. 1995).

Pathway Risk Conversion Factors for Radionuclides

Pathway risk conversion factors (PRCFs) are defined that are analogous to the PCDFs. The PRCFs, which provide annualized risks[140], are defined by substituting the pertinent morbidity risk coefficient (in excess lifetime total cancers/Bq) from the Health Effects Assessment Summary Tables (HEAST) (USEPA 2001a; b) or mortality risk coefficient (in excess lifetime total cancer fatalities/Bq) from Federal Guidance Report 13 (Eckerman et al. 1999) for the dose conversion factors in Table 45. These factors provide the mortality and morbidity risks corresponding to the doses using PCDFs.

The risks and doses for radionuclides obtained from these conversion factors are compared to determine whether risks alone can be used for comparison purposes. If the exposure risks to carcinogens generally present the same information as dose for radionuclides, then the risks related to radionuclide exposures can be compared—in the

[140] Using the pathway dose or risk conversion factors, an exposure concentration is converted to an annual dose or risk, respectively. That is, the exposure duration (in the standard EPA vernacular) is not included in the definition to prevent confusion concerning the results obtained. For example, a graph showing dose or cancer incidences or fatalities per year on the ordinate has much less chance of being misinterpreted than an ordinate suggesting a total number of fatalities.

proper context—to other risk metrics. This abstraction to risk would be important in that there is no strict, rigorous conversion from total dose to risk for radionuclides. If comparisons could be placed on a risk basis, then results would be more transparent.

Pathway Carcinogenic Risk Conversion Factors for Carcinogens

The pathway conversion factors provide estimates of the impacts to receptors from exposures to radionuclides that have migrated from the burial site. However, many chemicals buried in DOE sites are not radioactive but may represent significant risks to potential receptors. The EPA has an established methodology for estimating potential impacts (i.e., excess total lifetime latent cancer incidences) from exposure to chemical carcinogens (USEPA 1989). The pathway carcinogenic risk conversion factor (PCRCF) used in this study for the i^{th} chemical can be computed using the following relationship:

$$\mathrm{PCRCF_i} = \mathrm{X_i} \times \left(\frac{\mathrm{SF_i}}{\mathrm{BW} \times \mathrm{AT}} \right) \quad [25]$$

where X_i is the appropriate exposure relationship from Table 35 or Table 36, SF_i is the slope factor from the Risk Assessment Information System (Dolislager 2006), *BW* is the body weight as described in Table 44, and *AT* is the averaging time of 70 years for carcinogenic effects as suggested by the EPA (USEPA 1989).

Pathway Non-carcinogenic Risk Conversion Factors for Carcinogens

The hazard quotient (HQ) is an indication of an adverse health-related impact from exposure to a chemical. Pathway conversion factors are defined that relate the exposure media concentration to HQ. The pathway non-carcinogenic risk conversion factors (PNRCFs) differ from those developed for dose or risk in that the HQ is not annualized because such a value would have little meaning to the casual or untrained reader. An HQ greater than unity implies that an adverse health effect is possible.

The hazard quotient (HQ) is the ratio of the chronic daily intake, *CDI*, to a reference dose, *RfD*, expressed in the same units (mg/kg-day) over the same exposure period (USEPA 2001c):

$$HQ = \frac{CDI}{RfD}. \qquad [26]$$

The HQ has no statistical interpretation in that a value of 1×10^{-4} in no way implies there is a 1 in 10,000 chance that the effect will occur (USEPA 1989). However, the greater the HQ is above unity, the greater should be the concern regarding exposure to the chemical (USEPA 1989). The pathway non-carcinogenic risk conversion factor (using X_i from Table 35 or Table 36) becomes (USEPA 1989):

$$PNRCF_i = X_i \times \left(\frac{ED}{BW \times AT} \right) \times \frac{1}{RfD_i} \qquad [27]$$

where the averaging time (*AT*) in Equation 27 is equal to the exposure duration (*ED*) for non-carcinogenic effects instead of 70 years for carcinogenic effects.

Potential Comparisons of Dose, Risk, and Adverse Health Effects

Pathway conversion factors are defined that convert an exposure concentration to corresponding dose, risk, and adverse health effects. However, exposure is the common basis for these potential effects and highlights the possibility of comparing remedial options using exposure or risk information. Initially, dose and hazard results should be compared to those for risk to determine if remedial alternatives can be legitimately compared on solely a risk basis.

However, receptors may experience both exposure and standard industrial risks. It is desired to compare selected potential impacts (e.g., morbidity risks from radionuclides, latent cancer incidence risks for chemicals, worker injury risks, etc.) for remedial alternatives without significant loss of information (e.g., from not analyzing dose and adverse health effects). Whereas the possibility for comparing potential radionuclide impacts appears promising (i.e., mortality and dose are exposure-driven), the possibility of relating non-carcinogen effects to other metrics appears improbable. No common basis exists for comparing non-carcinogen and carcinogen effects for chemicals, and probabilistic interpretation of non-carcinogen effects is without foundation. Therefore, as is typically the case, potential non-carcinogen effects should be presented alongside those for the cancer-related impacts.

Standard Industrial Risk Analysis

A worker is exposed to various hazards. Depending on the workplace, hazards may involve potential exposure to hazardous chemicals or radiation, which can be evaluated using the methods provided in previous sections. However, during site cleanup, these exposure risks are often *not* the dominant sources of risk to the workers (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). Most site cleanups, especially those involving excavation and retrieval of hazardous and radioactive wastes, resemble heavy construction sites and the primary risk drivers are the same. If wastes are transported off-site over long distances, then transportation accidents, even without radionuclide or hazardous chemical releases, may be a significant, if not dominant, risk component.

The basic relationship for estimating standard industrial risk is that the annual risk of standard industrial injury or fatality is proportional to the time worked per year (i.e., Risk $\propto$ time worked). The proportionality can be represented by a risk factor, *RF*, derived from statistical analysis of historic accident information[141]. Data and statistics are available for injuries, fatalities, total recordable case, etc. for various categories of workers; the focus for the screening risk analysis will be injuries and fatalities to workers.

Screening Injury and Fatality Risk Factors

The total standard industrial risk of injury or fatality (i.e., *type*) is the relevant risk factor multiplied by the time worked ($t_{activity}$) for a given scenario is given by

$$\mathrm{Risk}^{\mathrm{scenario}}\left[\mathrm{type/year}\right] = \mathrm{RF}_{\mathrm{type}}^{\mathrm{scenario}}\left[\mathrm{type/year}\right] \times \sum_{\mathrm{activities}} t_{\mathrm{activity}}^{\mathrm{scenario}}\left[\mathrm{hr/yr}\right]. \qquad [28]$$

The time spent on-site for all activities must be used to estimate risk because the risk factors from typical historical data (BLS 2005; 2007) are given on an industry—not activity—basis. The aggregation of the injury and fatality statistics by industry (and thus worker type) may inaccurately estimate standard industrial risks depending upon the types of activities being performed by the workers. For example, the time spent on-site by either the direct or support worker (for exposure risk) is allocated between various activities (i.e., resting, sedentary, light activity, heavy activity). More accurate standard industrial risks may be estimated if risk factors could be obtained on an activity (and certainly DOE site) basis instead of an industry-wide or worker-type basis.

141 An example of historical information is available at the U.S. Bureau of Labor Statistics (BLS 2007).

Two parameters are used in the standard industrial risk model (Equation 28): time spent per activity per year and the risk factor providing the injuries or fatalities expected per hour worked on a particular type of activity. It would be advantageous if risk factors were available in the open literature for DOE sites for the types of workers considered in this study. Despite the existence of DOE-specific accident information[142], the risk factors for this screening risk model are determined in the same manner as for the Idaho Site short-term risk analysis (Schofield 2002) for the Idaho Site Subsurface Disposal Area (SDA) and a study (Hoskin et al. 1994) examining hazardous waste site remedial alternatives (including both excavation and capping) where U.S. Bureau of Labor Statistics (BLS 2007) information was used to estimate standard industrial risks.

The analysis of standard industrial risks is managed in a similar manner to that for exposure risks. Scenarios are defined to represent the most important characteristics of the standard industrial risks that confront workers during routine or remedial actions. For example, the direct and support worker scenarios for non-remedial activities are the same as those for exposure risks. However, during remedial actions, the worker scenarios for *both* exposure and standard industrial risks are characteristic of the remedial activities performed. Only a single type of direct or support worker (i.e., routine or remedial) is represented in the model. Additional scenarios may be added to better characterize the variation in standard industrial risks during the lifetime of the buried waste site.

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[142] The Computerized Accident Incident Reporting and Recordkeeping System (CAIRS), available at http://www.hss.energy.gov/CSA/Analysis/cairs/ (accessed March 13, 2008), is used to track *injury and illness* information for DOE sites. However, CAIRS is not available to the general public nor did it appear to lend itself to the type of analysis needed to provide the risk factors *for this study*. Despite various claims concerning DOE workplace safety versus other sites, such comparisons are difficult to make. Because of these difficulties, one former member of the DNFSB stated that "one conclusion one can reasonably draw from these statistics is that working in DOE facilities appears to be no more of an industrial risk to workers that [sic] employment in other industrial sectors of our economy" (Dinunno 2002).

Process Steps for Potential Remedial Activities

However, standard industrial risk estimates can be made more accurate by defining risk factors that are most closely aligned with the types of activities the worker is involved. Potential remedial alternatives for the buried waste sites can be conceptually divided into either 1) managing buried wastes in-place or 2) retrieving the wastes for treatment and disposal elsewhere[143]. These alternatives represent combinations of process steps that form the basis for estimating worker risks as described in Table 12 and Table 17 in Chapter IV (Brown et al. 2005; Schofield 2002; Zitnik et al. 2002).

For each process step defined in Chapter IV, a task or set of tasks likely to dominate worker risks was selected for use in the screening risk tool[144]. These tasks are

0. **Routine Work**—perform routine tasks associated with day-to-day operations;
1. **Burial Site Characterization**—characterize buried waste site, estimate extent of contaminant migration, determine the ability to target *high-risk* wastes, and mobilize personnel and equipment for remedial action (Schofield 2002)[145];
2. ***In Situ* Grouting (ISG) for Subsurface Stabilization**—grout burial site areas to stabilize subsurface areas prior to surface barrier installation[146];
3. **ISG for Subsurface Stabilization and Contaminant Immoblization**—grout burial site areas to stabilize subsurface areas and to treat (i.e., immobilize or "fix") subsurface contamination prior to surface barrier installation where additional risks are posed to workers if ISG is used for both purposes as indicated in Appendix A and Appendix B;

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[143] As noted in Appendix D, this conceptual division oversimplifies potential remedial alternatives. Only certain *high-risk* wastes may be targeted for retrieval, and some waste may be disposed of back in the original burial site.

[144] This simplification was deemed warranted because of the lack of specific worker risk information that would be required to describe possibly 60 different tasks associated with a remedial alternative.

[145] For the Bear Creek Burial Grounds (BCBG), there are areas that will not be directly sampled because of both the high costs potentially involved and the hazards associated with unstable, explosive, and pyrophoric materials buried (SAIC 1996a; b).

[146] GoldSim *Source* elements can only be used if the inventory is entered when the simulation begins (i.e., at time zero). GoldSim *Cell Pathway* elements were designed as "disposal units" to hold the wastes after grouting. Treated wastes are conceptually moved so that the impact of grouting can be represented.

4. **Excavate, Retrieve, and Segregate Buried Wastes: Excavate Soil Overburden and Store Soil**—remove the soil (likely contaminated) covering the targeted areas of the buried waste site and place the contaminated soil in temporary storage to act as the underburden for any retrieved wastes that can be placed back in the original burial location (Austad et al. 2003);

   Retrieve Wastes from Selected Areas—retrieve buried wastes (and collocated soil) from the waste areas identified in the *Burial Site Characterization* step, segregate the wastes, if needed, into transuranic (TRU), non-TRU, and metallic wastes, and store the wastes temporarily for subsequent treatment and disposal;

   Excavate Soil Underburden—remove the soil lining the bottom of the retrieval areas of the buried waste site, if present, and place the contaminated soil in temporary storage with the soil from the above *Retrieve Wastes from Selected Areas* step;

5. ***Ex Situ* Treatment**—treat retrieved wastes and soil using compaction for SDA wastes (Zitnik et al. 2002) and calcining for pyrophoric BCBG wastes[147] (Holdren et al. 2006) and temporarily store the treated waste prior to either on- or off-site disposal;

6. **Package Retrieved Wastes and Soil**—package the treated wastes for subsequent disposal where the Non-TRU wastes are placed in B-25 bins, TRU waste and soil composite material and metallic wastes, if present, are placed in 55-gallon drums that are then placed in TRUPACT-II containers (Schofield 2002);

7. **Intermediate Storage of Wastes and On-Site Disposal: Internment of Soil Overburden as "New" Underburden**—transport the original contaminated soil overburden to the burial site to act as the underburden for the closed burial site (Austad et al. 2003);

   Return Non-TRU and Non-HLW Wastes to Burial Site—return treated and packaged wastes that are neither TRU nor high-level wastes to the original burial site in an excavated waste area that is not prone to inundation or shallow groundwater flow;

   Place Clean Soil Overburden—place enough clean soil to fill excavated areas and a clean soil overburden on the excavated burial site (where the new overburden is assumed to be the same depth as the original layer);

8. **Surface Barrier Preparation and Emplacement**—install a surface barrier over the entire burial site assuming that the barrier can be installed in three sections (regardless of whether an evapotranspiration or RCRA Subtitle 'C' cap);

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[147] The pyrophoric wastes that would be retrieved from the BCBG would be treated in the same method described for pyrophoic uranium wastes at the Rocky Flats Plant (Holdren et al. 2006).

9. **Long-term Stewardship Activities: Monitor, Maintain, and Minor Repair of Cap**—monitor, maintain, and repair minor cap failures; these activities are seen extensions of characterization activities, and it is assumed that routine maintenance prevents the need for making requirements change with time; and

10. **Off-Site Shipment and Disposal at WIPP**—transport necessary wastes to the appropriate off-site area for final disposition assumed to be TRU wastes transported and disposed of at the Waste Isolation Pilot Plant (WIPP). Significant amounts of TRU wastes are not expected from BCBG retrievals.

The manner in which process steps are assembled for the baseline conditions and potential remedial alternatives is shown in Table 46. The characteristic remedial worker scenarios are also provided in this table where North American Industry Classification System (NAICS) categories are listed. Remedial workers are exposed to additional standard industrial risks (when compared to other site workers). There will also be additional exposure pathways including those related to unstable, explosive, and pyrophoric materials (for the BCBG) or high radiation fields and potential criticality events (for the SDA).

To account for increased risks to workers associated with the unstable, explosive, and pyrophoric materials that may be unearthed in the BCBG during characterization, excavation, or retrieval operations, the injury risk factor is used for fatality risk and total recordable cases is used for the injury risk when there is possible exposure to pyrophoric or unstable materials. The rationale for this use is described in additional detail later in this Chapter. It is assumed that pyrophoric materials may be unearthed in all BCBG Waste Areas as described in Appendix D.

Table 46. Process Steps, Characteristic Worker Scenarios, and Additional Exposure Hazards for Potential Remedial Alternatives

Process Step	Baseline	Manage-in-Place	Retrieval	Arid Direct Worker (NAICS)[a]	Arid Support Worker (NAICS)[a]	Humid Direct Worker (NAICS)[a]	Humid Support Worker (NAICS)[a]	Humid Additional Hazards[b]	Arid Additional Hazards[b]
0. **Routine Work**	√			56299	561	56299	561	---	---
1. **Burial Site Characterization**	√	√	√	56299	561	56299[c]	561	A_x,I_{py}	---
2. ***In Situ* Grouting (ISG) for Subsidence Control**		√	√	213111	561	213111[c]	561	A_x,I_{py}	R_{cr}
3. **ISG for Subsidence Control and Immobilization**		√	√	213111	561	213111[c]	561	A_x,I_{py}	R_{cr}
4. **Excavate, Retrieve & Segregate** Excavation of Soil Overburden			√	562212	48411	562212[c]	48411	A_x,I_{py}	R_{hr}
Retrieval of Buried Wastes			√	562212	561	562212[c]	561	A_x,I_{py}	R_{hr}
Excavation of Soil Underburden			√	562212	48411	562212	48411		
5. ***Ex Situ* Treatment**			√	332116	561	3272[c]	561	A_x,I_{py}	R_{cr}
6. **Package Retrieved Wastes**			√	56221	561	56221	561	---	---
7. **Storage and On-Site Disposal** Internment of Soil Overburden			√	562212	48411	562212	48411	---	---
Return Non-TRU/Non-HLW Wastes to Burial Site			√	562212	561	562212	561	---	---
Place Clean Soil Overburden			√	562212	48411	562212	48411	---	---
8. **Surface Barrier Installation/Repair**		√	√	562212	48411	562212[c]	48411	A_x,I_{py}	---
9. **Long-term Stewardship** Monitor, Maintain, and Repair	√	√	√	56299	561	56299	561	(d)	(d)
10. **Off-Site Shipment and Disposal at WIPP**[e]			√	---	---	---	---	---	---

a. *North American Industry Classification System* (available at http://www.naics.com/). Descriptions and distributions associated with the codes are provided in Table 47. Support workers for those activities involving soil transport are transportation-based. It is assumed that clean soil is available locally.

b. These columns represent additional exposure pathways during remedial activities for the humid and arid prototype sites. For the Bear Creek Burial Grounds (BCBG), accidents involving pyrophoric or unstable compounds present a potential explosive (A_x) or inhalation (I_p) hazard to remedial workers. High radiation (R_{hr}) fields (from either concentrated contaminants or highly radioactive materials) may be uncovered in the Subsurface Disposal Area (SDA) at the Idaho Site. Also, because of the large amounts of fissile material in the SDA, treatment in the form of compaction or contaminant movement during grouting might produce a potential criticality (R_{cr}) concern.

c. Because of the nature of the risks associated with unstable and pyrophoric wastes in areas in the BCBG, the total recordable case factor is used for the injury risks and the injury risk factor is used for fatality risks as described in the text. For BCBG characterization and barrier installation, this increase in risk is only applicable when wastes (i.e., unstable and pyrophoric) in the area have not been retrieved or, in the case of characterization, wastes will not be retrieved (i.e., Manage-in-Place remedial alternative).

d. These additional hazards are associated with long-term exposure affects.

e. The standard industrial and exposure risks to workers and the general public for shipment of transuranic (TRU) wastes from the burial sites to the Waste Isolation Pilot Plant (WIPP) are managed on a *per-shipment basis* using information from the WIPP Supplemental Environmental Impact Statement (USDOE 1997). Significant amounts of TRU wastes are not expected from the BCBG.

For example, a remedial worker involved with the excavation and handling of wastes would be at an increased risk when compared to a typical support or office worker over the same period of time on the same site. As a first approximation for screening purposes, worker categories (with NAICS[148] industry codes in parentheses) were defined based on U.S. Bureau of Labor Statistics (BLS) descriptions for various industries (BLS 2007)[149]:

- *Administrative and support and waste management and remediation services/Administrative and support services* (NAICS 561)—this category is considered to represent background support worker risks during remedial actions at a buried waste site;

- *Remediation and other waste management services/All other waste management services* (NAICS 56299)—this category was used to represent hazards to direct worker site characterization and long-term stewardship activities because other categories are considered either too general or specific;

- *Support activities for mining/Drilling oil and gas wells* (NAICS 213111)—there was no closely related category for *in situ* grouting; however, some of the same hazards might be experienced by direct workers during drilling operations;

- *Waste treatment and disposal/Solid waste landfill* (NAICS 562212)—although only a single year of data was available for this category, it was deemed important to use this category because it most closely describes the activities and hazards that might be experienced by direct workers for these steps;

- *Truck transportation/General freight trucking, local* (NAICS 48411)—for those activities involving soil transport, the risk factors for support workers are transportation-based assuming that operations are local and that clean soil is available locally;

- *Forging and stamping/Metal stamping* (NAICS 332116)—Idaho Site SDA *ex situ* treatment activities are assumed to be compaction; the selected category was considered to most closely represent the hazards to a direct worker;

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[148] The *North American Industry Classification System* is available at http://www.naics.com/ (accessed March 13, 2008).

[149] Available data are given by industry and not activity; therefore, these statistics apply to a given type of worker (e.g., direct versus support) and cannot be used for specific types of activities (e.g., sedentary versus light activity).

- *Nonmetallic mineral product manufacturing/Glass and glass product manufacturing* (NAICS 3272)—Oak Ridge BCBG *ex situ* treatment activities are assumed to be calcination; the selected category was considered to most closely represent the hazards to a direct worker; and

- *Waste management and remediation services/Waste treatment and disposal* (NAICS 56221)—this category was considered the most appropriate for waste packaging activities after treatment because any packaging activities would likely by captured in this category and more appropriate categories were identified for related activities.

For non-remedial activities, construction workers were considered representative of the industries in the available BLS data. On the other hand, waste management activities were considered the representative for remedial workers, especially considering the nature of the activities and the hazards involved.

The general injury and fatality risk factors used in the screening risk model are described in Table 47. These factors were computed from available BLS Statistics data across the United States because sufficient specific data by state or locale were not available. Because of a revision made in Occupational Safety and Health Administration (OSHA) requirements for recording occupational injuries and illnesses, comparable data are only available from 2003 to 2006 (BLS 2007). If pertinent U.S. Department of Energy (DOE) or site-specific data become available, the risk factors used in the model should be revised to reflect more accurate information.

Table 47. Injury and Fatality Risk Factors per Industry used in the Screening Model (Computed from Bureau of Labor Statistics Data (BLS 2007))

	Category	NAICS Code[a]	Probabilistic[b]	Point-Value[c]	Comment
Injury (Upper) and Fatality (Lower) Risk Factors	Administrative & support services	561	LN2($1x10^{-5}$,15%) LN2($2x10^{-8}$,10%)	$1x10^{-5}$ hr^{-1} $2x10^{-8}$ hr^{-1}	Describes background remedial support worker risks.
	All other waste management services	56299	LN2($5x10^{-5}$,40%) LN2($2x10^{-7}$,10%)	$9x10^{-5}$ hr^{-1} $2x10^{-7}$ hr^{-1}	Use for hazards to direct workers for characterization and long-term stewardship.
	Drilling oil & gas wells	213111	LN2($4x10^{-5}$,25%) LN2($3x10^{-7}$,10%)	$6x10^{-5}$ hr^{-1} $3x10^{-7}$ hr^{-1}	Use for *in situ* grouting hazards for lack of better category.
	Solid waste landfill	562212	LN2($5x10^{-5}$,25%) LN2($9x10^{-8}$,30%)	$8x10^{-5}$ hr^{-1} $1x10^{-7}$ hr^{-1}	Used because describes excavation and retrieval activities with RSD from #56221.
	General freight trucking, local	48411	LN2($4x10^{-5}$,15%) LN2($1x10^{-7}$,15%)	$6x10^{-5}$ hr^{-1} $2x10^{-7}$ hr^{-1}	Use for transportation-based support worker risks.
	Metal stamping	332116	LN2($5x10^{-5}$,10%) LN2($3x10^{-8}$,45%)	$5x10^{-5}$ hr^{-1} $5x10^{-8}$ hr^{-1}	Use for SDA *ex situ* treatment (compaction) with RSD from #3272 for fatality risk factor
	Glass & glass product manufacturing	3272	LN2($5x10^{-5}$,15%) LN2($4x10^{-8}$,45%)	$5x10^{-5}$ hr^{-1} $7x10^{-8}$ hr^{-1}	Use for BCBG *ex situ* treatment (calcining).
	Waste treatment & disposal	56221	LN2($5x10^{-5}$,25%) LN2($7x10^{-8}$,30%)	$7x10^{-5}$ hr^{-1} $1x10^{-7}$ hr^{-1}	Use for packaging activities for lack of better information.
High-Risk Factors	All other waste management services	56299	LN2($8x10^{-5}$,45%) LN2($5x10^{-5}$,40%)	$1x10^{-4}$ hr^{-1} $9x10^{-5}$ hr^{-1}	For hazards to direct workers during characterization & long-term stewardship in areas with pyrophoric & unstable wastes.
	Drilling oil & gas wells	213111	LN2($7x10^{-5}$,25%) LN2($4x10^{-5}$,25%)	$1x10^{-4}$ hr^{-1} $6x10^{-5}$ hr^{-1}	For *in situ* grouting hazards for lack of better category in areas with pyrophoric and unstable wastes.
	Solid waste landfill	562212	LN2($9x10^{-5}$,15%) LN2($5x10^{-5}$,25%)	$1x10^{-4}$ hr^{-1} $8x10^{-5}$ hr^{-1}	Describes excavation and retrieval activities for areas with pyrophoric & unstable wastes (RSD from #56221).
	Glass & glass product manufacturing	3272	LN2($9x10^{-5}$,10%) LN2($5x10^{-5}$,15%)	$1x10^{-4}$ hr^{-1} $5x10^{-5}$ hr^{-1}	For BCBG *ex situ* treatment (calcining) for areas with pyrophoric and unstable wastes.

a. *North American Industry Classification System* (available at http://www.naics.com/).

b. Only four years (2003-2006) of comparable data are available (BLS 2007). Summary statistics from the "Cases with days away from work, job transfer, or restriction" category were used to define a log-normal distribution, LN2(mean, %rel stdev), for the injury risk factors. For fatality risks, the "total fatalities" data (which include transportation accidents) are used to define risk factors. The maximum reported value (for four years) was selected as the mean and a liberal percent standard deviation (based on that obtained from the reported values) is used to represent the variation in risk factor.

c. The point-value is the relevant upper or lower 95%-quantile value unless otherwise indicated. Appropriate units are provided with this value. The units are actually $(person\text{-}hr)^{-1}$.

Workloads for Potential Remedial Activities

The best information available for the work loads needed to perform remedial actions for either prototype site was the feasibility study provided for the maximum retrieval case for the Idaho Site SDA (Schofield 2002; Zitnik et al. 2002). This site is larger than the Oak Ridge BCBG site, for which no comparable feasibility study documents were located. The basic approach to define work loadings for remedial actions is to first develop a basis set of work load information for the SDA maximum retrieval case (Schofield 2002) and then to assume, for screening purposes, that work loads and process step durations for areas within the SDA or BCBG can be scaled using relative volume for excavation- and retrieval-related activities and relative area for other activities (Eide and Wierman 2003).

The basis set of workload information developed from the SDA maximum retrieval case (Schofield 2002) is provided in Table 48. The duration information was developed either from SDA feasibility study (Zitnik et al. 2002) or based on judgment. Implementation and interpretation were simplified by sequencing the process steps so there was no overlap in operations or risk results. The numbers of direct workers needed were based on either the short-term risk analysis for the SDA (Schofield 2002) or judgment. Additional information on how the workload information was derived is provided in the notes at the bottom of Table 48.

Estimating the Probability of Injury and Fatality for Remedial Actions

The overall injury or fatality rates for the direct and support worker scenarios evaluated in the screening risk model were developed in previous sections. However, the likelihood of an injury or fatality is related not only to the time spent in the work area, but also the number of workers involved in the activities. The Poisson distribution is often used to model accidents because the number of accidents (i.e., "failures") is much smaller than the number of opportunities for accidents (i.e., "trials") (Hoskin et al. 1994).

Table 48. SDA Basis Work Loads and Durations Used in the Screening Risk Model

Process Step	Original Duration[a] (months)	Basis Duration[a] (month)	Min. Duration[a] (months)	SDA Direct Worker Code[b]	Direct Workers	SDA Support Worker Code[b]	Support Workers[c]
1. **Burial Site Characterization**	6	18	6	56299	15	561	15
2. ***In Situ* Grouting (ISG) for Subsidence Control**[d]	18	18	6	213111	10	561	10
3. **ISG for Subsidence Control and Immobilization**[d]	48	48	12	213111	10	561	10
4. **Excavate, Retrieve, and Segregate** Excavation of Soil Overburden	18	18	6	562212	50	48411	50
Retrieval of Buried Wastes	180[e]	180	60	562212	50	561	50
Excavation of Soil Underburden	18[f]	18	6	562212	50	48411	50
5. ***Ex Situ* Treatment**	192	48	36	332116	25	561	25
6. **Package Retrieved Wastes**	192	48	36	56221	25	561	25
7. **Storage and On-Site Disposal** Internment of Soil Overburden	18	18	6	562212	50	48411	50
Return Non-TRU Wastes to Burial Site	204	48	16	562212	50	561	50
Place Clean Soil and Overburden	18	18	6	562212	50	48411	50
8. **Surface Barrier Installation**	36	36	24	562212	50	48411	50
9. **Long-term Stewardship** Monitor, Maintain, and Minor Repair of Cap	(g)	(g)	(g)	56299	10	561	10
10. **Off-Site Shipment and Disposal at WIPP**[h]	---	---	---	---	---	---	---

a. Zitnik et al. (2002) suggest that steps 4-7 would require approximately 17 years (204 months) to complete concurrently. Implementation is simplified by sequencing the steps and "scaling" durations as needed for serial implementation. The retrieval step is not shortened (approximately 15 years or 180 months), but the other major steps (i.e., 5-7) were shortened significantly by increasing capacity. Each step is assumed to require approximately one-fourth of the original duration, or 48 months. The resulting total duration is longer than the original 17 years but is relatively short when compared to the assessment period of 1,000 years. The remaining basis durations are developed from either SDA remedial investigation information (Schofield 2002; Zitnik et al. 2002) or judgment. The characterization duration appeared insufficient for such a complicated site and was increased appropriately. Minimum durations are based on judgment. The number of direct workers required are either developed from information in Schofield (2002) or judgment. Workloads are currently not treated stochastically in the screening risk tool.

b. *North American Industry Classification System* (available at http://www.naics.com/).

c. It is assumed that one support worker is required for each direct worker (Eide and Wierman 2003).

d. The ISG duration was based on grouting 25% of the untreated SDA areas for subsidence control at a rate of 100 holes/day and a coverage of 0.4 m^2 (4 ft^2) per hole (Schofield 2002). For screening risk purposes, the entire area was assumed untreated. ISG for both subsidence control and contaminant immobilization was assumed to require coverage of three times the area for subsidence control alone.

e. This duration was based on a 76 m^3/day retrieval rate assuming 200 days worked/yr (at four 10-hr shifts per week) (Zitnik et al. 2002).

f. The same amount of time is assumed to be required for underburden removal as for the soil overburden.

g. These steps are treated as routine (i.e., non-remedial) operations.

h. Standard industrial and exposure risks to workers and the general public for shipment of TRU wastes from the original burial sites to the Waste Isolation Pilot Plant (WIPP) are predicted on a *per-shipment basis* using information from the WIPP Supplemental Environmental Impact Statement (USDOE 1997).

For a Poisson distribution with mean $\mu = x{\cdot}p$ (i.e., x injuries or fatalities each with probability p), the probability, f(x), of exactly x injuries or fatalities is given by (Hoskin et al. 1994):

$$f(x) = \frac{e^{-\mu} \times \mu^{x}}{x!} \, . \qquad [29]$$

The probability of *at least* one injury or fatality is $f(x \geq 1) = 1 - f(0) = 1 - \exp(-\mu)$. Equation 29 is used in the screening risk model to estimate the probability of at least one injury or fatality based on worker characteristics (e.g., number of workers, time spent on-site, etc.) and the appropriate injury or fatality risk factor. As the number of workers, time spent on-site, or risk factor increases, so does the probability of injury or fatality.

Worker Risks during Routine Remedial Activities

Both routine and accident conditions must be considered when evaluating the worker risks associated with remedial actions. For routine operations, workers are assumed to be exposed to only external radiation; the proper use of personal protective equipment precludes direct exposures to radionuclides and hazardous contaminants during routine operations. However, a detailed analysis of dose rates for handling radioactive wastes and the resulting external radiation hazards to workers requires the use of Microshield or a similar assessment code. Such a detailed analysis is unwarranted for a screening risk analysis where the focus is on risk trade-offs and not absolute risk estimates. Furthermore, use of a dose assessment code does not lend itself to probabilistic assessment. A simpler approach to estimating external exposure risks during routine remedial operations was taken by using the results provided by Schofield (2002) for the Idaho Site SDA short-term risk evaluation.

External exposure risks for workers during routine remedial actions are estimated using the dose rate results from Schofield (2002) to center the dose rate distributions; bounding dose rates correspond to the 100 mrem/hr limit defined in the WIPP Final Environmental Impact Statement (FEIS)[150]. The numbers and types of workers and needed handling operations on per drum and per bin bases are used to estimate external exposure rates (e.g., in mSv/yr) for characteristic direct and support workers. Because no corresponding dose rate analysis was found for BCBG waste retrieval actions, the information from the SDA study is used for both sites.

The simple exposure analysis provides total effective dose equivalents (e.g., in mSv/yr) to the workers from external radiation but not equivalent risks. Morbidity and mortality conversion factors of $8x10^{-2}$ risk/Sv and $6x10^{-2}$ risk/Sv, respectively, are used to convert from total effective dose equivalents to corresponding risks (ISCORS 2002; USEPA 1999). These conversion factors are deemed appropriate for external sources of low linear energy transfer (LET) beta and gamma radiation. For radionuclides that are underestimated by these factors, risk should be underestimated by less that a factor of 3; whereas, risks for certain bone-seeking transuranic elements may be overestimated by a factor of 10 (ISCORS 2002).

[150] The WIPP FEIS is available at http://www.eh.doe.gov/nepa/eis/eis0026/0026toc.htm (accessed March 13, 2008).

Nonroutine Worker Risks during Remedial Activities

The processes involved with remedying buried waste sites may involve different or increased accident risks or exposures to radiation or chemical hazards. At the Idaho Site SDA, high radiation fields from airborne contaminants or spent fuel elements (or their research analogues) may be encountered during excavation and retrieval actions (Brown et al. 2005; Holdren and Broomfield 2004; Schofield 2002; Zitnik et al. 2002). At the Oak Ridge BCBG, unstable and pyrophoric materials were buried (SAIC 1996a)[151], and their retrieval could result in fire and explosion hazards as well as increased exposure hazards from airborne, respirable contaminants produced from the fire or explosion. A single, relevant example of an additional remedial hazard is implemented in the model to illustrate the potential impacts on accident and exposure risks to workers.

The additional accident hazard modeled for the SDA is fashioned after the worst-case scenario for the maximally-exposed retrieval worker developed for the feasibility study (Schofield 2002)[152]. During retrieval operations, a direct worker wearing an air-supplied hood with a protection factor of 10,000 inadvertently uncovers a large pocket of highly contaminated material resulting in the resuspension of contaminated particulates. For Pu-239, 10% of the inventory is assumed uncovered in the pocket of which 1% of this material is resuspended into an air volume of 27 m^3, and 1% of the resuspended

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[151] A concern is raised about the generation of toxic gases from BCBG areas; however, the gases are not identified (SAIC 1996a; b; c; d; e; f) and thus cannot be part of the screening risk analysis. For example, if cyanide were one of the gases generated, the results might be more hazardous than those analyzed.

[152] As indicated in Appendix D, there is a large quantity of fissionable material buried in the SDA. A criticality accident would seem possible. However, preliminary safety analyses (Abbott and Santee 2004; Abbott 2003; Santee 2003) for proposed remedial actions indicate that any conceivable criticality accident would have a frequency less than once in 10,000 years. The conclusion from these analyses is supported by SDA criticality analyses (Sentieri 2002; Sentieri 2003a; b; 2004), which indicate that criticality accidents are either extremely unlikely or not credible. No specific information is available on the nature of the spent or research fuel rods that were buried in the SDA so neither of these scenarios is examined in this research.

material is respirable (Schofield 2002). The annualized effective dose equivalent (*$TEDE_i$*) for the i[th] radionuclide under these worst-case conditions is:

$$\mathrm{TEDE_i}\left[\frac{\mathrm{Sv}}{\mathrm{yr}}\right]=\mathrm{C_i^{resp}}\left[\frac{\mathrm{Bq}}{\mathrm{m^3}}\right]\times\frac{1}{\mathrm{PF}}\times\mathrm{V_{br}}\left[\frac{\mathrm{m^3}}{\mathrm{hr}}\right]\times\mathrm{DCF_i^{inh}}\left[\frac{\mathrm{Sv}}{\mathrm{Bq}}\right]$$
$$\mathrm{C_i^{resp}}\left[\frac{\mathrm{Bq}}{\mathrm{m^3}}\right]=\sum\mathrm{C_i}\left[\mathrm{Bq}\right]\times f_{inv}\times f_{res}\times f_{resp}\times\frac{1}{\mathrm{V_{air}}[\mathrm{m^3}]} \quad [30]$$

where dose is a function of the respirable concentration, C_i^{resp}, protection factor (*PF*), volumetric breathing rate (V_{br}), and dose conversion factor (DCF_i^{inh}). There are analogous conversion factors for morbidity and mortality risks.

In Equation 30, the respirable concentration is related to the total inventory (ΣC_i) in the burial site using the inventory, resuspension, and respirable fractions, f_{inv}, f_{res}, and f_{resp}, respectively, and the volume of air (V_{air}) in which the particulates are resuspended. The annual dose obtained from Equation 30 will be very large[153]; however, the exposure duration for a trained worker is likely to be three minutes or less (Blaylock et al. 1995; Schofield 2002). Best professional judgment and the available information is used to define probability distributions for the parameters in Equation 30 for use in the screening risk model (Blaylock et al. 1995; Eide and Wierman 2003; Schofield 2002).

However, radionuclides are not the only potential hazard when buried wastes are unearthed. Volatile and semi-volatile organic compounds were also buried in both the SDA and BCBG. For these compounds, the maximum air concentration resulting from an evaporating (or spilled) liquid can be estimated from the partial pressure (assumed to be the vapor pressure, VP_i) of the component (Blaylock et al. 1995):

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[153] For an accident scenario involving Pu-239, Schofield (2002) estimated a worst-case total dose to an SDA remedial worker of 0.5 Sv (49.5 rem). Tthe analgous case using the screening risk model gave a total annual dose of 3.1×10^7 mSv/yr for all radionuclides translating into a lifetime dose of less than 0.2 Sv.

$$LCF_i^{max}\left[\frac{1}{yr}\right] = C_i^{max}\left[\frac{mg}{m^3}\right] \times \frac{1}{PF} \times V_{br}\left[\frac{m^3}{yr}\right] \times \frac{DCF_i^{inh}\left[\frac{kg \cdot d}{mg}\right]}{AT[d] \times BW[kg]}$$

$$C_i^{max}\left[\frac{mg}{m^3}\right] = 1000 \cdot M_i\left[\frac{mg}{mol}\right] \times \frac{VP_i[mmHg]}{R\left[\frac{m^3 \cdot mmHg}{K \cdot mol}\right] \times T[K]} \quad [31]$$

where the maximum air concentration is proportional to the vapor pressure (using the ideal gas law)[154]. The annual number of excess latent cancer incidences or fatalities (LCF) is obtained by multiplying the intake by the slope factor (DCF_i^{inh}) normalized by the averaging time (*AT*) and body weight (*BW*). An relationship analogous to Equation 31 was developed to describe the maximum non-carcinogenic impact to the direct remedial worker under the worst-case conditions defined by Schofield (2002).

The additional hazard considered for workers retrieving wastes from the BCBG comes from the large amounts of pyrophoric uranium buried at the site. Unlike the hazard considered for the SDA from unearthing a highly contaminated area, the pyrophoric uranium in the BCBG presents both exposure and physical hazards. The exposure risk is from radioactive uranium released into the air via the rapid oxidation of uranium metal with high specific surface areas. The large amount of energy released via rapid oxidation poses a physical hazard to remedial workers in the area. The exposure hazard is evaluated using Equation 30 and using judgment to define the parameters needed for the model[155].

[154] The available volatile organic inventory is examined in the model to assure that sufficient material exists to provide the maximum concentration from Equation 31 and is adjusted to this value if insufficient material is available.

[155] Despite a wealth of information pertaining to the release and respiration of depleted uranium from the use of munitions, similar information is not available for uranium fines.

A lack of information also exists concerning the frequency and possible consequences (i.e., standard industrial risks) for the physical hazards to remedial workers by the rapid oxidation of pyrophoric uranium metal unearthed during retrieval activities. In the screening risk model, standard industrial risks are evaluated using the relationship in Equation 28. However, risks due to events such as fires, explosions, etc. are included in overall risk factors and cannot be separated out by event. For lack of better information, the risk factors for those activities (e.g., characterization, excavation, etc.) that might unearth unstable or pyrophoric materials are adjusted to reflect the increased physical risks involved. The logic for redefining specific risk factors for the worst-case impacts of working in an area with pyrophoric and other unstable materials is:

- Assuming that some accidents leading to injuries without pyrophoric materials involved might instead result in fatalities, the extant injury risk factor is used to estimate direct remedial worker fatality risks during retrieval activities,
- Assuming that recordable incidents that result in neither injury nor fatality without pyrophoric or unstable materials present might result in at least an injury, the direct remedial worker injury risk factor for retrieval actions will be based on that for total recordable cases[156], and
- Because of their distance from the retrieval site, support worker standard industrial risk factors are not impacted by retrieval activities that might unearth pyrophoric materials.

Specific accident frequency and consequence data should be used whenever possible; however, risk factors defined using the above logic reflect the more dangerous nature of retrieving unstable and pyrophoric materials.

The accidental releases of material and potential worst-case impacts on remedial workers are computed in the model; however, contaminant transport is not impacted by

[156] The total recordable and injury case results are used "as-is" without attempting to account for possible double-counting of cases; this decision, which likely overestimates the injury risk factor for retrieval, is based on the *possibility* that additional cases might be reported if pyrophoric materials were involved.

these accident conditions. The dose and risk results obtained from these equations are treated analogously to limits or objectives, that is, primarily for comparison purposes. These potential impacts are hypothetical and maximum; such large impacts are not typically expected when retrieving buried wastes, but are useful in bounding expected impacts especially for planning purposes.

Simplified Retrieval and Handling Risk Evaluations

If retrieval activities are deemed necessary, large amounts of waste in various forms and contaminated soil will be removed from the site. The manner in which retrieved material must ultimately be dispositioned is evaluated using numerous factors including waste form, contaminant identities and concentrations, toxicity, radiation, test results, etc. Some waste may be disposed of on-site either in the original burial site or in an approved landfill. Other wastes must be transported off-site for disposal including the Waste Isolation Pilot Plant (WIPP) for transuranic wastes. Because contaminants and waste forms are intermixed in ways that cannot be known accurately (without prohibitive site characterization), the simple method provided by Schofield (2002) in the short-term risk evaluation for the SDA is used to segregate the waste inventory by disposal type.

Wastes retrieved from the SDA are segregated into either a transuranic (TRU) waste fraction that must be transported to WIPP for disposal or non-TRU waste and soil composite that can be disposed of on-site (Schofield 2002). None of the wastes retrieved from the BCBG would likely be classified as TRU wastes. The SDA TRU waste fraction for WIPP disposal is further subdivided into either a TRU waste and soil composite or a metallic fraction[157]. The retrieved and segregated wastes will be treated *ex situ* (using compaction in the SDA and calcination in the BCBG) and then packaged based on the assumptions and fractions provided by Schofield (2002). The non-TRU waste and soil composite is packaged in B-25 bins for disposal on-site is the original burial location[158]. The TRU waste and soil composite is placed in B-25 bins for storage whereas metallic TRU wastes are stored in 55-gallon drums. TRU wastes are packaged for disposal in 55-gallon drums and then placed in TRUPACT-II containers for shipment to WIPP[159].

Simplified On-Site and Off-Site Disposal Risk Evaluations

Two TRU waste streams result from the retrieval, treatment, and packaging steps. These streams are the TRU waste and soil composite material and the metallic TRU wastes that are stored in 55-gallon drums and placed in TRUPACT-II containers for shipment to and disposal in WIPP. A detailed analysis of the doses and risks associated with the disposal of TRU wastes retrieved from the burial site would require the use of Microshield or similar code (Schofield 2002). However, such a detailed dose analysis does not lend itself to neither a screening-level assessment nor probabilistic analysis. Because the focus of this screening analysis is the risk trade-offs involved in potential remedial alternatives, a simple analysis using information from the WIPP Supplemental

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[157] The total volumes (i.e., waste plus soil) associated with the SDA TRU pits and trenches has changed between the time of the short-term risk evaluation and the latest remedial investigation study (Holdren et al. 2006). The fractions of waste to soil will be used from Schofield (2002) with the most recent pit and trench dimensions to determine the associated soil volumes.

[158] A new disposal site and the risks involved are not considered in the screening risk tool. The sites considered already have active low-level waste disposal areas and thus returning the retrieved and treated wastes back to the original site, in essence, maximizes potential exposure risks.

[159] It is assumed that 14 55-gallon drums can be placed in a TRUPACT-II container. For metallic TRU wastes, it is further assumed that three TRUPACT-II containers can be transported per shipment. For the TRU waste and soil composite material (which is assumed to have a higher specific activity), two TRUPACT-II containers can be transported per shipment.

Environmental Impact Summary (SEIS) (USDOE 1997) is warranted that provides a reasonable basis for screening risks and risk trade-offs[160].

For the transport of wastes from DOE sites to WIPP, non-radiological impacts including accidents, injuries, fatalities, and pollution-related health effects are estimated on a per shipment basis (USDOE 1997). Although potential impacts depend on the total number of shipments, inventory, and treatment options, the estimated impacts using WIPP SEIS information are assumed to be representative of the TRU wastes from the prototype sites that must be disposed of in WIPP. The number of shipments required to transport the retrieved wastes to WIPP is calculated as well as the total time required for shipment based on the number of shipments from the site that can be processed at the WIPP (without storage at WIPP to simplify implementation). If a very large area from the site is slated for retrieval, then the number of and time to ship all the wastes to WIPP would likely be prohibitive[161].

Probability distributions are defined for many of the parameters describing the segregation, treatment, packaging, and off-site transport process steps to determine the impact of these parameters on the resulting dose and risk estimates. These parameter distributions are described in the screening risk model with corresponding bases or references. If these parameters are found to be significant from a risk perspective, then additional effort can be expended to better define the distributions.

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[160] The decision to use a simplified approach using information available in the WIPP SEIS (USDOE 1997) is reinforced by the fact that the WIPP site is already open and receiving TRU wastes from numerous sites and thus the impact on overall WIPP-related risks may be small.

[161] Schofield (2002) estimated that approximately 31,000 shipments would be required to transport the retrieved and treated TRU wastes from the Idaho Site Subsurface Disposal Area (SDA) to the WIPP.

Screening Risk Tool Verification

Although the screening risk tool is necessarily a highly abstracted representation of a burial site, the tool does incorporate a large number of interconnected exposure media, transport pathways, exposure routes, receptors, etc. which require copious amounts of data to implement even for screening purposes. In developing the model, numerous tests were run to verify that the model was performing as anticipated. A representative subset of the verification tests is provided in Appendix G for reference. For all runs, it is assured that mass is conserved[162] before risks results are used.

Because of the desire to model two very different types of DOE burial sites in a single, integrated model, the ability to select between sets of model parameters and to select desired transport mechanism was added to the model. The ability to control conditions and transport using the "switches" programmed into the model (i.e., Figure 33) was tested using primarily visual tests to confirm either that the desired parameters were being used[163] or that transport via the selected pathways was occurring as expected. Numerous numerical tests were also performed using MatLab, MathCad, and Microsoft Excel to assure that dose and risk computations were correct for given exposures.

Model Validation

The screening risk model provides the ability to evaluate life-cycle disposal risks for two very different hypothetical buried waste sites, one arid and one humid. The large uncertainties involved with evaluating potential impacts from an actual buried waste site

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[162] Because both radioactive decay and organic compound degradation are modeled, the model can be run with both of these processes disabled to verify mass conservation. However, the lack of mass conservation is frequently obvious even when radioactive and organic degradation are employed.

[163] The GoldSim software has built-in features allowing the data being used in the model to be examined.

make accurately predicting risks for the hundreds or thousands of years needed for regulatory compliance difficult, if not impossible. Furthermore, sites where contaminants move through the subsurface cannot be validated even if site data are available (Konikow and Bredehoeft 1992). Therefore, the query as to whether a given model provides accurate exposure or risk results is not appropriate when placed in the proper context.

The correct question for models developed for buried waste site evaluation should instead revolve around whether enough the relevant characteristics of the site, source term, transport, receptors, etc. are represented so that dose and risk results are reasonable for comparison. For example, the latent cancer incidence for a given exposure using a "slope factor" can in no way be considered an estimate of the true risk associated with the exposure. On the other hand, two risks computed in this same fashion for two remedial alternatives can be compared to each other (i.e., less exposure is "better"). However, comparisons based on different risk metrics (e.g., latent cancer incidence, accident injuries, hazardous chemical effects, etc.) must be made very carefully.

Comparison of Remedial Alternatives

The screening risk modeled implemented in GoldSim calculates a large number of values over the selected assessment period (e.g., 1,000 years) for contaminant fluxes from sources to receptors, contaminant concentrations in exposure media, and potential impacts to receptors. The model provides an incredible amount of information that must be processed to evaluate the risks and risk trade-offs for proposed remedial alternatives. Metrics are suggested for comparing remedial alternatives.

Although contaminant fluxes are the primary information needed to characterize transport from the buried waste site to potential receptors, fluxes (and corresponding media concentrations) are intermediates for estimating doses, risks, and hazards; these latter parameters are the metrics on which the screening analysis focuses. For example, the doses for different remedial alternatives can be compared like risks or hazards on the same bases. However, comparison of dose and risk is much more problematic although there are reasons to believe that radiation doses and risks, which are both functions of exposure to radionuclides, are highly correlated. When this is the case, the focus can be placed on radiation *risks* (with all due respect to the Nuclear Regulatory Commission).

Assuming that *risks* (e.g., morbidity and mortality) can be used to characterize radiation exposure impacts for screening risk purposes, three other types of *risks* are also computed in the model and may be compared—albeit only with great care. These risks are the carcinogenic risk from chemical exposure and the standard industrial injury and fatality risks for accidents involving workers. Whereas the basis for radiation exposure impacts is firm, it has been very difficult to link chemical exposures to specific human health impacts, especially at the low levels expected for human exposures.

Because models must be used to estimate health impacts at low chemical doses, it has been argued that the true health risk for a given low dose is as likely to be zero as that calculated from the low-dose model. On the other hand, worker injury and fatality risks for workplace accidents are based on the statistical analysis of historic information. Although there are large uncertainties in workplace risk estimates, the data used in the analysis represent actual, reported injuries and fatalities to real workers. Despite the potential to compare different risk estimates for remedial alternatives, the hazard indices for non-carcinogenic chemical exposures are not likely comparable to other risk metrics.

Screening Risk Model Evaluations

The primary basis for comparing risks and risk trade-offs for a buried waste site is the baseline risk analysis where no remedial actions are assumed to be taken. The baseline conditions represent the maximum expected *exposure* health effects to the general public that might be impacted by contaminants migrating from the site. Remedial alternatives for the buried waste site are grouped into two categories: 1) manage wastes in-place or 2) retrieve wastes for treatment and disposal elsewhere. Retrieval actions may be targeted on *high-risk* wastes reducing the footprint and impact of the remedial actions.

For each remedial alternative, the potential dose, risk, and hazard impacts on six characteristic receptors (i.e., four general public and two worker scenarios) are evaluated. The metrics that are evaluated for the six receptors over time include:

- *radiation* dose, morbidity risk, and mortality risk
- *chemical* carcinogen risk
- *chemical* non-carcinogen hazard index
- *standard industrial* injury and fatality risks

However, not all risk metrics are important for each receptor. For example, remedial workers wearing the proper personal protection equipment are not exposed to chemical hazards but would be impacted by external radiation. Standard industrial risks do not apply to the general public receptors. Although many transport pathways may impact receptors near and far from the buried waste site, impacts will be grouped into either those via the atmospheric pathway or all pathways to enhance understandability (Tauxe 2004). Figure 51 outlines the model runs that will be executed and information generated to evaluate remedial alternatives for the Idaho Site Subsurface Disposal Area (SDA) and Oak Ridge Bear Creek Burial Grounds (BCBG).

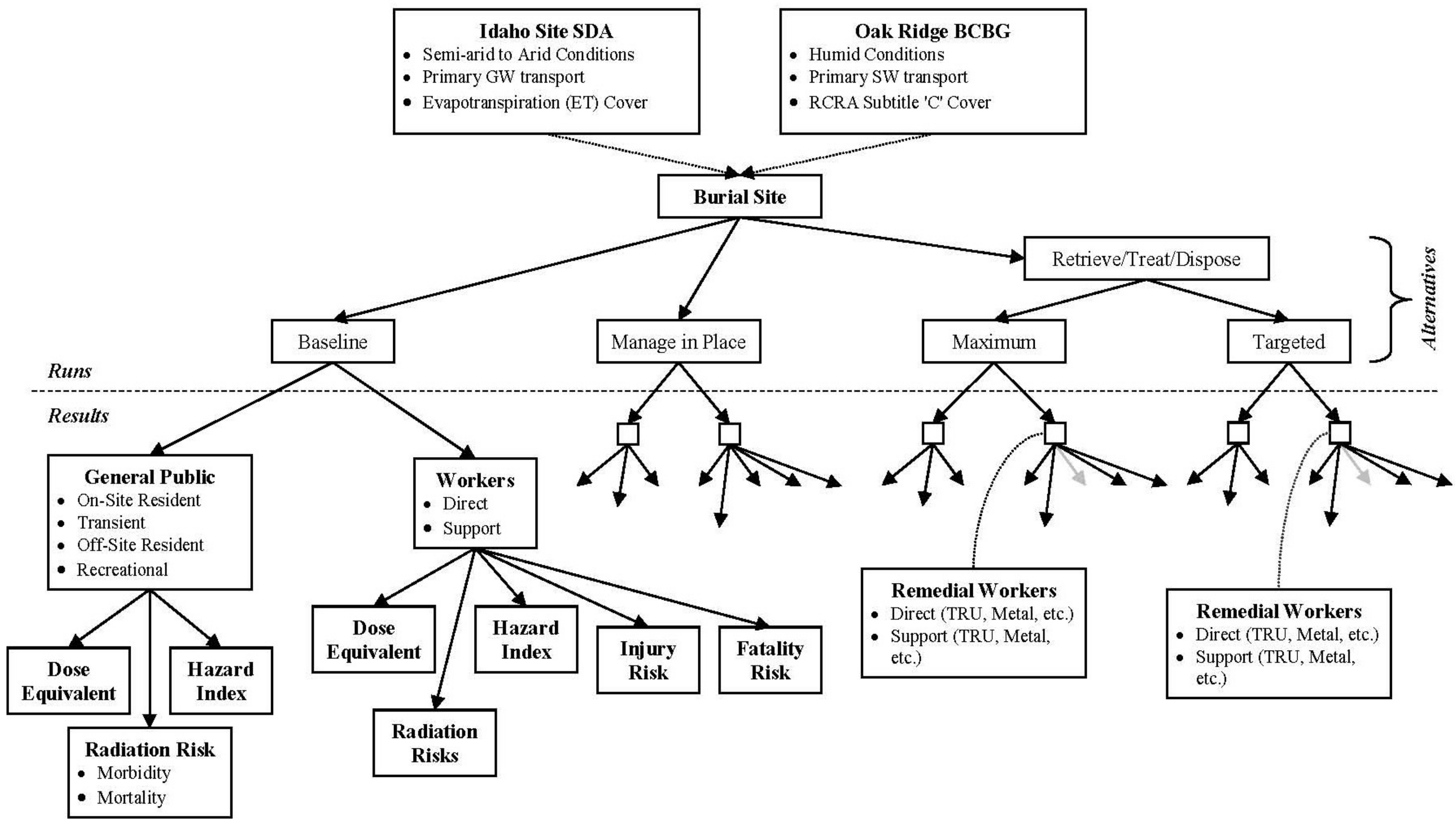

Figure 51. Model Runs and Results needed to Evaluate Remedial Alternatives

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CHAPTER VII

APPLICATION OF THE SCREENING RISK TOOL TO TWO DEPARTMENT OF ENERGY (DOE) BURIED WASTE SITES

A first-of-a-kind screening tool was developed to allow estimates of the risks associated with remedial actions for U.S. Department of Energy (DOE) buried waste sites to be predicted. The tool integrates the ability to evaluate both exposure and standard industrial risks for baseline conditions and remedial actions for the life-cycle of the buried waste site disposition. The broad nature of the source term, fate and transport, exposure, and receptor implementations allows typical baseline risk evaluations but is extended to consider life-cycle and remedial actions to the same level of detail.

The screening risk tool can be used to evaluate three quantitative phases of the risk analysis framework defined in Chapter III. Phase 2A, shown in Figure 9 (Chapter III), provides screening risk estimates for *baseline conditions* as a basis for comparison. Upon determination that the site requires remedial action and selection of alternatives, screening estimates of *remedial action* risks (in Phase 2D) and corresponding *residual risks* (in Phase 2B) can be evaluated using the screening risk tool[164]. Phase 2B and Phase 2D are illustrated in Figure 9 and Figure 10 (Chapter III), respectively.

In this chapter, the screening risk tool is used to evaluate exposure and standard industrial risks for two prototypic sites (i.e., the Idaho Site Subsurface Disposal Area and Oak Ridge Bear Creek Burial Grounds). The exposure and standard industrial risks are

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[164] Risks are estimated for those remedial actions required to place the waste in a protective state. For example, if transuranic (TRU) wastes are encountered and retrieved, these wastes must be transported to the Waste Isolation Pilot Plant (WIPP) for final disposal. Both on-site and off-site risks are included in the analysis performed using the screening risk tool.

first evaluated for baseline conditions and then for proposed remedial alternatives (described in Chapter IV). Remedial alternatives are classified as either 1) managing wastes in-place or 2) retrieving wastes for treatment and disposal. The results of the screening quantitative risk evaluations are compared to the qualitative risk evaluations described in Chapter IV (which summarizes the detailed analyses in Appendix A and Appendix B) and the CERCLA remedial investigations and feasibility study results generated for the sites (Holdren et al. 2006; Holdren et al. 2007; SAIC 1996a; e).

Prototype Site Descriptions

As described in Chapter IV, two DOE sites were selected for evaluation using the risk analysis framework defined in Chapter III. The prototype sites are the Idaho Site Subsurface Disposal Area (SDA) and the Oak Ridge Bear Creek Burial Grounds (BCBG) in Tennessee. These sites were selected because they bracket the types of contaminants, hazards, and conditions that are expected from DOE buried waste sites and should, therefore, demonstrate the effectiveness and flexibility of the approach defined in this research A brief description is provided for each site; the remedial investigation reports for the SDA (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) and the BCBG (SAIC 1996a; b; c; d; e; f) should be referred to for more detailed information for each.

Idaho Site Subsurface Disposal Area (SDA)

The SDA comprises a 97-acre area in the Idaho Site Radioactive Waste Management Complex (RWMC). Figure 13 and Figure 14 in Chapter IV describe the RWMC and SDA. Transuranic (TRU) wastes, received from the Rocky Flats Plant (RFP) near Denver, Colorado, were buried in the SDA before 1970 and stored retrievably (i.e., aboveground) in the RWMC after that. Other wastes including small amounts of other TRU-contaminated materials and large amounts of fission products and organic solvents were buried in the SDA. The wastes buried in the SDA are unique both in their magnitude and diversity. The wastes are diverse in the variety of contaminants (i.e., radioactive and hazardous) and how contaminants are intermixed.

Waste zone monitoring indicates that VOCs, plutonium isotopes, Am-241, and uranium isotopes have migrated from the original burial site (Holdren et al. 2006). Radionuclides have migrated into the vadose zone beneath the SDA including Tc-99, Am-241, Pu-239, Pu-240, Sr-90, and Pu-238. VOCs and nitrates have migrated to the sole-source Snake River Plain Aquifer (SRPA) underlying the SDA.

The contaminants of potential concern (COPCs) identified in the SRPA include carbon tetrachloride, trichloroethylene, uranium isotopes, and Cs-137. Non-COPC contaminants including tritium, sulfate, chloride, chromium, and toluene originating from the SDA buried wastes have also been detected in the aquifer. SDA contaminants including C-14, nitrates, Pu-238, Am-241, Pu-239, Pu-240, tetrachloroethylene, and methylene chloride have been detected intermittently in the SRPA.

The only contaminant whose measured concentration in the aquifer exceeds its Maximum Contaminant Level (MCL) is carbon tetrachloride (Holdren et al. 2006). The risks associated with COPCs for the SDA are provided in Table 9 (Chapter IV). There is an on-going probing project in the SDA to identify the extents of contamination and to reduce important uncertainties (Miller 2003; Salomon 2004).

Oak Ridge Bear Creek Burial Grounds (BCBG)

The BCBG are located within the Beak Creek Valley, an area mostly contained in the Oak Ridge Reservation (ORR) approximately 20 miles northwest of Knoxville, Tennessee. The valley is over 10 miles long and runs from the eastern end of the Oak Ridge Y-12 Plant to the Clinch River. Figure 15 and Figure 16 in Chapter IV show the ORR and BCBG, respectively. There are multiple waste units in the valley containing hazardous and radioactive wastes primarily from Y-12 Plant operations. Groundwater has been contaminated throughout at least the eastern 3 miles of the valley (SAIC 1996a).

At the BCBG, solid and liquid wastes were disposed of in a series of unlined trenches (SAIC 1996a). Uranium dominates the wastes disposed with a total mass of approximately 19×10^6 kg (40×10^6 lb). Liquid waste disposal resulted in volatile organic compound (VOC) contamination in groundwater reaching depths of 200 m (600 ft). Contaminants in the BCBG include VOCs and metals in groundwater and VOCs, metals, and radionuclides in surface water, soils, waste materials, and leachates. Organic contamination is more widespread than inorganic and radionuclide contamination.

Unlike the impacts from SDA contaminants (where the vadose zone is deep and effects may be delayed), the effects of BCBG contaminants have a more immediate impact on the surrounding environment and receptors. Impacts for the BCBG are via the surface water pathway versus those for the SDA that are likely to impact groundwater resources over long periods of time[165] (Holdren et al. 2006). Not that both buried waste sites do not contain very long-lived radionuclides and stable contaminants—they both do—it is merely that the temporal aspects of risk are very different for the sites. Peak

[165] As expected for a complicated site like the SDA, there are notable exceptions including VOCs that have reached the Snake River Plain Aquifer running under the site after "only decades" of having been buried.

risks associated with various contaminants originating in the BCBG based on measured or predicted maximum concentrations are provided in Table 10 in Chapter IV.

Screening Risk Analysis of the Idaho Site Subsurface Disposal Area (SDA)

The screening risk analysis tool described in Chapter VI is initially applied to the Idaho Site Subsurface Disposal Area (SDA). This site was selected for initial evaluation based on past experience from the development of a qualitative risk assessment for the SDA as requested by the DOE (Brown et al. 2005). CERCLA remedial investigation reports (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002) and feasibility studies (Holdren et al. 2007; Schofield 2002; Zitnik et al. 2002) are available for comparison purposes. The quantitative results obtained in this chapter will also be compared to the qualitative assessment results in Brown et al. (2005) and Chapter IV.

SDA: Screening Quantitative Baseline Risk Assessment

The initial analytical step for evaluating a buried waste site is to determine whether or not remedial action is required. A baseline risk assessment is performed to determine if risks posed to selected receptors from the contaminants at the site are greater than appropriate concentration, risk, or other pertinent limits. The limits originally considered for use in the BRA include:

- 0.10 mSv/yr (10 mrem/yr) from radiation—Air pathway dose objective[166] (DOE G 435.1 § IV.P. p. IV-185 (USDOE 1999) or 40 *CFR* 61.92 (1989))
- 0.25 mSv/yr (25 mrem/yr) from radiation—All pathways dose objective (DOE G 435.1 § IV.P. p. IV-185 (USDOE 1999) or 10 *CFR* 61.41 (1987b))

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[166] The air and total pathway dose objectives can be found in the Implementation Guide for the DOE Radioactive Waste Management Order, DOE G 435.1 § IV.P. p. IV-185 (USDOE 1999). An average radon flux limit at the surface of 0.74 $Bq/m^2/s$ is also provided in this guide but is not considered in his research.

- 1 mSv/yr (100 mrem/yr) from radiation—Radiation protection standard for public and workers (10 *CFR* 20.1301 (1987a) and DOE Order 5400.5 (USDOE 1990))
- 5 mSv/yr (500 mrem/yr) from radiation—Radiation protection standard for workers (occasional ICRP standard (USDOE 1990))
- 0.15 mSv/yr (15 mrem/yr) from radiation—Proposed EPA standard to correspond to 10^{-6}/yr carcinogenic risk (Luftig and Weinstock 1997)
- 10^{-4} cancer risk—EPA CERCLA risk standard for site cleanup (CFR 1994)
- 10^{-6} cancer risk—EPA CERCLA *de minimus* risk standard (CFR 1994)
- 1—EPA Noncarcinogenic risk standard for chemicals (CFR 1994)

Because of the purpose of the screening BRA is to determine whether or not a contaminated site poses an unacceptable risk, the most restrictive of the above criteria are used. The selected criteria for the baseline risk assessment are:

- 0.15 mSv/yr (15 mrem/yr) from radiation from all pathways
- 0.10 mSv/yr (10 mrem/yr) from radiation from air pathway
- 10^{-6} cancer risk/70 years—EPA CERCLA *de minimus* risk standard divided by the conventional EPA lifetime exposure duration (USEPA 1989)[167]
- 0.1/1—Based on EPA Noncarcinogenic risk standard for chemicals[168]

As there are different limits that might be used to define "acceptable" dose or risk, there are also different scenarios that can be used to define the doses or risks associated with buried wastes and their disposition. The typical method used for CERCLA sites is to examine the reasonable maximum exposure to a member of the general public (USEPA 1989). The reasonable maximally exposed individual might be an on-site resident, off-site resident, recreational user, or combination (Holdren et al. 2006).

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[167] The *upper bound* risk is obtained by dividing the EPA 10^{-4} cancer risk standard for site cleanup by the national median time (50^{th}-percentile) exposure duration at one residence of 9 years (USEPA 1989).

[168] A value of 1/10 will be used for hazard quotient (HQ) screening purposes because of uncertainties in the HQ determination and the fact that the relative magnitude of HQ does not represent relative risk of effect.

The focus for the baseline risk assessment is to determine whether or not current or future site conditions pose unacceptable risks to important receptors. The simplifying assumption is made, *for this research*, that the site was abandoned at the time of burial and the "reasonable" maximal exposure corresponds to that for the *on-site resident during the entire assessment period*. In fact, the sites were not abandoned and residents will not be permitted to live on-site until after the Institutional Control (IC) period has expired. However, use of the on-site resident scenario maximizes predicted risks as illustrated in Figure 52 and simplifies the basis for comparison between various remedial alternatives as well as between sites. Exposures and risks for other receptors are computed in the screening risk tool and be used to define acceptability if needed.

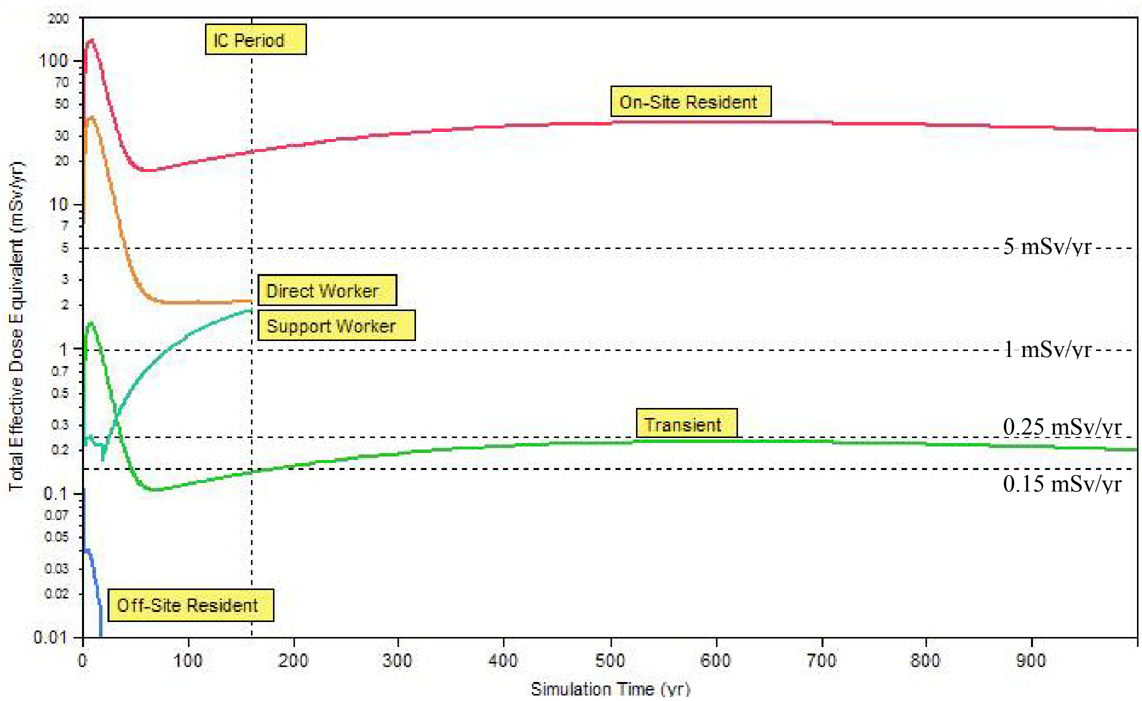

Figure 52. SDA General Public and Worker Scenarios from Chapter VI: Baseline Annual Total Effective Dose Equivalent (TEDE) for All Pathways Summed over all Radionuclides and Compared to Various Dose Limits

One advantage of using the screening risk tool developed in this research is the manner in which it can lend consistency and transparency to the site disposition analysis. The screening risk tool can be used to determine whether a buried waste site poses excessive risk and, if so, which contaminants may be of concern from a risk perspective using either a point-value or probabilistic analysis. Remedial actions can then be assessed for their potential effectiveness in reducing site disposition risks. The evaluations are performed using a consistent set of models, assumptions, scenarios, etc. that can be updated if more accurate results are warranted for the remedial decision.

A number of assumptions were made concerning the potential source release and transport pathways that also impacted the degree of exposure to receptors. The major source and transport pathway assumptions that impacted the results from the screening risk tool developed in this research for buried waste sites included:

- All wastes were buried at a single time instead of distributing burials over time.
- The contaminant source releases were controlled by the surface wash, dissolution, and diffusion mechanisms as modeled in Appendix E.
- The complex interactions of contaminants in the subsurface could be described using a simple linear partitioning (i.e., K_d-based) retardation model.
- The maximum concentrations of contaminants in the aqueous phase were independent and could be described using one solubility value for each.
- The position of the drinking water well intake in relation to the direction of flow did not substantially change the concentration in the drinking water. Although available in the screening risk tool, the GoldSim *plume function* (GTG 2005a) was not employed in this research because of the lack of information needed to define its ten parameters. The effect of using the plume function would be to reduce contaminant concentrations in the drinking water.
- The atmospheric and soil pathways including the vadose zone could be approximated using simple "box" models using GoldSim *Cell Pathway* elements.

The additional assumptions made in developing the screening risk tool that significantly impact risk predictions are described in the tool where they were made.

The assumptions have major impacts on predicted exposures and should be evaluated when considering whether a site poses an unacceptable risk. Because a baseline risk assessment (BRA) is used to determine if a site might pose an unacceptable risk, values are selected to represent the assumptions made in such a way as to maximize predicted exposure. The impacts of the assumptions made are explored in this chapter.

Simulations were run to determine whether a site might pose unacceptable risks. Point-value exposure, dose, and risk calculations are made using parameters representing best estimate and reasonable maximally exposed conditions to evaluate site acceptability in accordance with EPA guidance (USEPA 1989). If the site posed an unacceptable risk, then additional runs were used to identify contaminants of potential concern (COPCs), which require remedial action. The point-value simulations performed as input to the screening BRA and to the evaluation of the impacts on risk are described in Table 49.

The *DBRA-Expected* case provides point-value dose, risk, and hazards results for the best estimate and expected (i.e., 50^{th}-percentile) values for inventory and transport parameters employing all waste form release mechanisms and transport pathways. The total *dose* result (i.e., sum of annual total effective dose equivalents (TEDEs) in mSv/yr for all radionuclides) as a function of simulation time is represented in Figure 53. As illustrated in this figure, the total dose exceeds any of the limits defined above for use in this research by more than an order of magnitude. Therefore, the site contaminants pose unacceptable risks on a dose basis and remedial actions will be required for the SDA to place the site in a protective state.

Table 49. SDA Deterministic 1,000-yr Baseline Risk Assessment (DBRA) Simulations

Designation	Description[a]
DBRA-Expected	Baseline conditions with inventory and stochastic elements set to best or expected (i.e., 50th-percentile) values, respectively. Inventories are segregated by waste form and in containers (if applicable). All source release and transport mechanisms are in effect *except for organic degradation*[b]. Colloids are assumed screened by the Interbed Region.
DBRA-ExpLoose	Baseline conditions with inventory and stochastic elements set to best or expected (i.e., 50th-percentile) values, respectively. Inventories are assumed to be loose and not associated with waste forms. All transport mechanisms are in effect *except for organic degradation*[b].
DBRA-Maximum	Baseline conditions with inventory and stochastic elements set to their respective 95th-percentile upper or lower values depending on the estimated risk impact described in Chapter VI. Inventories are segregated by waste form and in containers (if applicable). All transport mechanisms are in effect *except for organic degradation*[b].
DBRA-WorstCase	Baseline conditions with inventory and stochastic elements set to their respective 95th-percentile upper or lower values depending on the estimated risk impact as described in Chapter VI. Inventories are assumed be loose and not associated with waste forms. All transport mechanisms in effect *except for K_d-based retardation, solubility, and degradation*[b]. Colloids are not screened by the Interbed Region.

a. Maximum resuspension as described in Tauxe (2004) was not used for any simulation in this research. The runoff and inundation pathways do not apply to the SDA as described in Chapter VI. Colloids were assumed to be screened by the Interbed Region except for the *DBRA-WorstCase* simulation.

b. It was decided that organic degradation would either be used for all cases and affected compounds or not at all. From a preliminary analysis, the impact of organic degradation was to degrade many organic compounds very rapidly thus "missing" their potential impacts on receptors. Because of the large uncertainties in the degradation rates for the organic compounds, it was decided to hold this analysis for further study.

The *individual* dose results are shown in Figure 54 for those radionuclides whose TEDE exceed the proposed EPA limit of 0.15 mSv/yr at any time during the assessment period for the on-site resident. The results for 15 radionuclides were predicted to exceed the proposed EPA limit and thus might require remedial action. The radionuclide posing the largest risk to the hypothetical on-site receptor is Co-60. However, because of the short half-life (i.e., 5.3 years) of Co-60, the inventory of this isotope has decayed to a stable nickel form and the radiation risk from Co-60 can be effectively ignored. Similar reasoning pertains to H-3 (with a 12.3-year half-life) as discussed in more detail in the results that follow.

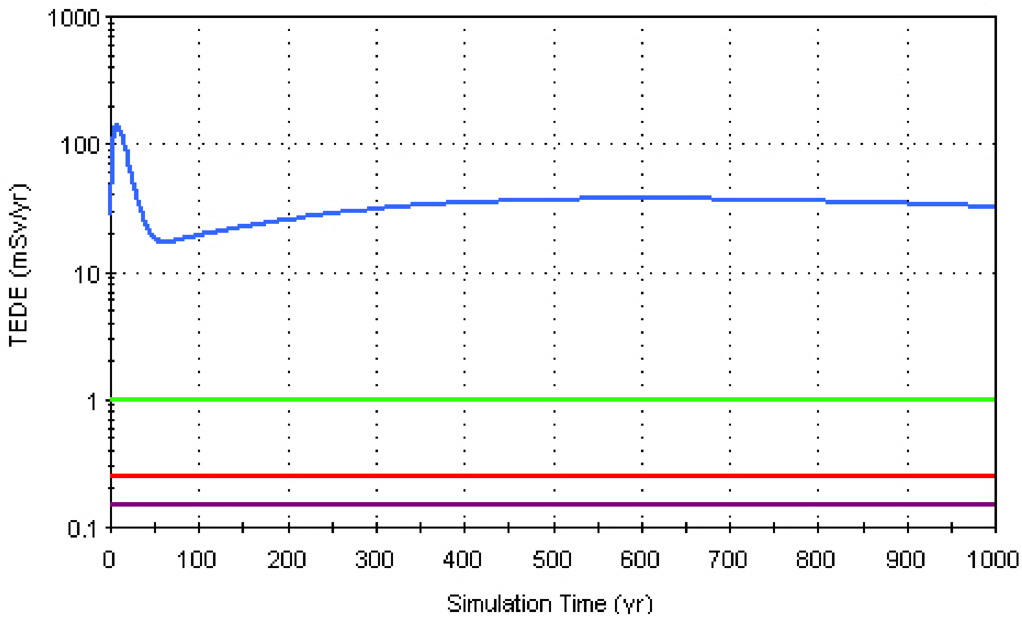

Figure 53. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways Summed over all Radionuclides in blue (compared to three possible dose limits at 1, 0.25, and 0.15 mSv/yr from top to bottom).

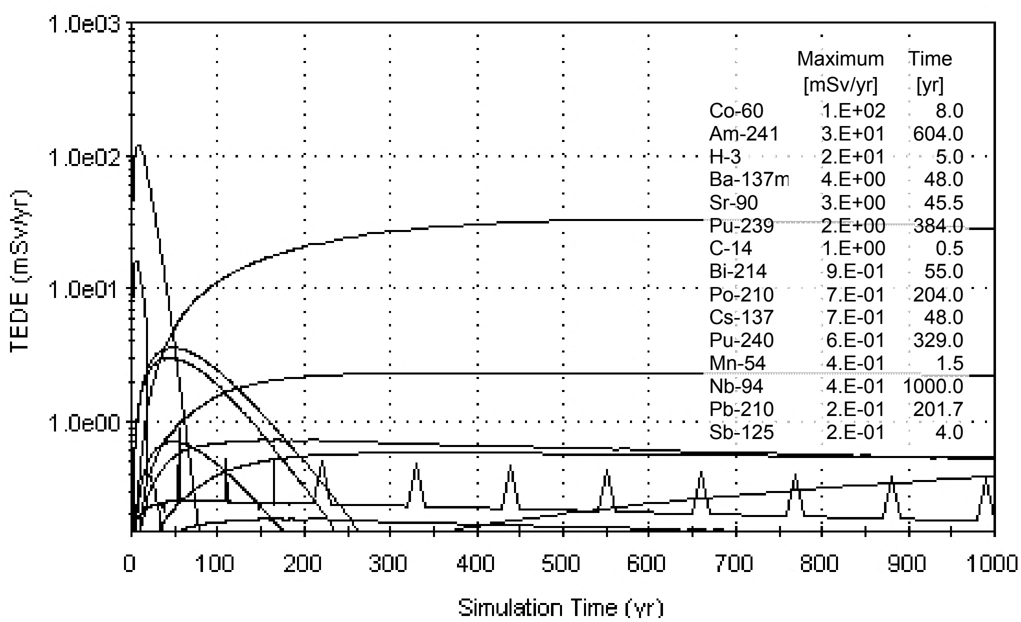

Figure 54. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways by Radionuclide that Exceeds 0.15 mSv/yr. (Spikes in predicted dose are due to periodic flooding impacts.)

The total and individual predicted dose-versus-time diagrams for radionuclides potentially impacting the on-site resident via the atmospheric pathway are shown in Figure 55 and Figure 56, respectively. These results indicate remedial action would also be required based solely on potential atmospheric pathway impacts when compared to the dose limit of 0.10 mSv/yr. However, no additional radionuclides (compared to those from Figure 54) were identified based on the atmospheric pathway results.

Proposed remedial actions must take into account that unacceptable risks are posed by multiple contaminants via multiple pathways. It is possible that multiple receptors and differing time frames may have to be accounted for in the identification of contaminants of potential concern for a site. The screening risk tool developed in this research provides the ability to complete such a comprehensive analysis using a set of consistent models and assumptions.

The total radionuclide morbidity risk results (in terms of the total annual latent cancer incidence rate) are provided in Figure 57 and, as for the dose results, exceed the EPA cancer risk "action limit" of 10^{-4} by more than an order of magnitude. The risk limit of 10^{-4} is "annualized" by dividing it by the national median time exposure duration at one residence of 9 years (USEPA 1989) for use in the diagram. The total cancer mortality results (i.e., total annual latent cancer deaths that are not shown) for the radionuclides paint a similar picture. Thus, based on both expected dose and cancer risk results from radionuclides, the contaminants buried at the site pose an unacceptable risk. This assessment agrees with the results of the CERCLA remedial investigation being carried out at the Idaho Site SDA (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002).

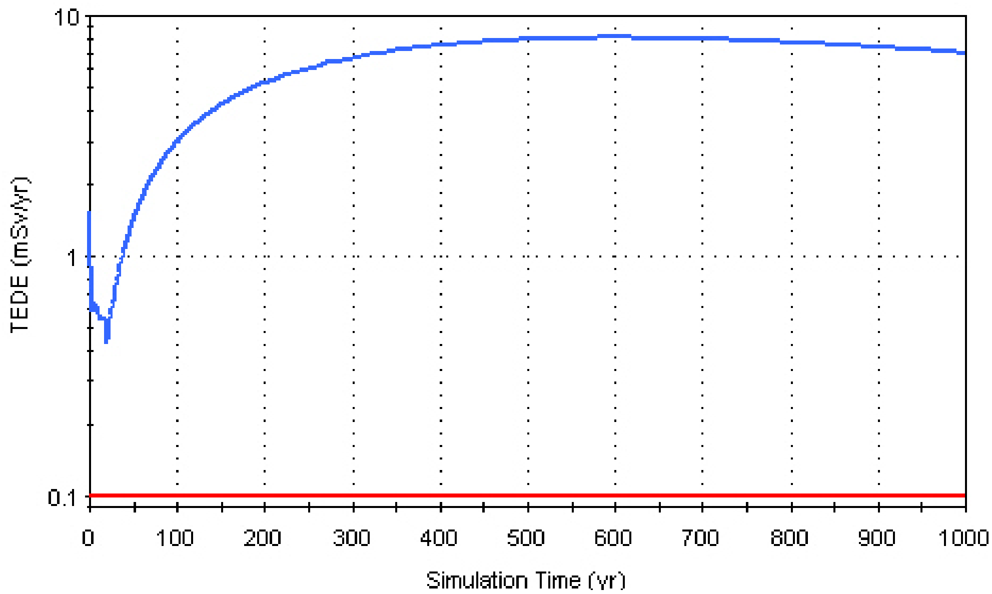

Figure 55. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for the Atmosphere Pathway Summed over all Radionuclides in blue (compared to the dose limit at 0.10 mSv/yr).

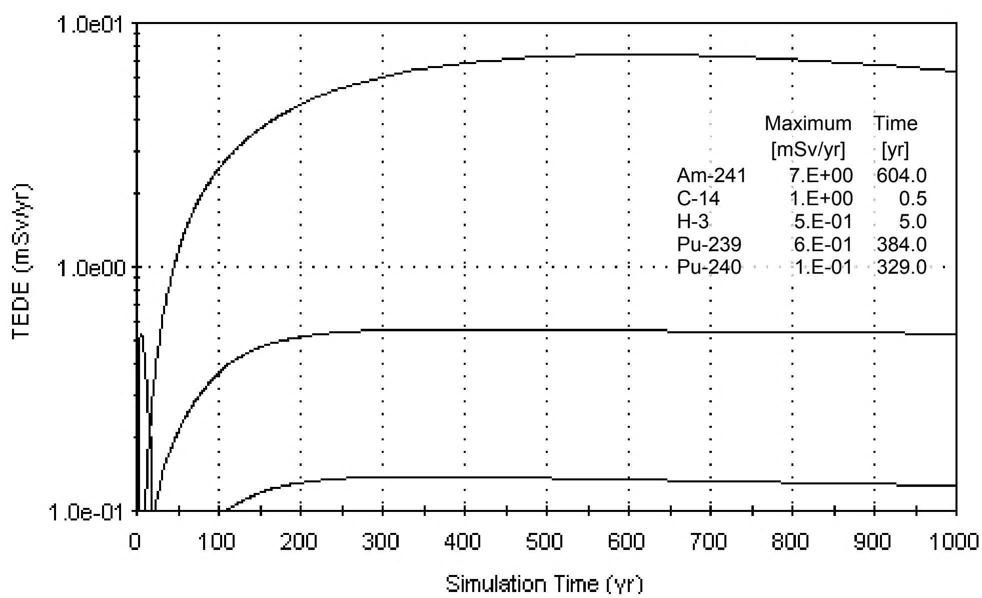

Figure 56. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for the Atmosphere Pathway by Radionuclide that Exceeds 0.10 mSv/yr.

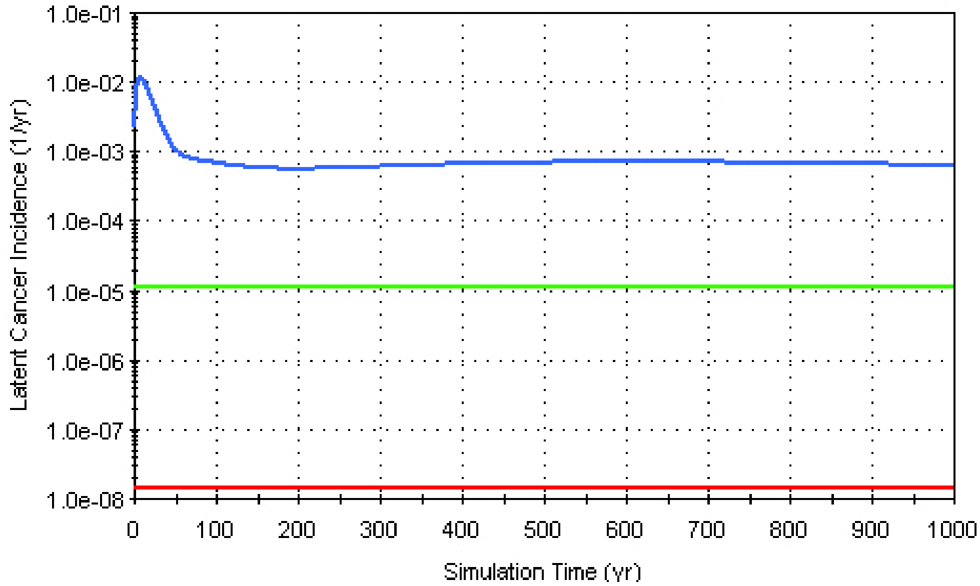

Figure 57. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Cancer Morbidity Rate for All Pathways and All Radionuclides in blue (compared to EPA 10^{-4} and 10^{-6} cancer risk limits from top to bottom converted to annual bases).

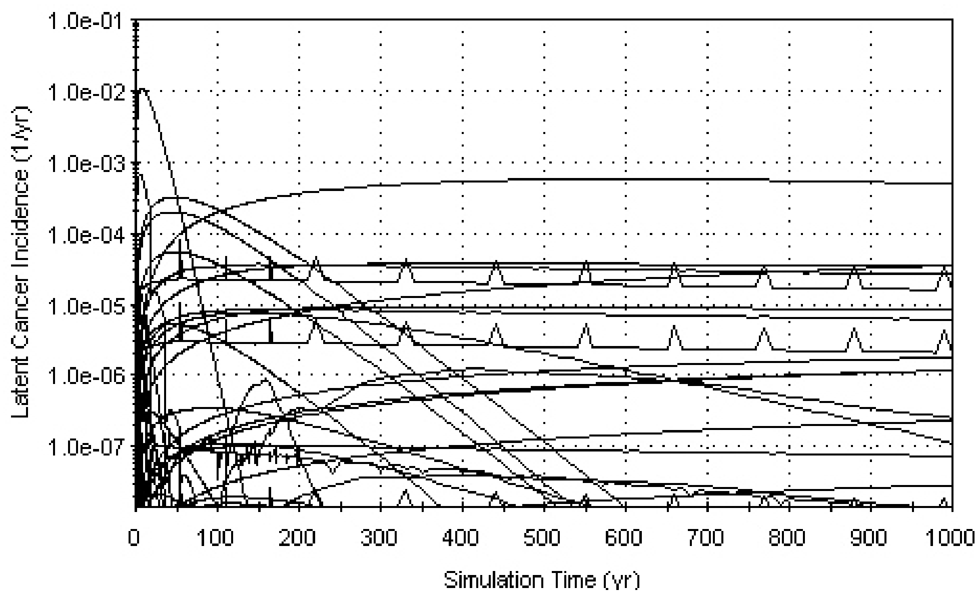

Figure 58. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Cancer Morbidity Rate for All Pathways by Radionuclide Exceeding the EPA 10^{-6} *de minimus* limit converted to an annual basis for use on this diagram. A total of 52 radionuclides, which are too numerous to list, exceed the limit. Spikes in predicted dose are due to periodic flooding impacts.

Morbidity results are shown in Figure 58 for those radionuclides whose rates exceed the EPA *de minimus* limit (of 10^{-6}) at any time during the assessment period for the on-site resident. A total of 52 radionuclides (including the 15 from the previous dose analysis in Figure 54) exceed the *de minimus* limit. As illustrated in Figure 59, 13 radionuclides have morbidity risks exceeding the EPA "action limit" (of 10^{-4}). The same 13 radionuclides were identified in the dose analysis; the two "missing" radionuclides are Pb-210 and Pu-240, which have peak values just less than the "action limit"[169]. The results obtained from the dose analysis are similar to those for latent cancer morbidity when bounded by the EPA "action limit".

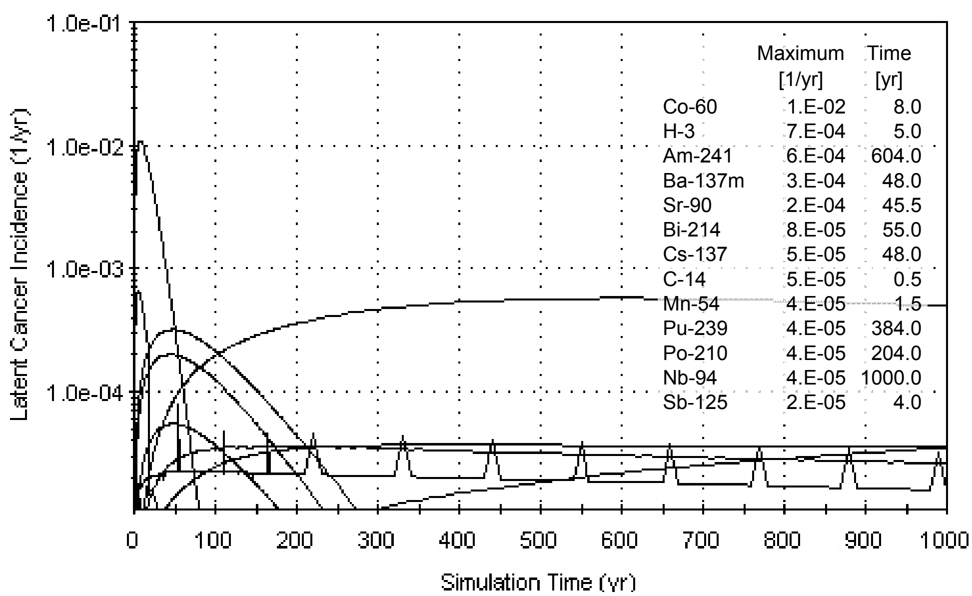

Figure 59. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Cancer Morbidity Rate for All Pathways by Radionuclide Exceeding the EPA "action limit" of 10^{-4} converted to an annual basis for use on this diagram.

[169] As described in footnote 167 on page 402, the EPA "action limit" of 10^{-4} is divided by the national median exposure duration at one residence of 9 years (USEPA 1989) for use on Figure 59. If one were to instead divide this limit by the conventional EPA lifetime exposure duration of 70 years (USEPA 1989), the two "missing" radionuclides (and more) would exceed the limit. These results demonstrate the impact of the exposure duration selection on predicted risk.

For the expected inventory and conditions case, the total predicted chemical cancer risks are illustrated in Figure 60. The total predicted chemical cancer risk exceeds the EPA "action limit" until approximately 500 years into the assessment period. The cancer risks for individual chemicals are illustrated in Figure 61 for those chemicals whose predicted lifetime cancer risks (over a 70-year exposure duration) exceed the EPA *de minimus* risk limit. For this case, seven chemicals are deemed to pose unacceptable cancer risks to the on-site resident. Peak risks for three of the seven chemical (i.e. carbon tetrachloride, benzene, and dichloromethane) pass within the first two years after the simulation begins (and thus long before the present time).

When examining the predicted total and individual non-cancer risks in Figure 62 and Figure 63, respectively, no additional chemicals pose unacceptable risks when compared to the EPA hazard limit of unity. (Carbon tetrachloride was previously identified as a chemical cancer risk in Figure 61.) However, the maximum hazard quotients for 1,1,1-trichloroethane (TCA), nitrates, tetrachloroethylene (PCE), and U-238 are within an order of magnitude of the EPA limit. Importantly, the hazard quotient for U-238 is increasing at the end of the assessment period (as shown in Figure 63).

Based on the original dose assessment results, contaminants buried in the SDA pose unacceptable risks. Unacceptable risks are also found based on predicted cancer morbidity and mortality rate (for radionuclides), chemical cancer incidence rates (primarily volatile organic compounds (VOCs)), and non-cancer effects (primarily uranium and VOCs). Therefore, as suggested in the SDA remedial investigation reports (Becker et al. 1998; Holdren et al. 2006; Holdren et al. 2002), there is little doubt that the SDA site poses unacceptable risks and requires remedial attention.

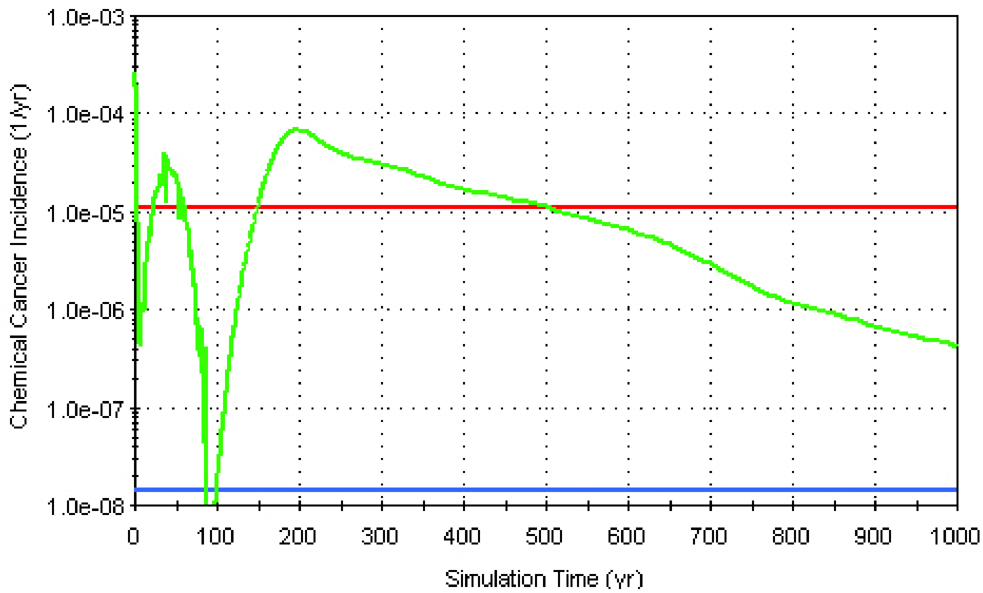

Figure 60. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Cancer Incidence Risk for All Pathways and All Chemicals in green (compared to the EPA 10^{-4} and 10^{-6} cancer risk limits from top to bottom converted to annual bases)

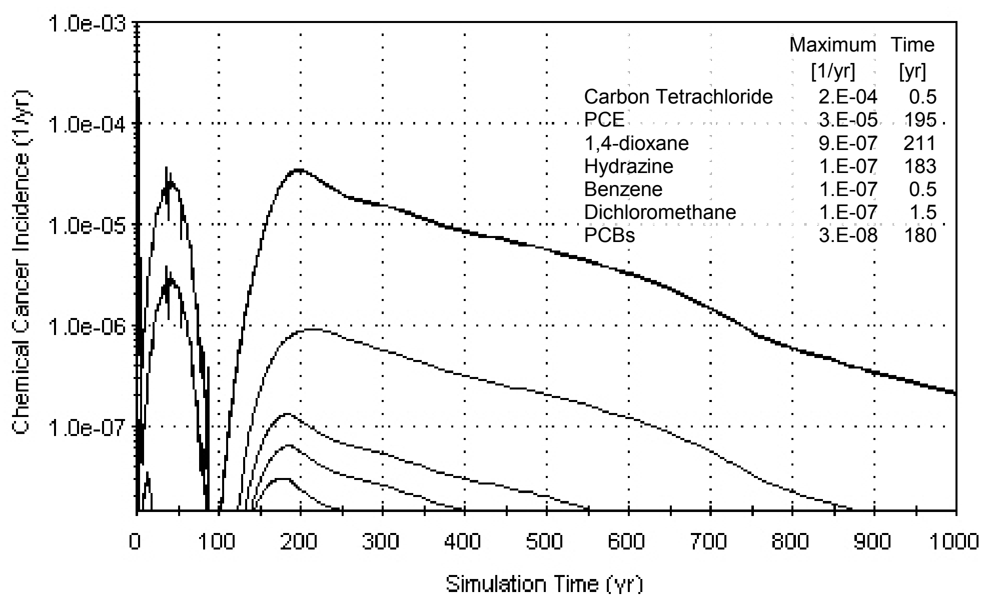

Figure 61. SDA *DBRA-Expected* On-Site Resident Scenario: Annual Cancer Incidence Risk for All Pathways by Individual Chemical Exceeding the EPA 10^{-6} *de minimus* risk limit converted to an annual basis. (The carbon tetrachloride, benzene, dichloromethane peak risks are spikes near the origin.)

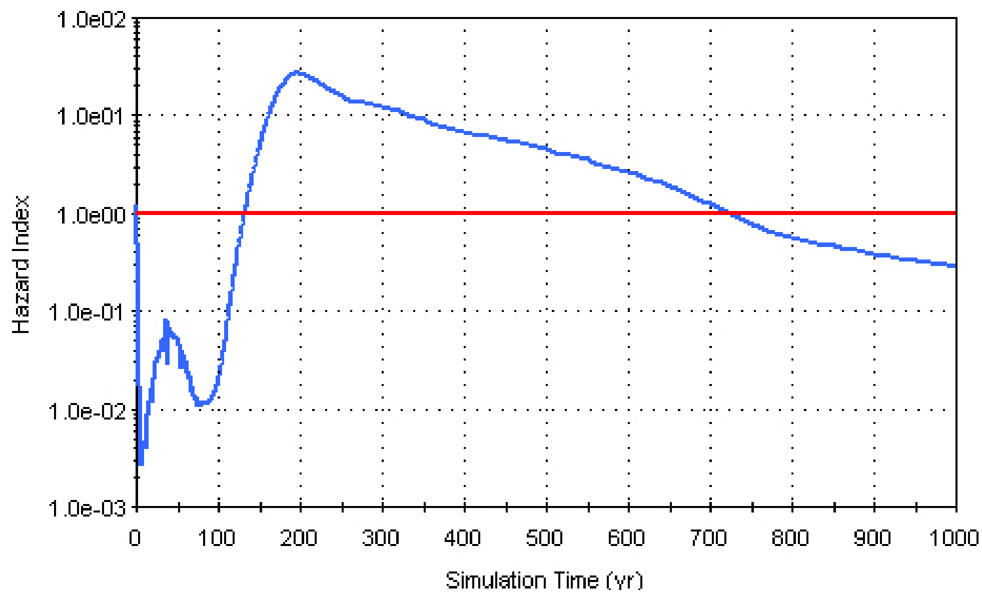

Figure 62. SDA *DBRA-Expected* On-Site Resident Scenario: Hazard Quotients Summed over All Chemicals and Pathways (Hazard Index) in blue (compared to corresponding EPA Limit of 1)

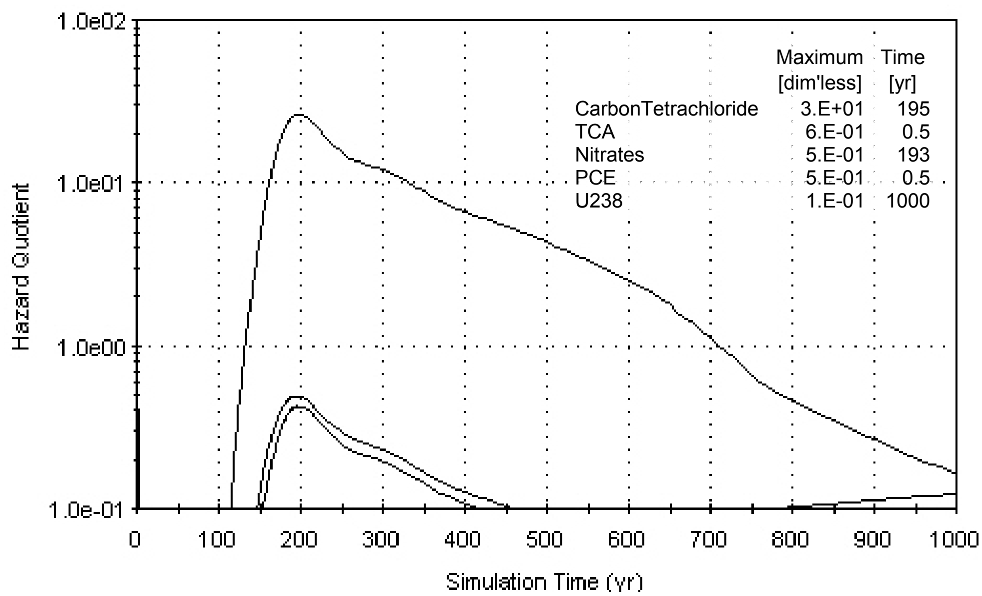

Figure 63. SDA *DBRA-Expected* On-Site Resident Scenario: Hazard Quotient for All Pathways by Individual Chemical Exceeding a Value of 1/10. (The tetrachloroethylene (PCE) and 1,1,1-trichloroethane (TCA) peak risks are spikes near the origin and the U-238 hazard quotient is increasing at the end of the simulation.)

For expected SDA conditions (i.e., the *DBRA-Expected* case in Table 49), the doses, risks, and hazards predicted using the screening risk tool are summarized in Table 50. For example, the large, predicted Co-60 doses and risks occurred before what would be considered present time (e.g., Year 60) because of the short, 5.3-yr half-life for this radionuclide. Those effects occurring before the present time are shown in italics. The impact of the timing can be quite dramatic; for example, the impacts for more than half of the radionuclides exceeding their limits for morbidity occur before the present time.

As shown in Table 50, many contaminants were predicted to pose unacceptable risks to the on-site resident during the 1,000-year assessment period. However, additional contaminants were identified in the Idaho Site remedial investigation including Ac-227, Am-243, I-129, Np-237, Pa-231, Th-229, Th-230, Th-232, U-235, U-236, and methylene chloride (Holdren et al. 2006). On the other hand, hydrazine was identified using the screening risk tool but not in the Idaho Site remedial investigation. According to Holdren et al. (2006), hydrazine has a short environmental half-life (as confirmed in Table 108 of Appendix D). Environmental degradation was not enabled in the screening risk tool to allow comparison to the original SDA remedial investigation results, which considered degradation on a case-by-case basis[170]. In the SDA remedial investigation, hydrazine was not identified as a contaminant of potential concern because of its short half-life (Holdren et al. 2006). Differences between the screening risk tool and remedial investigation results can be attributed to different models, scenarios, and assessment periods.

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[170] It was decided that either organic degradation would be used for all cases and affected compounds or not at all. From a preliminary analysis for the SDA, the impact of organic degradation was to degrade all many of the organic compounds very rapidly thus "missing" their potential impacts on receptors. Because of the large uncertainties in the degradation rates for the organic compounds, it was decided to hold this analysis for further study. Furthermore, it appeared inconsistent to assume the hydrazine had decomposed to non-hazardous compounds while not considering those organic compounds (e.g., trichloroethylene or PCE) buried in the SDA that might degrade to more hazardous compounds (e.g., vinyl chloride).

Table 50. *DBRA-Expected* Case: Baseline Doses and Risks for SDA Buried Wastes

	SDA COPC[a]	Predicted Point-Value Peak Dose or Risk Exceeding Limit[b]				
		Morbidity [risk/yr][c]	TEDE [mSv/yr][d]	Mortality [risk/yr][c]	Cancer [risk/yr][c]	Non-Cancer [HQ][e]
Co-60	(a)	*1.E-02*	*1.E+02*	*5.E-03*	---	---
H-3		*7.E-04*	*2.E+01*	*6.E-04*	---	---
Am-241	√	6.E-04	3.E+01	3.E-04	---	---
Ba-137m		*3.E-04*	*4.E+00*	*1.E-04*	---	---
Sr-90	√	*2.E-04*	*3.E+00*	1.E-04	---	---
Bi-214		*8.E-05*	*9.E-01*	*3.E-05*	---	---
Cs-137	√	*5.E-05*	*7.E-01*	*3.E-05*	---	---
C-14	√	*5.E-05*	*1.E+00*	*6.E-05*	---	---
Mn-54		*4.E-05*	*4.E-01*	*2.E-05*	---	---
Pu-239	(a)	4.E-05	2.E+00	2.E-05	---	---
Po-210		4.E-05	7.E-01	2.E-05	---	---
Nb-94	√	4.E-05	4.E-01	1.E-05	---	---
Sb-125		*2.E-05*	*2.E-01*	---	---	---
Pb-214		*1.E-05*	---	*4.E-06*	---	---
Pu-240	(a)	9.E-06	6.E-01	5.E-06	---	---
Pb-210	√	9.E-06	2.E-01	3.E-06	---	---
Ni-63		8.E-06	---	1.E-06	---	---
Eu-154		*7.E-06*	---	*3.E-06*	---	---
Co-58		*6.E-06*	---	*3.E-06*	---	---
Y-90		*5.E-06*	---	*1.E-06*	---	---
Fe-59		*4.E-06*	---	*2.E-06*	---	---
Cs-134		*3.E-06*	---	*1.E-06*	---	---
Ra-226	√	2.E-06	---	5.E-07	---	---
Fe-55		*2.E-06*	---	*6.E-07*	---	---
Pr-144		*2.E-06*	---	*7.E-07*	---	---
Nb-95		*1.E-06*	---	*6.E-07*	---	---
Tc-99	√	1.E-06	---	2.E-07	---	---
U-234	√	1.E-06	---	4.E-07	---	---
Ni-59		1.E-06	---	2.E-07	---	---
Ce-144		*1.E-06*	---	*2.E-07*	---	---
Zr-95		*8.E-07*	---	*3.E-07*	---	---
Eu-155		*5.E-07*	---	*2.E-07*	---	---
Pu-238	(a)	4.E-07	---	2.E-07	---	---
Pu-241		*3.E-07*	---	*2.E-07*	---	---
Sb-124		*3.E-07*	---	*1.E-07*	---	---
Be-10		2.E-07	---	3.E-08	---	---
Cr-51		*2.E-07*	---	*9.E-08*	---	---
Rh-106		*2.E-07*	---	*8.E-08*	---	---
Zn-65		*2.E-07*	---	*6.E-08*	---	---
Tl-208		1.E-07	---	5.E-08	---	---

a. The designation "SDA COPC" indicates whether the constituent is considered a contaminant of potential concern from the SDA remedial investigation (Holdren et al. 2006). Constituents (e.g., Co-60 and hydrazine) with either a short radioactive half-life or environmental degradation were omitted from the SDA remedial investigation. The three plutonium isotopes indicated were added to the SDA COPC list not because of high risk but instead as special case COPCs to acknowledge uncertainties about plutonium mobility in the environment (Holdren et al. 2006).

b. Those doses, risks, or hazard quotients in *italics* represent effects taking place before the present time.

c. The relevant limit is the EPA *de minimus* cancer risk limit (10^{-6}) converted to an annual basis (i.e., 1.4×10^{-8} yr^{-1}) using the conventional EPA lifetime exposure duration of 70 years (USEPA 1989).

d. The dose limit used is the proposed EPA limit of 0.15 mSv/yr.

e. The non-cancer hazard quotient (HQ) of 1/10 is used for screening purposes.

Table 50, Continued

	SDA COPC[a]	Predicted Point-Value Peak Dose or Risk Exceeding Limit[b]				
		Morbidity [risk/yr][c]	TEDE [mSv/yr][d]	Mortality [risk/yr][c]	Cancer [risk/yr][c]	Non-Cancer [HQ][e]
Eu-152		*1.E-07*	---	*5.E-08*	---	---
Bi-210		1.E-07	---	1.E-08	---	---
Te-125m		*6.E-08*	---	*1.E-08*	---	---
Cl-36	√	4.E-08	---	---	---	---
Rn-222		*3.E-08*	---	*2.E-08*	---	---
U-233		3.E-08	---	---	---	---
Cm-244		*2.E-08*	---	---	---	---
U-232		2.E-08	---	---	---	---
Ru-106		*2.E-08*	---	---	---	---
Bi-212		2.E-08	---	---	---	---
Sc-46		*2.E-08*	---	---	---	---
Sn-119m		*2.E-08*	---	---	---	---
Carbon Tetrachloride	√	---	---	---	*2.E-04*	3.E+01
Tetrachloroethylene (PCE)	√	---	---	---	3.E-05	*5.E-01*
1,4-dioxane	√	---	---	---	9.E-07	---
Hydrazine	(a)	---	---	---	1.E-07	---
Benzene		---	---	---	*1.E-07*	---
Methylene chloride (Dichloromethane)	√	---	---	---	*1.E-07*	---
PCBs		---	---	---	3.E-08	---
1,1,1-Trichloroethane (TCA)		---	---	---	---	*6.E-01*
Nitrates	√	---	---	---	---	5.E-01
U238	√	---	---	---	---	1.E-01

a. The designation "SDA COPC" indicates whether the constituent is considered a contaminant of potential concern from the SDA remedial investigation (Holdren et al. 2006). Constituents (e.g., Co-60 and hydrazine) with either a short radioactive half-life or environmental degradation were omitted from the SDA remedial investigation.

b. Those doses, risks, or hazard quotients in *italics* represent effects taking place before the present.

c. The relevant limit is the EPA *de minimus* cancer risk limit (10^{-6}) converted to an annual basis (i.e., 1.4×10^{-8} yr^{-1}) using the conventional EPA lifetime exposure duration of 70 years (USEPA 1989).

d. The dose limit used is the proposed EPA limit of 0.15 mSv/yr.

e. The non-cancer hazard quotient (HQ) of 1/10 is used for screening purposes.

However, the purpose of this research was not to reproduce results from the corresponding DOE site remedial investigations. Although the inventory information used is consistent, the models and assumptions used in the site remedial investigations are different than those in the screening risk tool, which was developed as an integrated platform with probabilistic capabilities. However, the COPCs identified during the SDA remedial investigation (i.e., in Table 10 in Chapter IV) was based on upper bound inventories; best inventory estimates were used in the screening risk tool to generate Table 50. Different scenarios were used. The screening risk tool incorporated additional transport pathways that may impact results. Both approaches represented here merit attention when identifying COPCs; however, the fact that they identified different COPCs for the SDA reinforces the idea that explicit declaration of the assumptions and value judgments made when performing a risk analysis is essential to its transparency.

Three additional "deterministic" or point-value *baseline* simulations were run to evaluate the impacts of changes in waste form assumptions (i.e., *DBRA-ExpLoose*), bounding (i.e., 95th-percentile) values for inventories and parameters used in the screening risk tool (i.e., *DBRA-Maximum*), and bounding values with no retardation or solubility constraints (i.e., *DBRA-WorstCase*). These impacts were originally evaluated as one-at-a-time or few-at-a-time verification studies in Appendix G to demonstrate their intended operations and anticipated impacts on predicted dose and risk.

The impacts of the waste form assumptions on dose, risk, and hazard predictions were further evaluated using the *DBRA-ExpLoose* results compared to expected results (i.e., Figure 53 and Figure 54 for *DBRA-Expected* dose results). The only difference between the *DBRA-ExpLoose* and *DBRA-Expected* cases was that contaminants in *DBRA-ExpLoose* were assumed to be "loose" (i.e., not associated with any waste form)[171]. The *DBRA-ExpLoose* results for total and individual doses are presented in Figure 64 and Figure 65, respectively. The impact on the total dose prediction from all pathways and all radionuclides (i.e., Figure 64 versus Figure 53) was less than a factor of two (where both indicate that the risks posed by site conditions would be unacceptable).

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[171] Apart from the parameters associated with waste form and container failure, the parameters are at their expected values as in the *DBRA-Expected* baseline simulation described in Table 49.

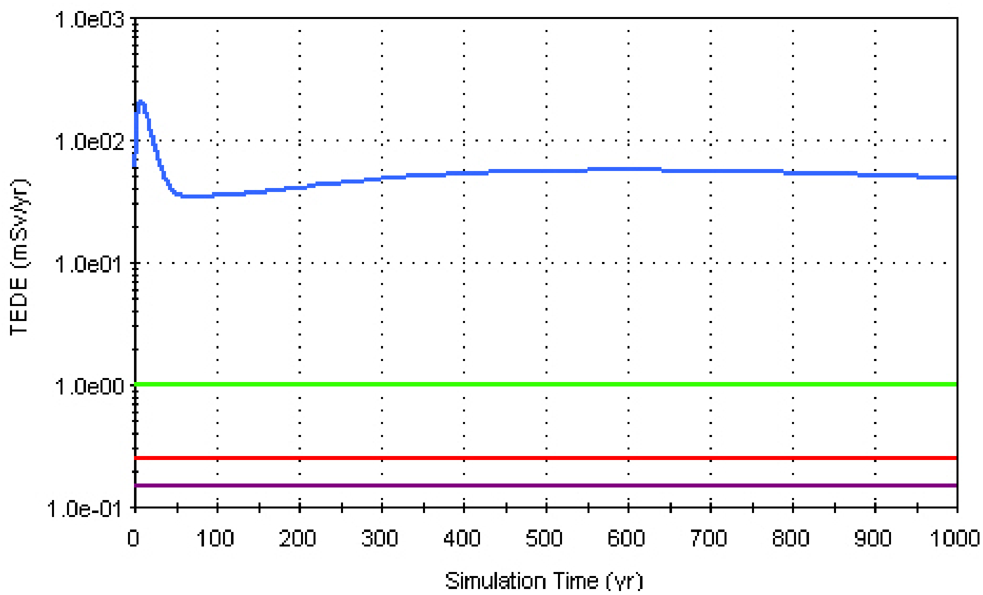

Figure 64. SDA *DBRA-ExpLoose* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides in blue (compared to dose limits at 1, 0.25, and 0.15 mSv/yr from top to bottom)

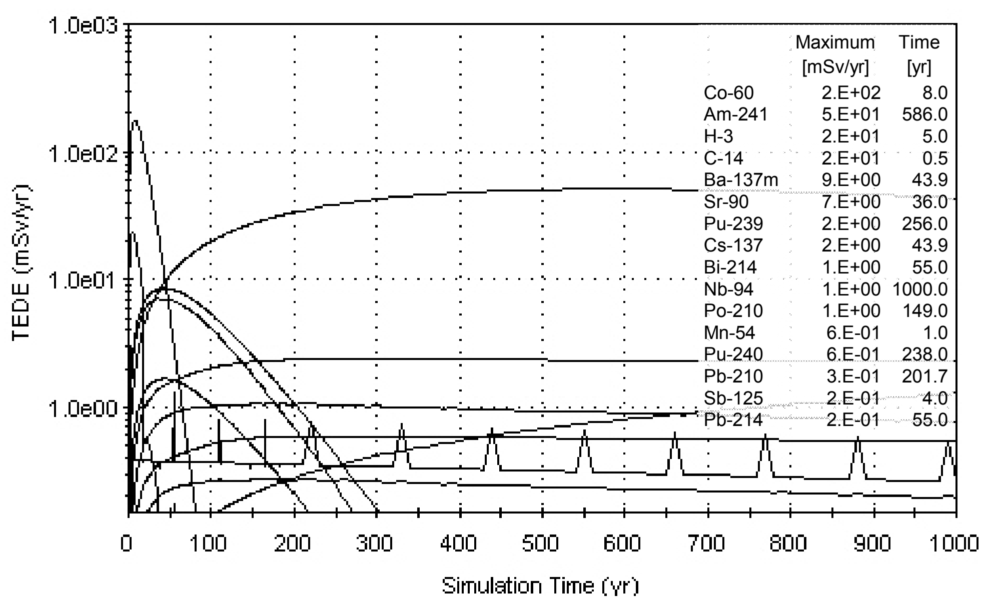

Figure 65. SDA *DBRA-ExpLoose* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways by Radionuclide that Exceeds 0.15 mSv/yr. (The spikes in predicted dose are due to periodic flooding impacts.)

Because radionuclides were heterogeneously distributed throughout waste forms and containers (both of which affect contaminant release), impacts much larger than by a factor of two were manifested for many individual radionuclides. Although Figure 54 and Figure 65 appear similar, ignoring waste form and containers (in *DBRA-ExpLoose*) increased predicted peak doses by up to 15 times (although most by less than a factor of 3) for COPCs than when waste forms and containers were considered. In fact, the dose for one radionuclide, Pb-214, became unacceptably large when waste forms were not considered in the simulation. These results were consistent with the corresponding risks and hazards (which is not surprising considering that exposure is the actual driver for these risk and hazard predictions).

The effects on dose predictions were even more profound when bounding (e.g., upper 95th-percentile[172]) inventories and transport parameters were employed in the "deterministic" or point-value simulation (i.e., *DBRA-Maximum*) for the SDA. The impact on the total dose prediction from all pathways and all radionuclides (i.e., Figure 66 versus Figure 53) was more than an order of magnitude (i.e., approximately 20) higher than the expected results in *DBRA-Expected* (where both results indicated that the risks posed by site conditions were unacceptable). When compared to the results obtained by assuming all contaminants were "loose" (i.e., compared to Figure 64), the total dose results for the bounding case (in Figure 66) were approximately one order of magnitude larger.

[172] Either the upper 95th-percentile or lower 5th-percentile value was used based on the judged impact on the resulting dose and risk predictions. Judgement was used to determine *a priori* the bounding values to be employed based on suspected primary release mechanism, transport pathways, and receptors.

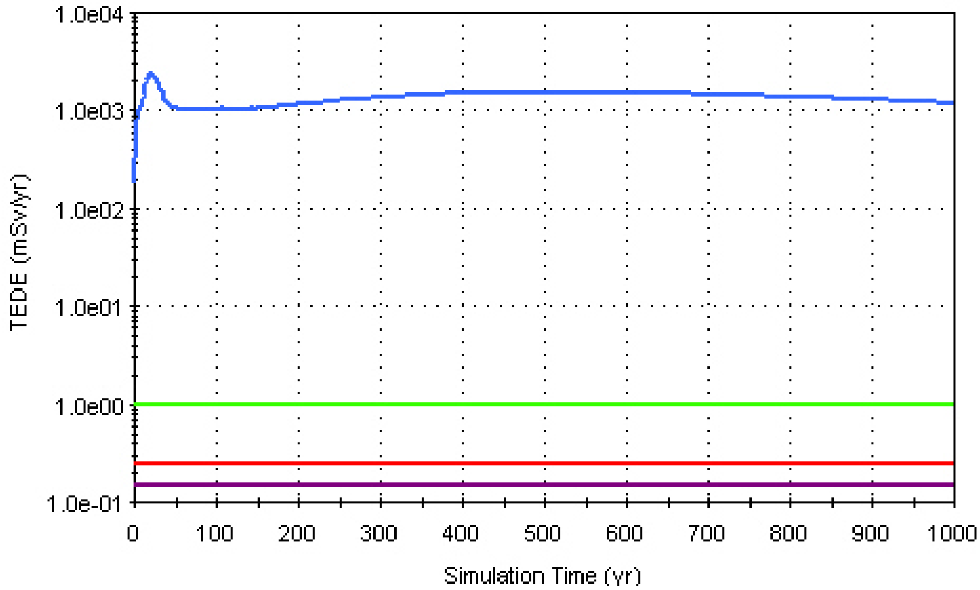

Figure 66. SDA *DBRA-Maximum* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides in blue (compared to dose limits at 1, 0.25, and 0.15 mSv/yr from top to bottom)

The individual dose predictions for the point-value case where bounding (e.g., upper 95th-percentile) values were used (i.e., *DBRA-Maximum* as described in Table 49) are provided in Figure 67. The ratios of the maximum predicted dose from the maximum case for COPCs to the expected predictions (for the *DBRA-Expected* case) ranged from less than an order of magnitude to almost a factor of 200. Furthermore, the number of radionuclides with unacceptable predicted doses increased from 15 for the expected results (i.e., for *DBRA-Expected* in Figure 54) to 28 as indicated in Figure 67. As expected the relative increases in morbidity risks and the number of radionuclides that pose unacceptable predicted risks increased even more dramatically if cancer incidence (morbidity) was used as the basis for acceptance.

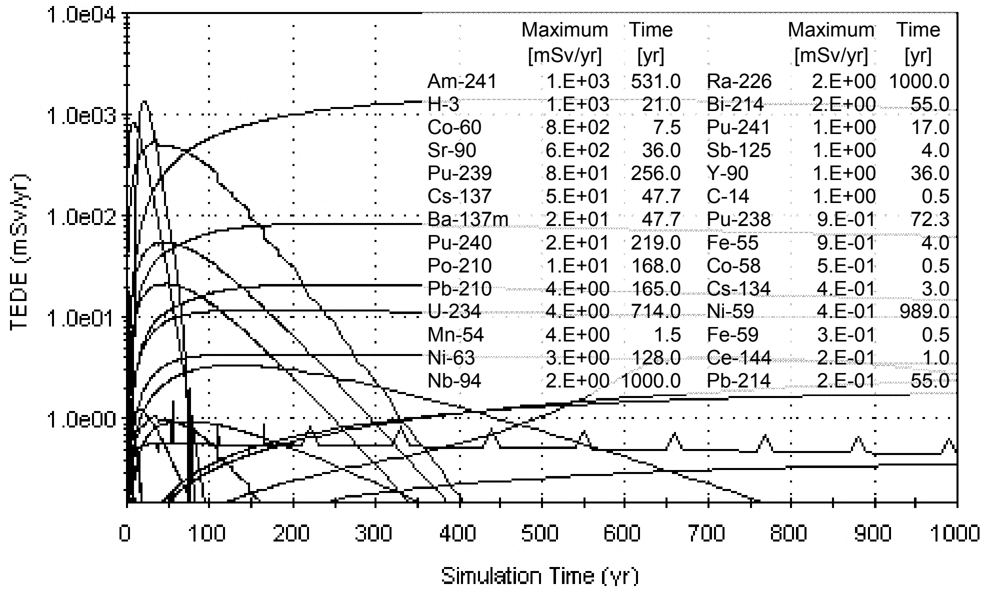

Figure 67. SDA *DBRA-Maximum* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways by Individual Radionuclide that Exceeds 0.15 mSv/yr. (The spikes in predicted dose are due to periodic flooding impacts.)

As illustrated in the preceding figures, the impacts of waste form and container failures as well as bounding inventory and transport parameters in effect reduced the release and transport of contaminants from the buried waste site into and through the environment. However, preliminary results from the verification tests (described in Appendix G) indicated that retardation (i.e., based on linear partition coefficients, K_d's) and solubility may impact predicted hazards, doses, and risks more profoundly. The purpose of the *DBRA-WorstCase* simulation (as described in Table 49) was to test this possibility. The impact on the total dose from all pathways and all radionuclides (i.e., Figure 68 versus Figure 53) was more than four orders of magnitude (i.e., a factor of approximately 40,000) than the expected results (for *DBRA-Expected*).

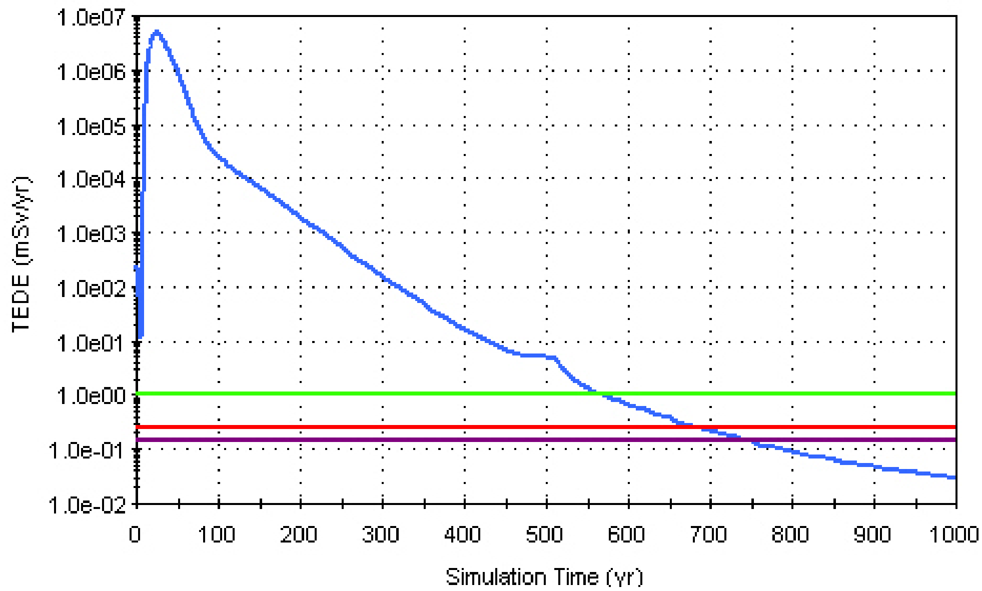

Figure 68. SDA *DBRA-WorstCase* On-Site Resident Scenario: Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides in blue (compared to dose limits at 1, 0.25, and 0.15 mSv/yr from top to bottom)

The impacts for individual radionuclides from employing worst-case assumptions (i.e., *DBRA-WorstCase* as described in Table 49) were the most profound of all for both dose and timing as shown in Figure 69. Changes in the relative magnitudes of the peak predicted doses for individual COPCs varied from less than one (for Ba-137m) to more than six orders of magnitude (for Pu-238 and Pu-241). The timing of potential impacts also changed dramatically where contaminants are "flushed" though the system relatively rapidly. The assumptions and risk metrics selected dictated the identities and numbers of COPCs. For the worst-case scenario, a total of 46 radionuclides of potential concern were identified if predicted annual dose was used to represent radionuclide risks to the on-site receptor during the assessment period.

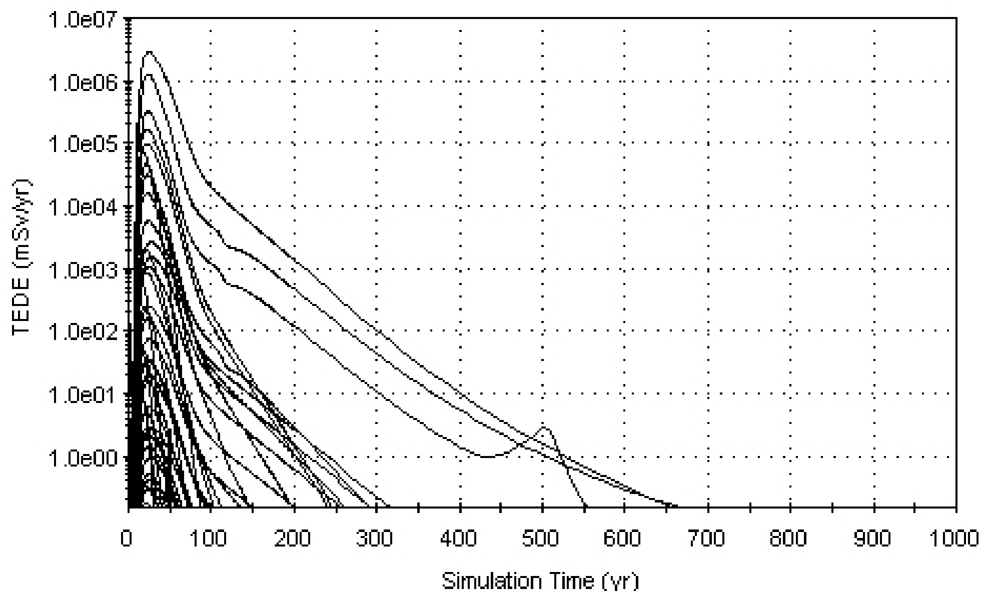

Figure 69. SDA *DBRA-WorstCase* for On-Site Resident: Annual Total Effective Dose Equivalent (TEDE) for All Pathways by Individual Radionuclide. (There are too many radionuclides exceeding 0.15 mSv/yr to list on this figure in an intelligible manner.)

SDA: Probabilistic Assessment of Baseline Risks

The list of contaminants of potential concern (COPCs) can vary significantly based on the assumptions, models, scenarios, value judgments, etc. used to assess potential risks for a site. Because of the many uncertainties involved in the assessment of baseline risks, it appears overly optimistic to use best inventory estimates and expected (50^{th}-percentile) transport and other parameters (i.e., the *DBRA-Expected* case) to identify COPCs. On the other hand, the use of upper-bound inventories and worst-case model parameters excessively overestimates exposure and risk. These inflated risk results might influence regulators to require unneeded or overly aggressive remedial actions be taken when the effort and money might be better spent elsewhere (although these are also value judgments that enter into the remedial decision). Perhaps another approach is warranted.

Although capable of performing point-value calculations (e.g., those described in Table 49), the screening risk tool can be used to perform stochastic computations involving contaminant fate and transport employing Monte Carlo simulation (GTG 2005a; b; c). Point-value calculations have been used historically to determine both whether or not a site is a candidate for remedial action and the goals for the actions (USEPA 1989; 1991), However, confusion may result from the various COPCs and remedial goals obtained from the different conditions or risk metric that can be used. A more straightforward (if computationally more intensive) alternative for defining COPCs is proposed here to instead base the identification of COPCs and the definitions of cleanup goals on the *probabilistic* information obtained from the screening risk tool.

The results from a Monte Carlo simulation describe the uncertainties in the resulting dose, risk, and hazard predictions and depend on the probability distributions defined for the uncertain parameters in the screening risk tool and the number of realizations performed. One very useful representation resulting from probabilistic risk assessments is the complementary cumulative distribution function (CCDF), also referred to as the risk exceedance curve, that illustrates the likelihood of exceeding a given risk metric. The CCDF is illustrated in Figure 70 for those radionuclides whose annual peak doses exceeded 0.15 mSv/yr from 50 realizations of the baseline SDA conditions using the probability distributions defined in Chapter VI[173]. The number of realizations can be increased for greater accuracy, but the method is demonstrated nonetheless.

[173] The distributions used are the same as those employed to define bounding values for the *DBRA-Maximum* and *DBRA-WorstCase* results. The results for C-14 are also shown in Figure 70 for reference because it was identified as a COPC in Table 50 based on the point-value peak dose results.

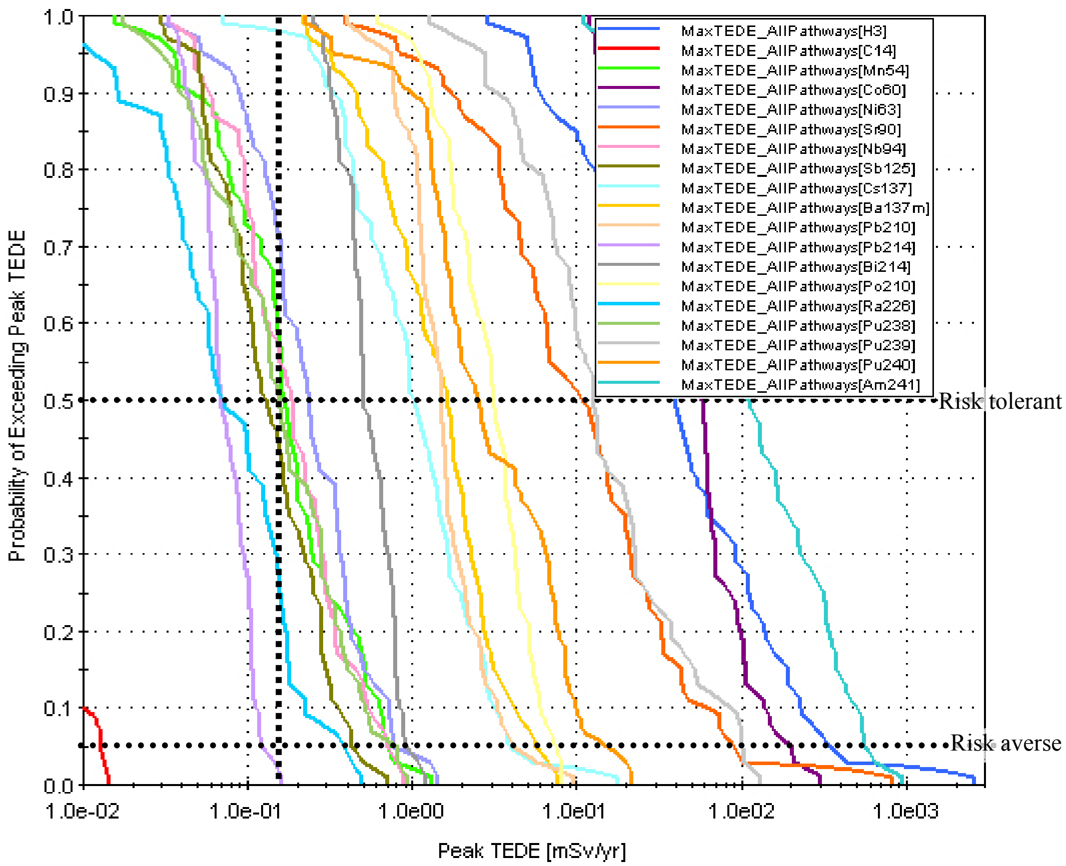

Figure 70. SDA Complementary Cumulative Distribution Function (CCDF) for the Peak TEDE Predictions for the On-Site Resident Scenario. (The black, dotted vertical line represents the proposed EPA limit of 0.15 mSv/yr. The black, dotted horizontal lines indicate whether one is risk averse or tolerant)

The CCDF in Figure 70 for the baseline SDA peak dose results illustrates one way of using probabilistic results to identify COPCs. The black, dotted vertical line on this figure indicates the limit used (in this case, the proposed EPA dose limit of 0.15 mSv/yr). Those contaminants with peak doses falling to the right of the vertical line are candidate COPCs. To define a COPC, a probability threshold, p, must be selected. Two thresholds are suggested here: risk averse ($p \leq 0.05$) or risk tolerant ($p \leq 0.50$). Those contaminants with CCDF curves that intersect the selected risk threshold to the right of the limit (e.g., 0.15 mSv/yr) would be identified as COPCs.

An example of the impact of the proper use of the probabilistic information in defining COPCs will help demonstrate the utility of the approach. If one adopts a risk tolerant view ($p \leq 0.50$), the COPCs identified correspond well to the *TEDE* list in Table 50 (with the addition of Ni-63 and exclusion of C-14). However, the benefit of the new approach comes into light as the aversion to risk increases (i.e., the threshold probability decreases). If one adopts a risk-averse stance (i.e., $p \leq 0.05$), a total of 18 radionuclides are identified as COPCs. The proper use of the probabilistic results significantly reduces the number of COPCs (when compared to the *DBRA-Maximum* results in Figure 67 where bounding values are used and 28 COPCs were identified). Therefore, the cleanup effort can be focused on those contaminants likely to pose unacceptable risks.

The results obtained from the CCDF do not include value judgments including the risk metric and conditions specified. This is not to say that use of the CCDF is not without value judgment. The probability distributions used in the tool are often based on expert judgment or available data. However, these same distributions are often used to define the bounding values used for the specific point-value case and any assumptions made in defining the distributions would impact both types of risks analysis.

The threshold probability (e.g., $p \leq 0.05$) selected to identify the contaminants that pose unacceptable risks is no more or less defensible than the percentile value (e.g., 95%) used to define the various model parameters describing the maximally reasonable exposed individual (especially when the same distributions are employed). Therefore, a reasonable argument can be made that the proper use of probabilistic information to identify COPCs can be much more transparent and consistent than the usual and customary manner in which point-value risk assessments are used to define COPCs.

SDA: Contaminants of Potential Concern (COPCs) and Waste Types

As illustrated by both point-value and probabilistic results, the primary radionuclides of potential concern included Co-60, tritium (H-3), various plutonium isotopes, Am-241, and fission products including Sr-90, Cs-137, and C-14. The primary contaminants posing chemical cancer and non-cancer risks included carbon tetrachloride, tetrachloroethylene (PCE), 1,4-dioxane, and dichloromethane. Other contaminants would be included based on the assumptions and value judgments made in defining the thresholds and criteria for unacceptable risks.

However, when considering remedial actions, concerns involving thresholds and risk aversion must be tempered by additional criteria that are also important. The first issue concerns the temporal nature of the exposure and risk factors predicted for contaminants. For example, Co-60 dominates many of the predicted risks associated with the SDA; however, because Co-60 has a short radioactive half-life (i.e., 5.3 years) and the bulk of this material was buried decades ago, no remedial action can impact the predicted Co-60 risks because the material has already decayed to a stable nickel isotope. As illustrated in Table 50, the temporal impact can be quite dramatic; the impacts for more than half of the radionuclides exceeding their dose limits occurred before what would be considered present day in the simulation.

Fission products (e.g., Sr-90, Cs-137, etc.) in the form of liquid wastes were primarily injected in boreholes in the SDA. Volatile organic compounds (VOCs) including carbon tetrachloride and trichloroethylene were originally buried with Rocky Flats Plant (RFP) wastes. There has been evidence that some fission products (e.g., Cs-137, C-14, etc.) and VOCs (especially carbon tetrachloride) have migrated as far as the Snake River Plain Aquifer (SRPA) beneath the SDA (Holdren et al. 2006). Therefore, remedial measures other than or in addition to retrieval would be required to manage unacceptable risks from these wastes. Vacuum vapor extraction is being used to remove and destroy VOCs, and the beryllium blocks (which serve as the primary source for C-14) have been grouted (Holdren et al. 2006). Because infiltrating water is a primary driver for contaminant migration, limiting the water that contacts the waste must be considered essential in controlling the risks posed by the site.

The spatial and temporal aspects of the risks posed by site wastes as well as the waste forms are essential factors in determining what remedial actions should be taken for a site. For example, excavation and waste retrieval will have little if any impact on those contaminants (e.g., carbon tetrachloride, C-14, Cs-137, etc.) that were released into the environment long ago and were mobile. Depending on conditions, this may also apply to certain plutonium isotopes in the SDA (Holdren et al. 2006).

Aggressive remedial actions such as retrieval must only be applied if they will remove a significant source of contamination posing unacceptable future risks[174]. Early actions including vacuum vapor extraction and beryllium block grouting (a source of C-14) have been performed to mitigate some of the most obvious, immediate sources of risk. These remedial actions appear warranted (although the vacuum vapor extraction action has not proven as effective in removing volatile organic compounds as originally anticipated (Holdren et al. 2006)).

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[174] There is a second part to this condition that is a value judgment on the part of the author. Aggressive remedial actions such as retrieval must be applied only if they will remove a significant source of contamination posing unacceptable risks without increasing life-cycle risks to potential receptors. Whether or not one is willing to risk multiple worker injuries or fatalities to reduce the number of hypothetical, future latent cancer incidences to receptors who may or may not be present is a value judgment. Ignoring worker risks and solely focusing on specific risks to the general public is also a value judgment.

From these considerations, it appears that, because of the nature of the risks presented by the RFP transuranic and VOCs remaining in the burial site, that these wastes should be the primary focus of any retrieval activities. These results agree with those used to define the areas for targeted retrieval activities for the SDA in Appendix D. The results in Appendix D were based on inventory information and SDA remedial investigation results and agree with the results from the screening risk tool.

SDA: Risk Metrics for Comparison Purposes

Many of the results computed using the screening risk tool were provided in terms of the annual total effective dose equivalent (TEDE) in mSv/yr. The TEDE is a well-used metric for directly predicting risks related to radiation exposures and is the primary U.S. Nuclear Regulatory Commission (NRC) metric for site cleanup. If only radionuclides are involved in estimating the risk due to a site and its cleanup, then dose in terms of the TEDE would be an excellent metric for assessing risks from the contaminated site.

However, for complicated sites including the DOE buried waste sites examined in this research, radionuclides are not the sole risk factors associated with the site and may not even be the primary risk drivers for substantial periods of time to workers or the general public. Furthermore, dose predictions do not necessarily correspond to other risk metrics—even those estimated for radionuclides (e.g., morbidity or mortality). The difficulty in comparison becomes even more problematic when other types of risk (e.g., standard industrial) must be factored into the remedial decision.

As illustrated in Table 50, the dose results for a contaminated site are often subsumed in those for the radiation morbidity and mortality risks predicted for the same exposures[175]. Furthermore, cancer incidence (i.e., morbidity) and fatality (i.e., mortality) predictions, albeit not the same, fall closer in meaning (than dose) to the standard injury and fatality risks often predicted for accidents related to typical occupational activities. The decision was made *for the purpose of this research* to compare trade-offs for different remedial decisions using the different types of risk factors (including chemical cancer risks) despite the different bases used to define them[176].

For the purpose of examining risk trade-offs for proposed remedial actions, a predicted cancer fatality (from radiation or chemical exposure) is *assumed* comparable to an accident fatality risk in that one has a finite likelihood to die (in terms of an accident) or might develop a fatal cancer over his or her lifetime (in terms of exposure to radiation or a chemical carcinogen). Non-fatal exposure risk (except to hazardous chemicals) and accident injury risk are also *assumed* comparable for the purpose of examining risk trade-offs. Risks posed by hazardous chemicals (typically represented by the hazard quotient) must be compared separately.

Comparisons of risk trade-offs are made despite their temporal differences. For example, latent cancer incidences and fatalities associated with contaminant exposure may take years to develop and the impact to the receptor is a function of when cancer develops and the years impacted or lost. On the other hand, impacts from accidents tend to be acute and measurable. One metric often considered to take these impacts into account is the years of potential life lost (YPLL) (CDC 1993; Cohen et al. 1997; Gilbert

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175 Like the dose, the morbidity and mortality results are obtained by multiplying the *exposure* by fixed constants defined by regulation. Thus the underlying, essential feature to dose or risk is the *exposure*.

176 Dose, morbidity, and mortality risks are obtained from dose-to-effect models that relate radiation doses to carcinogenic effects. For some radionuclides, there are data that are at or above the doses expected. On the other hand, chemical cancer effects are based on low-dose models with large uncertainties typically based either on animal studies or human data at doses much lower than expected.

et al. 1998; Romeder and McWhinnie 1977). Because adult receptors are assumed in this study, the YPLL would not add useful information. Future work should focus on expanding receptor scenarios to include sensitive and age-dependent receptors. This expansion should be supplemented with the ability to examine the YPLL or similar metric to better characterize the temporal nature of the risks involved.

SDA: Screening Quantitative Remedial Alternative Risk Evaluation

Figure 57 presents the predicted annual lifetime cancer incidence (morbidity) rate for expected baseline SDA conditions (i.e., *DBRA-Expected* in Table 49). As shown for the dose results (in Figure 53), cancer incidence risks for radionuclide exposures exceeded EPA limits (i.e., 10^{-4} to 10^{-6} cancer risk) at all times during the assessment period. Because of the unacceptable risks posed, the SDA would require remedial actions that involve, in general, managing the wastes in-place or retrieving wastes for treatment and disposal.

One typical basis for deciding on a remedial action is the degree to which hazards, doses, or risks would be averted to receptors at the site and its vicinity by completing the action. Often little thought is given outside the usual and customary occupational health and safety arena to the additional risks to remedial workers or, if wastes are treated and transferred elsewhere, to the resulting risk transfer or the risks (both to workers and the general public) associated with transporting the wastes from one site to another (Applegate and Wesloh 1998). That is, the life-cycle aspects of the waste disposition process are often not considered when making remedial decisions.

A more informed decision is made when all significant aspects of the buried waste disposition process are considered and the risks posed to all potentially impacted receptors by remedial actions are judged in the context of the risks averted or transferred (albeit likely to a more stable configuration). The screening risk tool described in Chapter VI can be used to estimate both the risks associated with proposed remedial actions and the residual risks presented by the contaminants remaining on the site.

In general, two remedial alternatives can be defined for buried waste sites: 1) manage the buried wastes in-place or 2) retrieve the wastes for treatment and disposal. There are many possible variations on these alternatives. One promising variation that was identified during a previous evaluation of SDA remedial actions was targeting retrieval actions based on the *highest-risk* wastes and the likely effectiveness of retrieving the wastes (Brown et al. 2005). It is possible that because of the intermixing of wastes and contaminant migration that the *highest-risk* wastes cannot be retrieved independently of other wastes. This potential issue must be considered in targeting wastes for retrieval. For example, potential retrieval areas in the SDA were identified based on the information for the pits and trenches in which wastes were originally buried. The effectiveness of any proposed retrieval process depends on whether contaminants have been released into the environment and how far the contaminants have migrated.

Predicted cancer incidence (morbidity) and fatality (mortality) results for the SDA were obtained from the screening risk tool for baseline; manage-in-place (MIP); and retrieve, treat, and dispose (RTD) alternatives. Figure 71 and Figure 72 illustrate the potential impact that the various manage-in-place options could have on exposure risks to the on-site receptor using morbidity and mortality, respectively. Three general MIP options are identified for the SDA depending on the possible use and extent of *in situ* grouting (ISG).

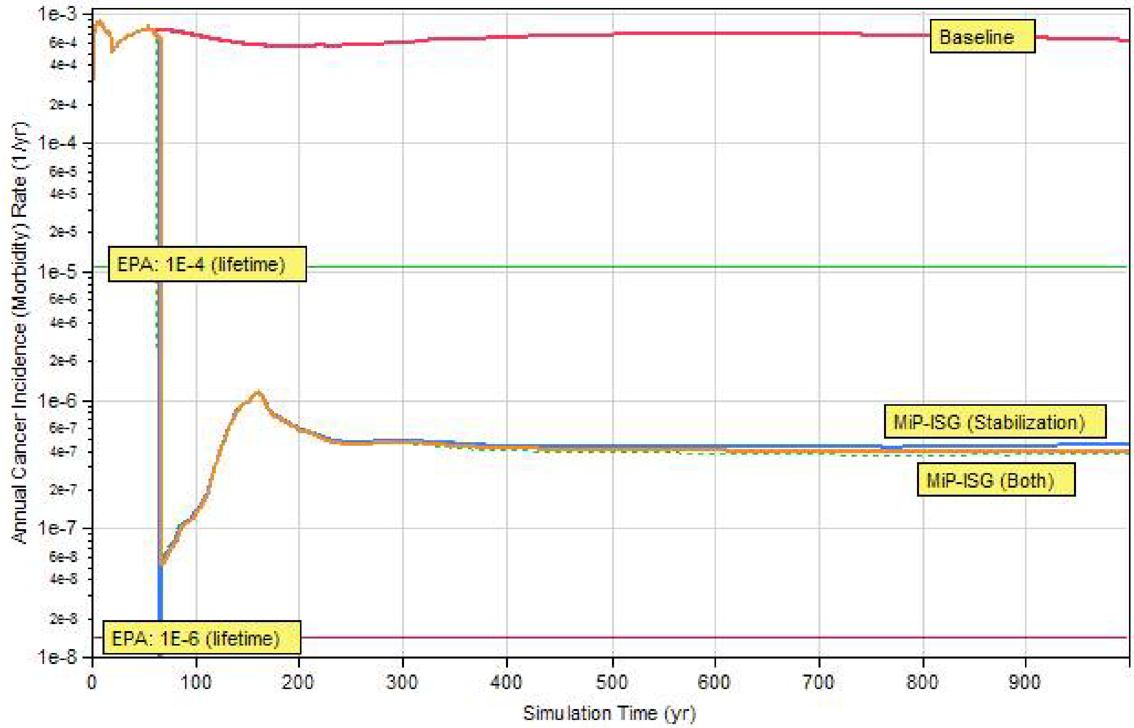

Figure 71. SDA *DBRA-Expected* Case for On-Site Resident: Solid Lines Represent Total Annual Cancer Incidence Rate (Morbidity) for Baseline and Manage-in-Place (MIP) Options (compared to EPA risk limits converted to annual bases). The dotted line represents the MIP with no *In Situ* Grouting (ISG).

Two of the *in situ* grouting (ISG) remedial options were 1) ISG for stabilizing the subsurface against subsidence or 2) ISG for both subsurface stabilization and contaminant immobilization. A third MIP option was considered where ISG was not used for either purpose. The results for this "no-ISG" option are illustrated by the sole dotted line in Figure 71 or Figure 72 and indicate that the assumptions pertaining to the ISG process step produce results that differ little, if any, in exposure risk reduction. These assumptions include rupturing of any containers remaining in the treated area and that the contaminant immobilization process when using ISG would be less than 100% efficient.

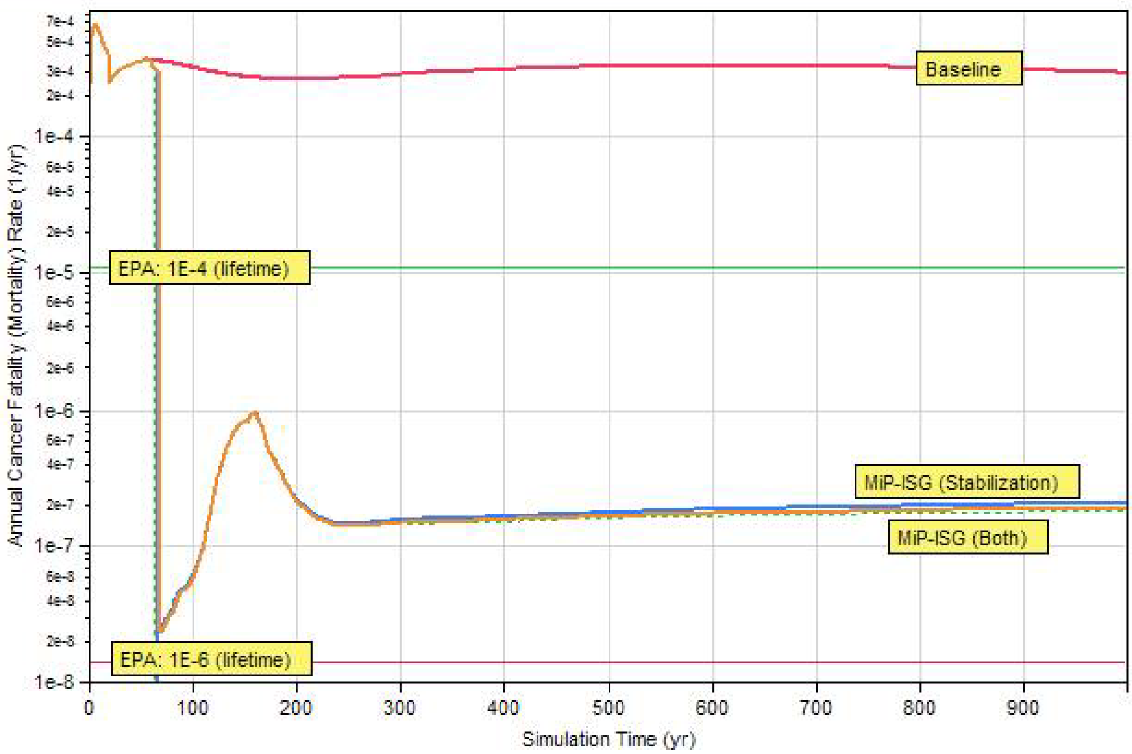

Figure 72. SDA *DBRA-Expected* Case for On-Site Resident: Solid Lines Represent Total Annual Cancer Fatality Rate (Mortality) for Baseline and Manage-in-Place (MIP) Options (compared to EPA cancer risk limits converted to annual bases). The dotted line represents the MIP option with no *In Situ* Grouting (ISG).

Because contaminants were predicted to have been released into and migrated through the environment surrounding the SDA before remedial actions were taken (which has been confirmed by environmental sampling and measurement (Holdren et al. 2006)), none of the MIP remedial actions appeared capable of providing a final, *unrestricted* release state for all contaminants and times based on the assumptions made. However, before proceeding with ISG at a site even for subsurface stabilization, the possible impacts on drum integrity and grouting efficiency should be evaluated to see if the model assumptions require updating.

The MIP options as illustrated in Figure 71 and Figure 72 were predicted to be capable of producing a final SDA state that would be protective (i.e., poses less risk than the EPA "action limit") for future restricted release of the area based on expected conditions[177]. This restricted final state of the SDA would match expectations that this area would remain classified as industrial and mixed-use acreage before and after the 100-year Institutional Control (IC) period expires (Holdren et al. 2006).

Because the MIP options studied cannot produce a final protective state for future *unrestricted* release of the SDA, other remedial options were considered for the site. Figure 73 illustrates the impact that the retrieve, treat, and dispose (RTD) remedial options would have on predicted cancer incidence rates[178] from exposures to site contaminants to the hypothetical on-site resident. The RTD options included targeted[179] and maximum retrieval cases based on the presence of wastes received from the Rocky Flats Plant (RFP) as detailed in Appendix D[180]. The RTD remedial options provided significant reductions in exposure risks over time to the on-site receptor (and thus other receptors). The most significant result in terms of exposure risks was that the maximum retrieval case appeared to present little benefit for the likely increased worker and general public risks associated with the increased excavation, retrieval, treatment, and off-site shipment activities.

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[177] The on-site resident is the most restrictive scenario for SDA conditions. Any alternative general public scenario (e.g., off-site resident, transient, etc.) will pose lower risks than those for the on-site resident.

[178] Because of the similarities between the morbidity and mortality results in Figure 71 and Figure 72, illustrations focus on morbidity although the analyses are performed for both radiation risk metrics.

[179] For the RTD targeted retrieval case, the areas not selected for retrieval are treated using *in situ* grouting.

[180] Because RFP wastes were distributed throughout the SDA, the maximum retrieval scenario, in essence, requires that wastes from nearly the entire site be retrieved for treatment and disposal in WIPP. This scenario is in agreement with that performed by Schofield (2002) when evaluating the short-term risks associated with SDA remedial actions.

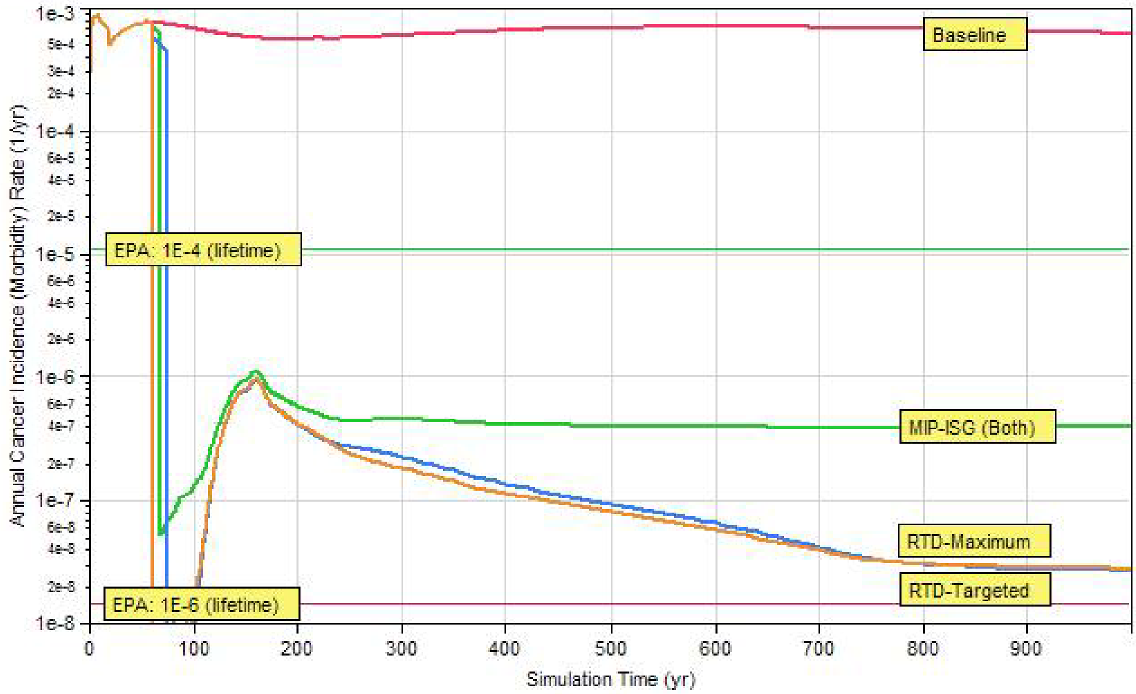

Figure 73. SDA *DBRA-Expected* Case for On-Site Resident: Annual Cancer Incidence Rate (Morbidity) for All Pathways Summed over all Radionuclides for Baseline and Retrieval Alternatives (compared to EPA risk limits). The Co-60 results were removed for clarification.

Another important result shown in Figure 73 was that, like the manage-in-place options, none of the retrieval scenarios were predicted to provide a final state that could be released for *unrestricted* use. The results presented thus far were based on best inventories and expected parameters. Thus there was no need to perform stochastic simulations to determine the likelihood that an unrestricted final state would be produced from proposed remedial actions (because it will be very low). Because of the nature of the contamination in and around the SDA (i.e., radioactive and volatile chemicals that have migrated 70 meters or more through the vadose zone), site restoration will be difficult. However, both the manage-in-place and retrieval alternatives were predicted to provide a final state protective for future *restricted* release of the SDA.

The remedial alternatives considered in this research were predicted to be capable of providing a final state posing acceptable risks[181] for the on-site resident during the assessment period. The predicted risks in Figure 71 through Figure 73, although based on expected inventory and parameters, should not be confused with actual risks that would be experienced by receptors in the vicinity of the SDA. The on-site resident scenario was selected to simplify the presentation of results and impacts although it necessarily overestimates expected risks. Depending on land-use assumptions, the site could theoretically be cleaned up to an acceptable (albeit "restricted") final state even considering that the risks predicted for the on-site resident were intentionally biased high.

SDA: Impact of the Conceptual Model on Predicted Risk Results

The results for the SDA baseline and remedial actions presented in this Chapter were based on the concept that, if colloids were created and transported unretarded through the vadose zone, they would be "screened out" by the Interbed Region. As illustrated in Appendix G when verifying the operation of the colloidal transport mode, this "screening out" of colloids has a profound impact on predicted groundwater exposure, dose, and risk results. The impact of varying the conceptual model and thus the manner in which transport is modeled will be explained through an example.

Figure 74 illustrates that predicted risks for the baseline and remedial alternatives were more than two orders of magnitude higher if colloids were not screened by the Interbed Region. Furthermore, no remedial option considered would be able to place the site in a final state that would be protective for even *restricted* use. However, it may be

[181] Although not shown here, the corresponding results for carcinogenic and non-carcinogenic risks for chemicals illustrate that the resulting risks will be lower than their respective limits. These results apply not only to the on-site resident scenario, but also to all the other receptor scenarios considered in this research.

comforting to note that several sedimentary interbed regions lie between the SDA and the aquifer below (that are modeled as a single region in this research) that would have to be ineffective at screening out colloids. If this were the case, it is likely that more than the current sporadic indications of plutonium would be detected in the monitoring wells.

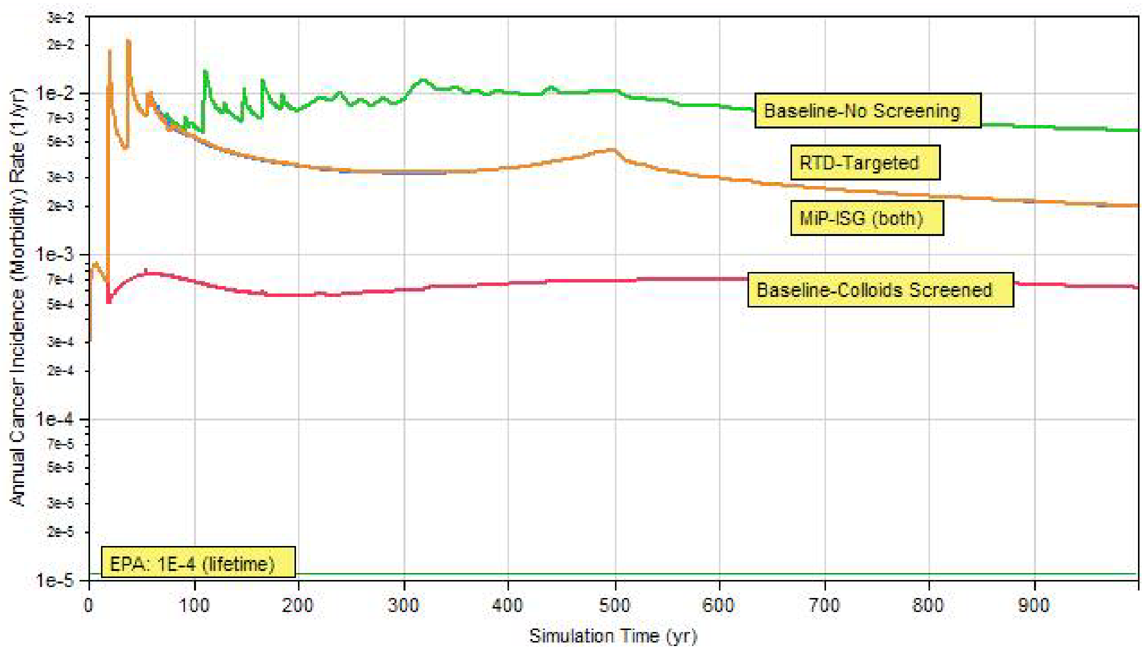

Figure 74. SDA *DBRA-Expected* Case for On-Site Resident: Annual Cancer Incidence Rate (Morbidity) for All Pathways and Radionuclides for Baseline, Manage-in-Place (using ISG for both Stabilization and Immobilization), and Targeted Retrieval (RTD) Alternatives Assuming No Screening of Colloids by the Interbed Region.

SDA: Exposure and Standard Industrial Risks for Workers

As suggested in the risk analysis framework and methodology in Chapter III, the primary foci for determining whether a buried waste site poses unacceptable risks are the short- and long-term exposure impacts to potential receptors. The focus in this section has been on the on-site resident scenario designed to overpredict exposure risks to members of the general public. However, the general public will not be the only ones potentially impacted by wastes buried in the site. The potential impacts to workers must also be considered when comprehensively evaluating the risks posed by the site. In much the same way that the on-site resident was selected to overpredict exposure risks for members of the general public, the direct worker will be used to represent the workforce.

Figure 75 illustrates, for the direct worker scenario, the predicted annual cancer incidence (morbidity) rate[182] for baseline (including both worker and on-site resident for scale) and selected manage-in-place and retrieval alternatives. The results for the time until the Institutional Control (IC) period ends (i.e., Year 160) are shown because the workers (including direct, support, and remedial) were assumed to be no longer on the site. From these results, the direct worker scenario translated into lower predicted exposure risks than for the corresponding on-site resident scenario by just under an order of magnitude. These results applied to the other risk metrics (i.e., mortality, dose, and chemical cancer and non-cancer effects) because exposure was the driver for these risks.

However, *lower* exposure risk does not mean that workers would not be at risk from multiple contaminant sources via multiple transport pathways (albeit excluding groundwater effects) and exposure routes like the on-site resident. These exposure risks are likely to be magnified and increased by additional standard industrial risks (e.g., slips, trips, etc.) when remedial actions are taken at the site.

[182] For these results, the impact of Co-60 is not removed from total morbidity prediction because workers would have been subjected to these impacts. Thus the baseline results may differ substantially until the Co-60 decays significantly.

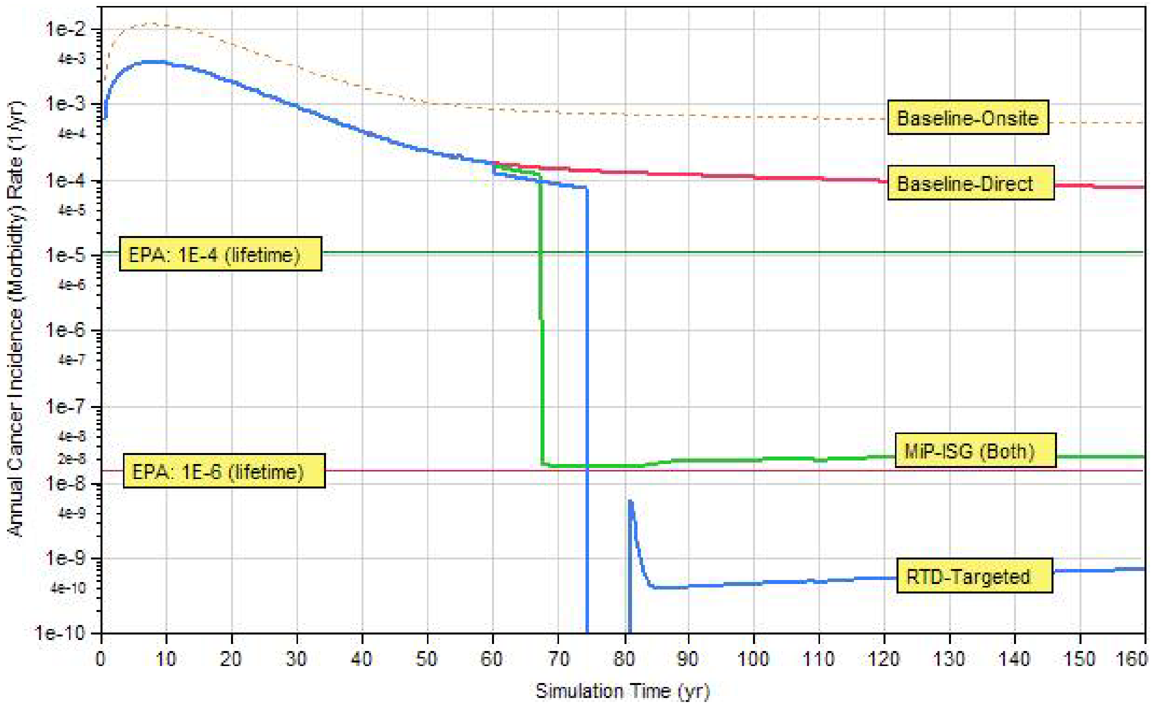

Figure 75. SDA *DBRA-Expected* Case for Direct Worker: Annual Cancer Incidence Rate (Morbidity) for All Pathways and Radionuclides for Baseline, Manage-in-Place, and Retrieval Alternatives (compared to EPA risk limits and Baseline On-Site Resident Risk to indicate Relative Magnitudes of Risks).

Figure 76 through Figure 81 present the expected standard industrial risks for remedial workers for the three manage-in-place (MIP) remedial options considered in this research during the time period up to the initial barrier repair action for clarity. The process step numbers correspond to those evaluated for the SDA and BCBG remedial alternatives in Chapter IV and Appendix A and Appendix B. Table 51 provides a summary of the process steps identified in the various figures to follow where substeps (e.g., excavation of overburden, retrieval of wastes, etc.) are numbered to prevent confusion, and an 11th process step (i.e., major surface barrier repair) was added; risks and probabilities for the new repair step were computed using information from previous steps.

Table 51. Process Steps for Proposed Remedial Alternatives

Process Step	Baseline	MIP	RTD
0. **Routine Work**	√		
1. **Burial Site Characterization**	√	√	√
2. ***In Situ* Grouting (ISG) for Subsidence Control**		√	√
3. **ISG for Subsidence Control and Immobilization**		√	√
4. **Excavate, Retrieve & Segregate** 4a. Excavation of Soil Overburden			√
4b. Retrieval of Buried Wastes			√
4c. Excavation of Soil Underburden			√
5. ***Ex Situ* Treatment**			√
6. **Package Retrieved Wastes**			√
7. **Storage and On-Site Disposal** 7a. Internment of Soil Overburden			√
7b. Return Non-TRU/Non-HLW Wastes to Burial Site			√
7c. Place Clean Soil Overburden			√
8. **Surface Barrier Installation/Repair**		√	√
9. **Long-term Stewardship** Monitor, Maintain, and Repair	√	√	√
10. **Off-Site Shipment and Disposal at WIPP**			√
11. **Major Repair of Installed Surface Barrier**		√	√

The standard industrial injury and fatality risks for the manage-in-place (MIP) remedial options eschewing *in situ* grouting (ISG) for either subsidence control or contaminant immobilization are described in Figure 76 and Figure 77, respectively. These remedial worker results provide a baseline for accident risks because no intrusive remedial actions were taken. The "background" worker risks in the diagrams to follow represent the annual injury or fatality risk for the corresponding direct or support worker. For example, the results for direct workers (from the exposure scenario) are compared to the direct remedial workers in Figure 76 and Figure 77. The remedial worker risks and probabilities were computed for the time frame required to perform the remedial action.

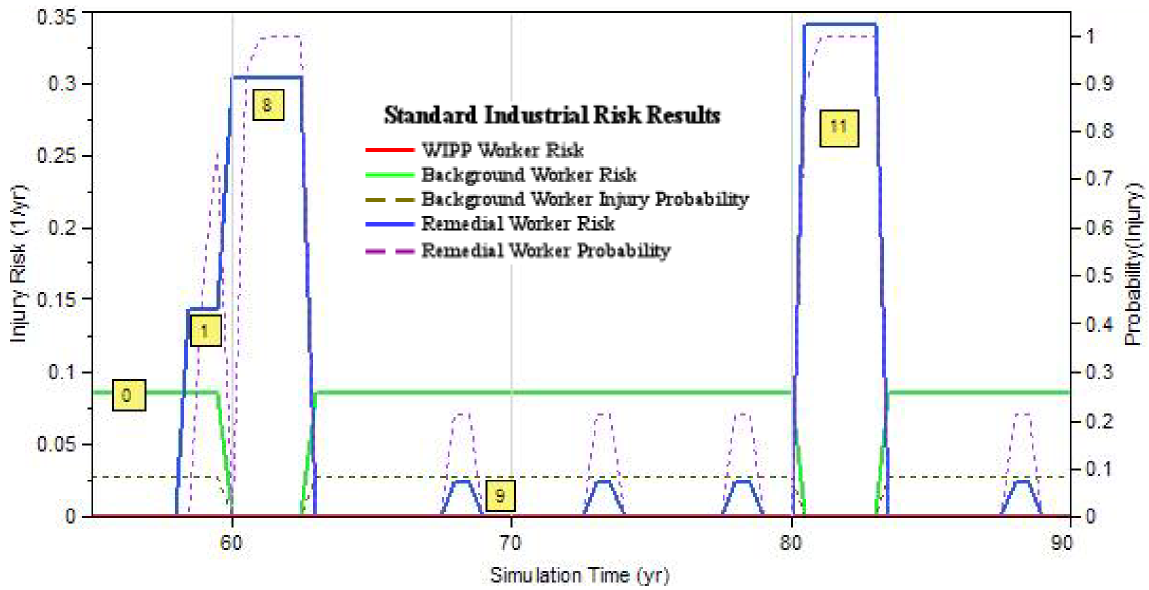

Figure 76. SDA *DBRA-Expected* for Direct Remedial Workers: Standard Industrial Injury Risks and Probabilities for the Manage-in-Place (MIP) Scenario with No *In Situ* Grouting (ISG). (Steps correspond to those in Table 51.)

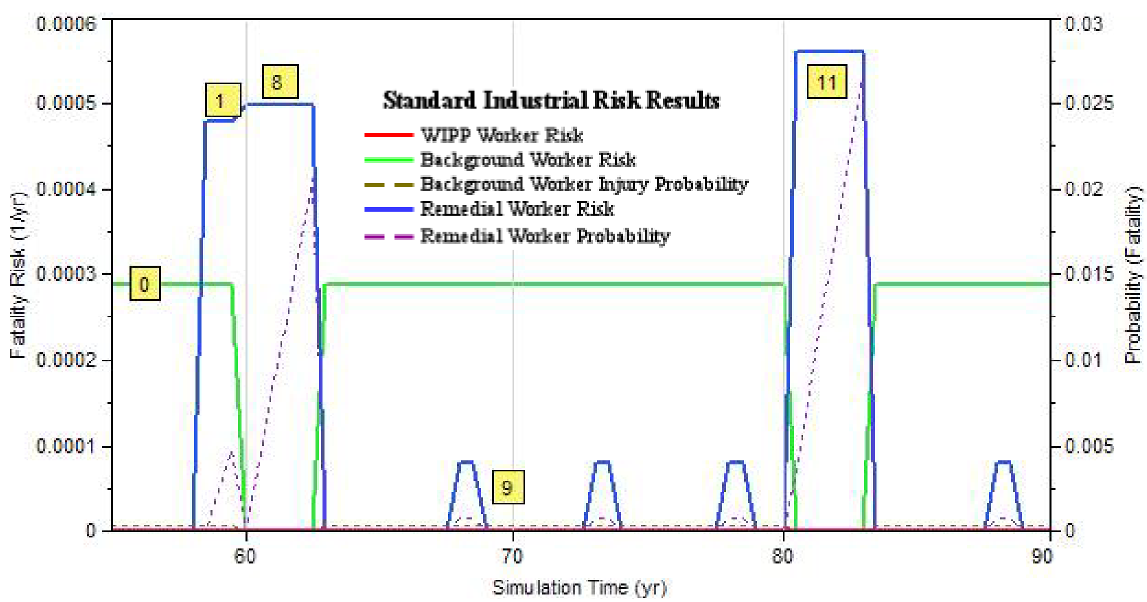

Figure 77. SDA *DBRA-Expected* for Direct Remedial Workers: Standard Industrial Fatality Risks and Probabilities for Manage-in-Place (MIP) Scenario with No *In Situ* Grouting (ISG). (Steps correspond to those in Table 51.)

The results in Figure 76 and Figure 77 indicate that remedial action risks (even for the least aggressive of the remedial actions considered for the SDA) were higher than the "background" worker risks by a factor of three to four[183]. On the other hand, maintenance and long-term stewardship activities appeared less risky because their durations were considerably less than one year (i.e., the standard used for "background" risk)[184]. In terms of the probabilities associated with the different types of effects, injuries during remedial and major barrier repair activities tended to be very likely (i.e., with a probability approaching unity for the major steps) and were as much as an order of magnitude higher than the probability associated with "background" injury risks (which tended to be less than 0.1). As illustrated in Figure 77, fatality risks for remedial workers were about twice as high as "background risks"; however, the probability of a fatality was approximately two orders of magnitude higher than that for corresponding "background" risk. The maximum predicted probability for remedial worker fatality risks was still less than 0.03.

To evaluate how the aggressiveness of a proposed remedial action might impact the resulting predicted standard industrial risks (where additional information was provided in Appendix G), the manage-in-place (MIP) alternative employing *in situ* grouting (ISG) in all three Waste Areas for contaminant immobilization was examined. The standard industrial injury and fatality risks for this MIP remedial option are described in Figure 78 and Figure 79, respectively. The three ISG steps indicated on these figures correspond to the three Waste Areas treated.

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[183] The significance of the higher risks are examined subsequently when the uncertainties in the risk results are examined based upon stochastic results using the screening risk tool.

[184] Appendix G provides additional information on the various impacts of the selected remedial options on the remedial worker standard industrial risks.

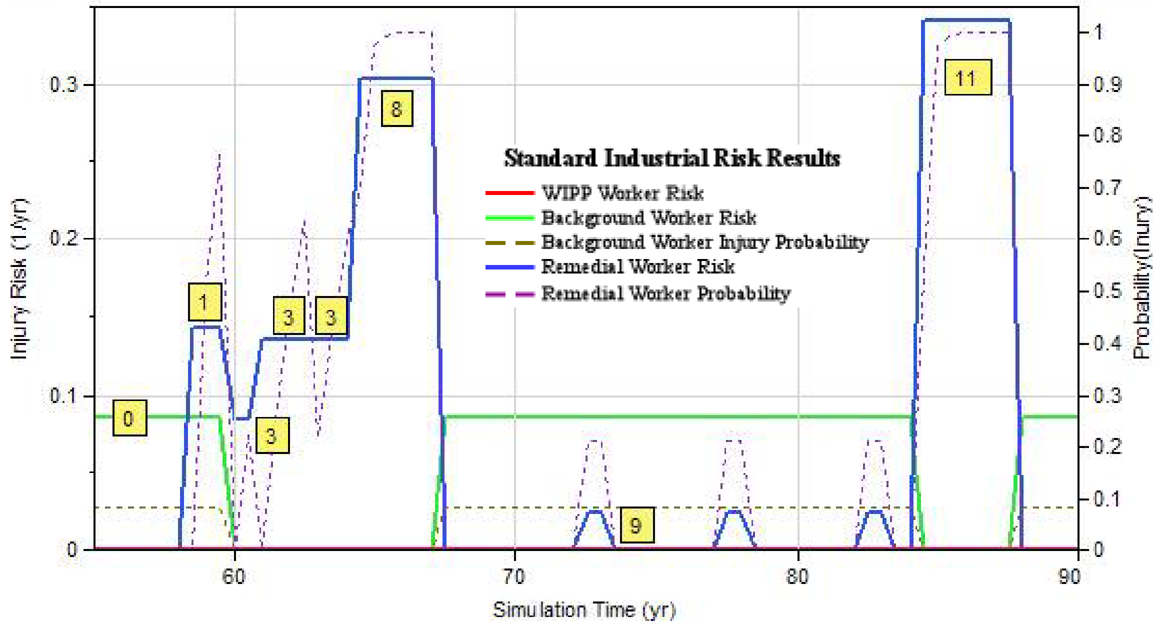

Figure 78. SDA *DBRA-Expected* for Direct Remedial Workers: Standard Industrial Injury Risks and Probabilities for Manage-in-Place (MIP) Scenario with *In Situ* Grouting (ISG) used for both Subsidence Control and Contaminant Immobilization. (Steps correspond to those in Table 51.)

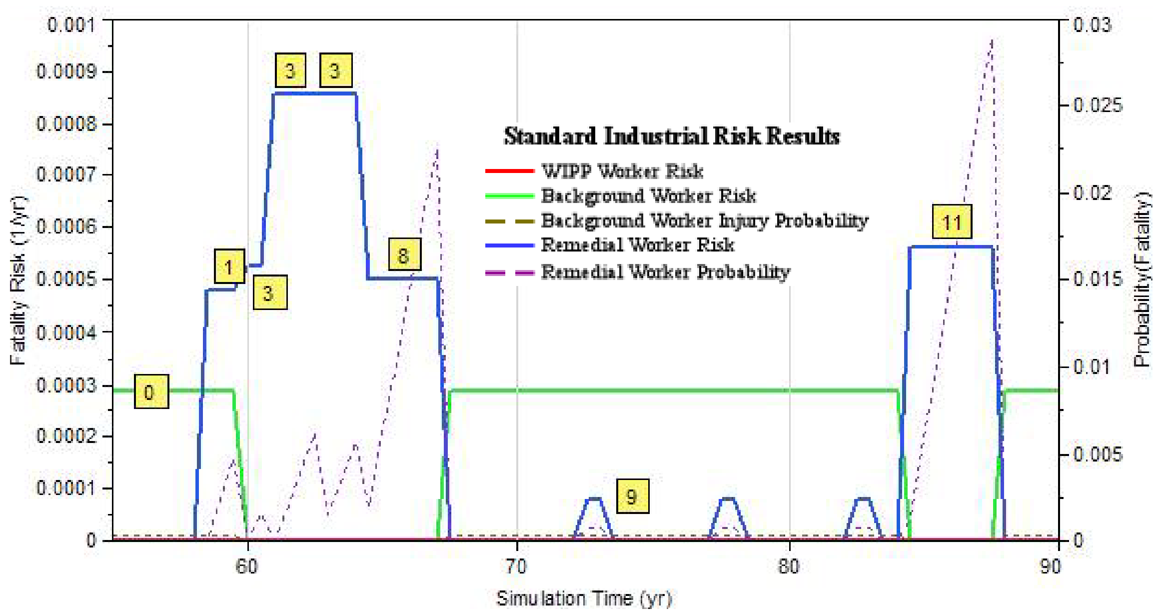

Figure 79. SDA *DBRA-Expected* for Direct Workers: Standard Industrial Fatality Risks and Probabilities for Manage-in-Place (MIP) Scenario with *In Situ* Grouting (ISG) used for both Subsidence Control and Contaminant Immobilization. (Steps correspond to those in Table 51.)

For the remedial injury risks illustrated in Figure 78, the impact from adding the *in situ* grouting (ISG) steps to both control subsidence and immobilize contaminants in the three SDA Waste Areas was not very dramatic. The injury risks for the additional ISG steps were no worse than either the "background" (for Waste Area 01) or the burial site characterization step (for Waste Areas 02 and 03). On the other hand, a much more dramatic impact was discovered when examining the predicted remedial worker fatality risks (Figure 79) associated with the additional ISG steps. For these conditions, the fatality risks associated with ISG were approximately twice those of the other process steps and almost thrice that of the "background" risks. The fatality risk probabilities for the ISG steps were considerably higher than those for the corresponding "background" risks to non-remedial workers; however, again the predicted probabilities were relatively low (i.e., less than 0.01).

Therefore, the aggressiveness of proposed remedial actions may have a large influence on the standard industrial injury and fatality risks predicted for remedial workers. The retrieve, treat, and dispose (RTD) alternative included not only *in situ* grouting (ISG), which posed relatively large fatality risks for the manage-in-place (MIP) alternatives, but also the most aggressive of the remedial actions proposed for the SDA disposition (i.e., excavation and retrieval). From the verification analyses presented in Appendix G, the maximum RTD option was likely to pose the largest risks to workers over the longest time. The standard industrial injury and fatality risks for the maximum RTD remedial option are described in Figure 80 and Figure 81, respectively.

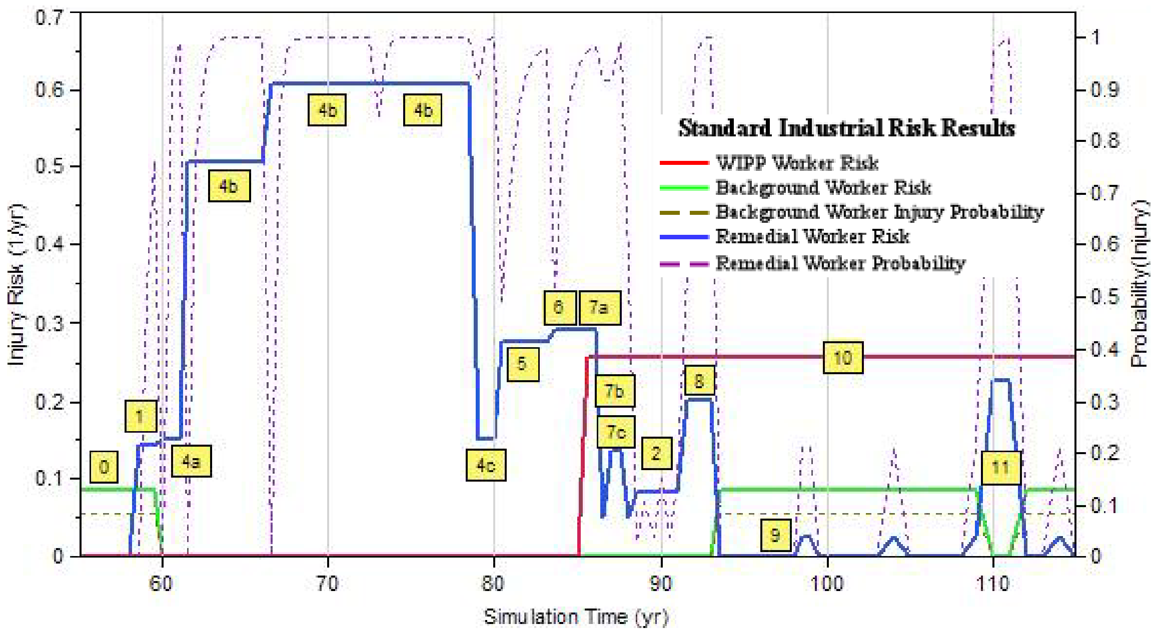

Figure 80. SDA *DBRA-Expected* for Remedial Workers: Standard Industrial Injury Risks and Probabilities for the Maximum Retrieve, Treat, Dispose (RTD) Scenario. (Steps correspond to those in Table 51.)

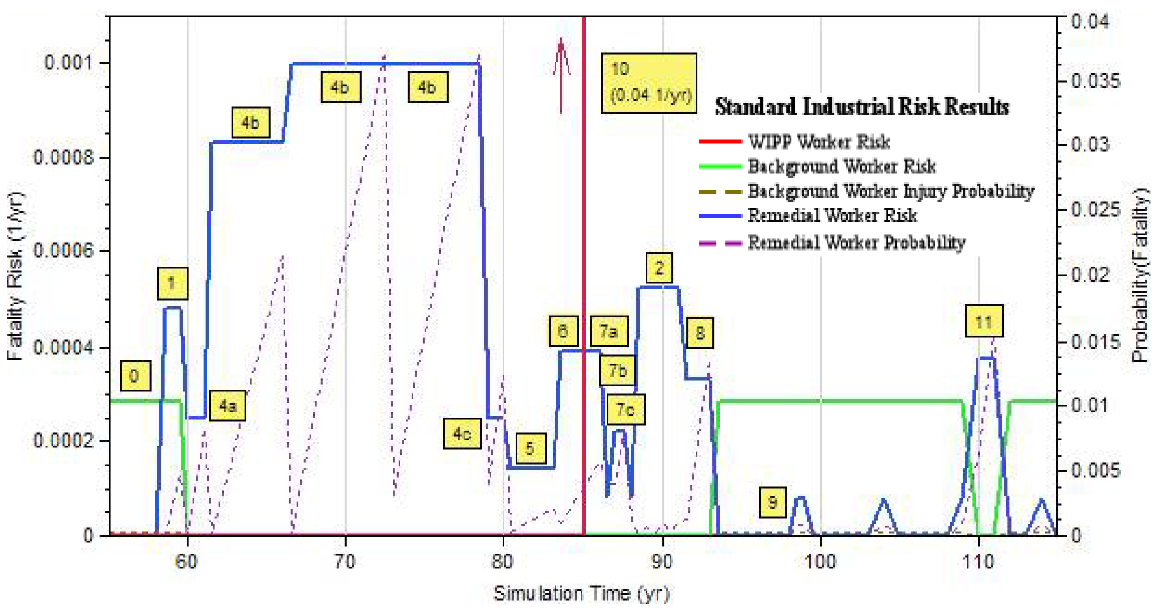

Figure 81. SDA *DBRA-Expected* for Remedial Workers: Standard Industrial Fatality Risks and Probabilities for the Maximum Retrieve, Treat, Dispose (RTD) Scenario. The fatality risk for Off-Site Shipments to WIPP (Step 10) is more than 40 times larger than any of the fatality risks shown. (Steps correspond to those in Table 51.)

For the remedial injury risks shown in Figure 80, the impact from using both retrieval and *in situ* grouting (for subsidence control) was large only for retrieval actions (i.e., more than a factor of six greater than "background"). Again, remedial worker *injury* risks for the ISG steps tended to be less than "background" and the probabilities were correspondingly higher. On the other hand, dramatic impacts were found for the remedial worker *fatality* risks (illustrated in Figure 81) for both ISG and retrieval steps. The fatality risk probabilities associated with retrieval were considerably higher than those for the corresponding "background" risks; however, the probabilities were again relatively small (i.e., less than 0.04). Fatalities during ISG operations appeared relatively unlikely when compared to the fatality risks for the other remedial actions.

However, the real impact on predicted remedial worker risk resulted from shipping transuranic (TRU) wastes from the SDA to the Waste Isolation Pilot Plant (WIPP). For example, fatality risks were more than 40 times greater than those for any of the other steps and two orders of magnitude greater than "background" risks. Because of the large number of shipments that would be required to transport the TRU wastes to WIPP, the probability for these risks were expected to approach unity.

SDA: Uncertainties in Exposure and Standard Industrial Risks

For purposes of clarification, the results that have been presented were primarily based on "deterministic" or point-value analyses using best inventories and expected parameters supplemented by limited evaluations to demonstrate the profound impacts of the assumptions made on model predictions. These results bracketed the baseline exposure risks from the SDA and the potential impacts of proposed remedial actions on future exposure risks to the general public. These results were supplemented by examining, using the screening risk tool and the same models, assumptions, etc., the exposure and accident risks for workers during both normal and remedial operations.

A number of important results became evident from the evaluations based on best inventory and expected parameter results:

1) It is *likely* that either manage-in-place (MIP) or retrieval (RTD) alternatives could place the SDA into a final state protective for *restricted* release.

2) It is *unlikely* that any of the proposed remedial alternatives could place the SDA into a sufficiently protective final state for *unrestricted* release.

3) The proposed remedial actions were *likely* to increase injury and fatality risks to workers and, perhaps, the general public.

4) The more aggressive the remedial action, the more worker risk *likely* to result.

These conclusions are unlikely to change if the uncertainty analyses performed in Chapter IV and Appendix A were expanded. However, because there was no clear remedial alternative for the SDA on a risk basis, a screening quantitative uncertainty analysis appears warranted to help limit future risk and uncertainty analyses to those remedial actions and parameters that are most likely to impact the remedial decision.

The screening risk tool developed in this research can be used for point-value evaluations of exposure and accident risks as presented in this chapter. However, because the tool was developed in the GoldSim Monte Carlo simulation software, stochastic evaluations of the same risks can be made to characterize the uncertainties in the various doses, risks, and hazards predicted for potential receptors. The uncertainty results are structured like the risk analysis framework defined in Chapter III. However, because the uncertainty results themselves are unlikely to dramatically impact the remedial decision, a brief evaluation of the uncertainties in the risk metrics is provided. Results were based on 50 realizations for each uncertainty analysis.

The initial evaluation examined whether or not the baseline conditions for the SDA pose unacceptable risks to potentially impacted receptors. The first risk metric evaluated was the total effective dose equivalent (TEDE) for the on-site resident evaluated for all pathways and radionuclides as illustrated in Figure 53. The uncertainty in the predicted dose for the 1,000-yr assessment period is illustrated in Figure 82. The uncertainty moments including the mean and mode and bounds including the 95% and the upper and lower bounds are compared to the results for the *DBRA-Expected* and *DBRA-Maximum* Cases described in Table 49. These results demonstrate that the point-value results obtained earlier do, in fact, bracket the upper-bound estimates of exposure risks as expected; however, use of the point-value estimates for bounding assumptions dramatically overestimates the exposure risks.

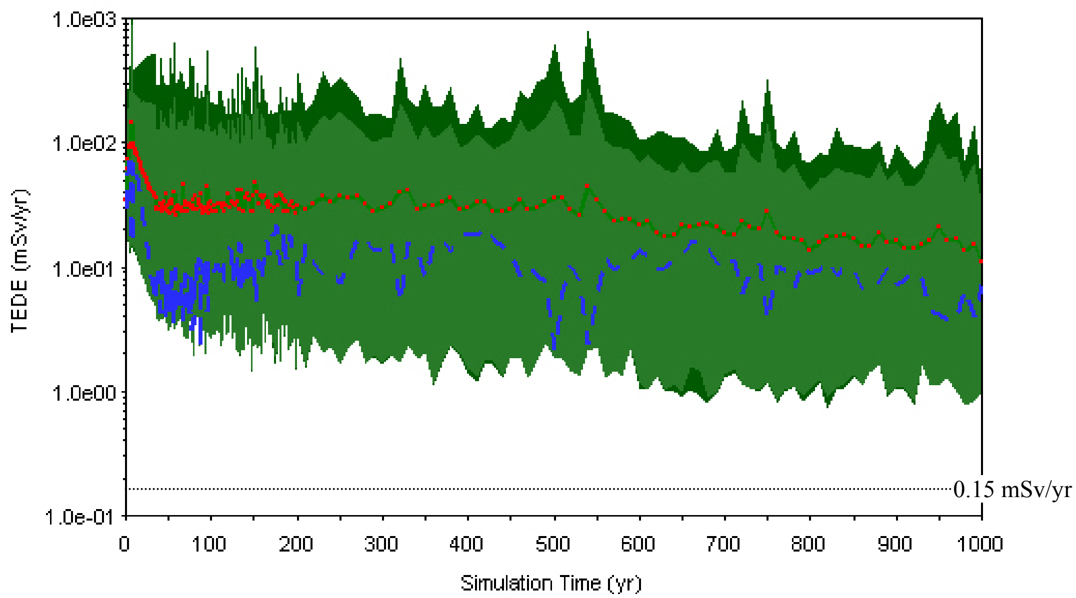

Figure 82. SDA On-Site Resident Scenario: Uncertainties in the Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides on the Same Scale as Figure 53. The red dotted line is the median, blue hashed line the mean, and other bounds represent 95% bounds and upper and lower bounds.

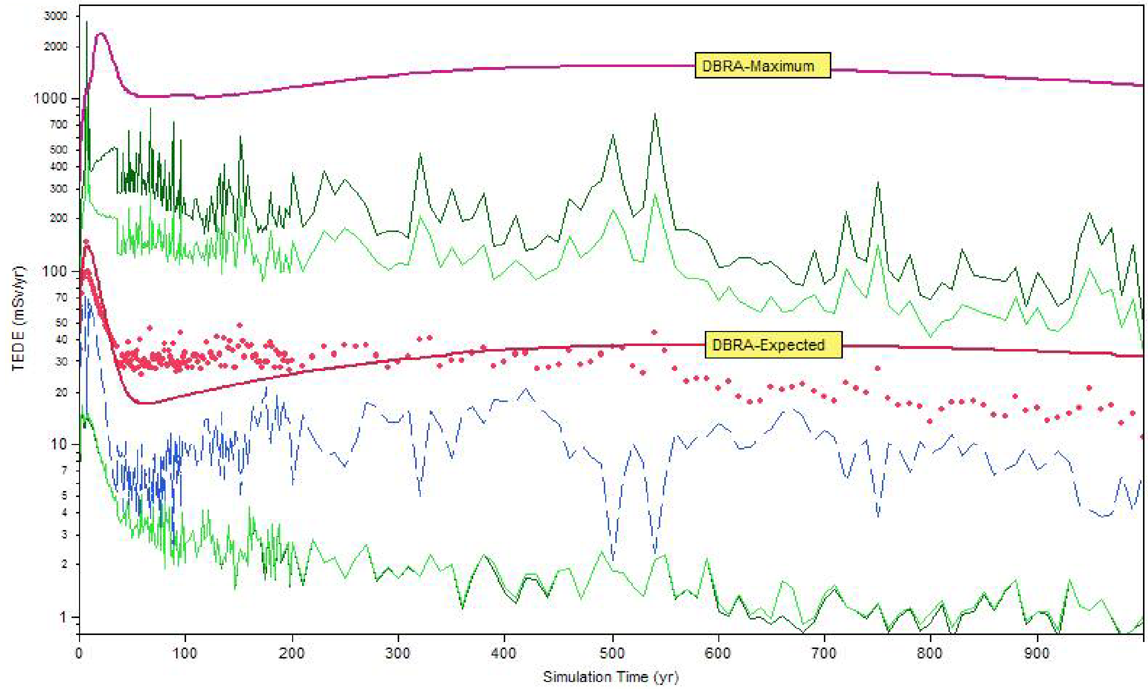

Figure 83. SDA On-Site Resident Scenario: Uncertainty Bounds in the Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides from Figure 82 Compared to Previous Results from the *DBRA-Expected* and -*Maximum* Cases. The red dotted line is the median, blue hashed line the mean, and other bounds represent 95% bounds and upper and lower bounds.

The complementary cumulative distribution function (CCDF) or exceedance curve for the peak dose predictions is illustrated in Figure 84. From these results, the probability that the peak TEDE for the on-site resident exceeded the proposed EPA limit of 0.15 mSv/yr was 100%. Because of the temporal nature of the risks associated with a contaminated site, it is suggested that both temporal and exceedance curves be provided. Not surprisingly, there was a great deal of uncertainty in the predicted dose; however, the lower bound did not intersect the proposed EPA limit of 0.15 mSv/yr at any time during the assessment period illustrating the unacceptable nature of the radiation risks posed by the SDA contaminants.

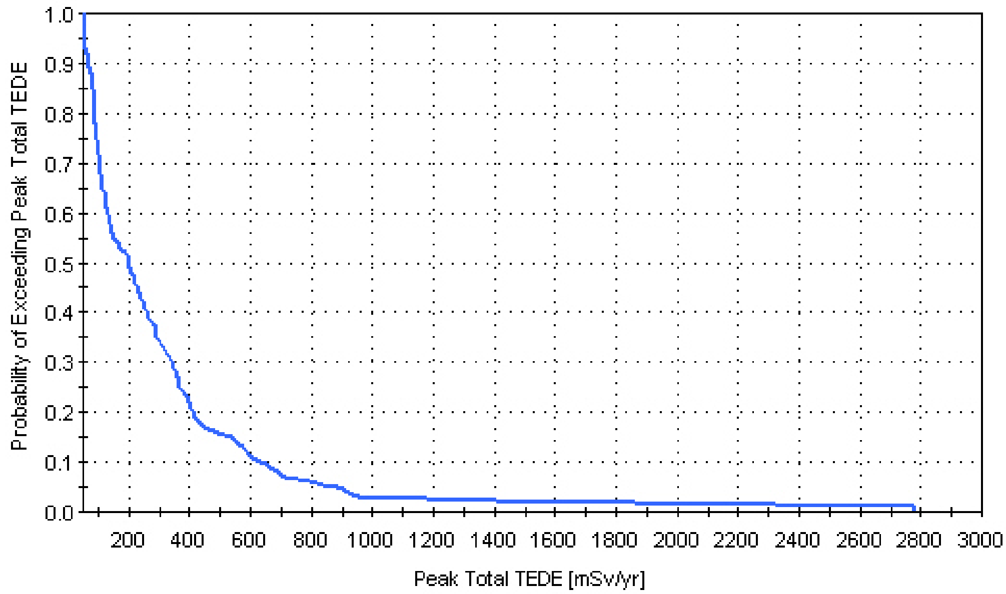

Figure 84. SDA On-Site Resident Scenario: Exceedance Curve in blue for the Annual Total Effective Dose Equivalent (TEDE) for All Pathways and Radionuclides (The TEDE values corresponding to $p = 0.05$ and $p = 0.5$ are 880 and 200 mSv/yr, respectively. The proposed EPA limit of 0.15 mSv/yr essentially coincides with the ordinate.)

The cancer incidence (i.e., morbidity) predictions used to evaluate exposure risks and the impacts of remedial actions throughout this chapter also exhibited large uncertainties as illustrated in Figure 85 and Figure 86. Despite the large uncertainties in the predicted morbidity risks, the lower prediction bounds remained greater than the EPA "action limit" corresponding to 10^{-4} cancer risk during the 1,000-yr assessment period. From the exceedance curve, there was a 100% chance that the peak risk exceeded the "action limit." Because of the scale selected for Figure 85 (i.e., to correspond to Figure 57 showing both EPA cancer risk limits), the uncertainties in predicted total doses appear smaller than they are. For example, the standard deviation in the predicted doses for each year in the simulation tended to be more than 100% of the mean value for that year.

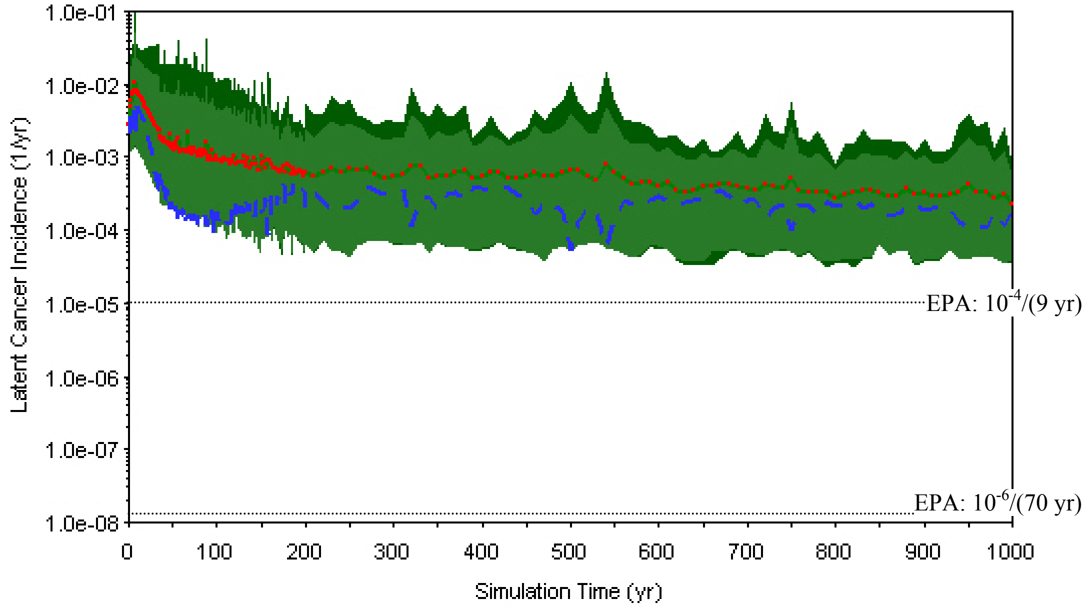

Figure 85. SDA On-Site Resident Scenario: Uncertainties in the Annual Morbidity Rate for All Pathways and Radionuclides using the Dimensions from Figure 57. The red dotted line is the median, blue hashed line the mean, and the other bounds represent the 95% bounds and upper and lower bounds.

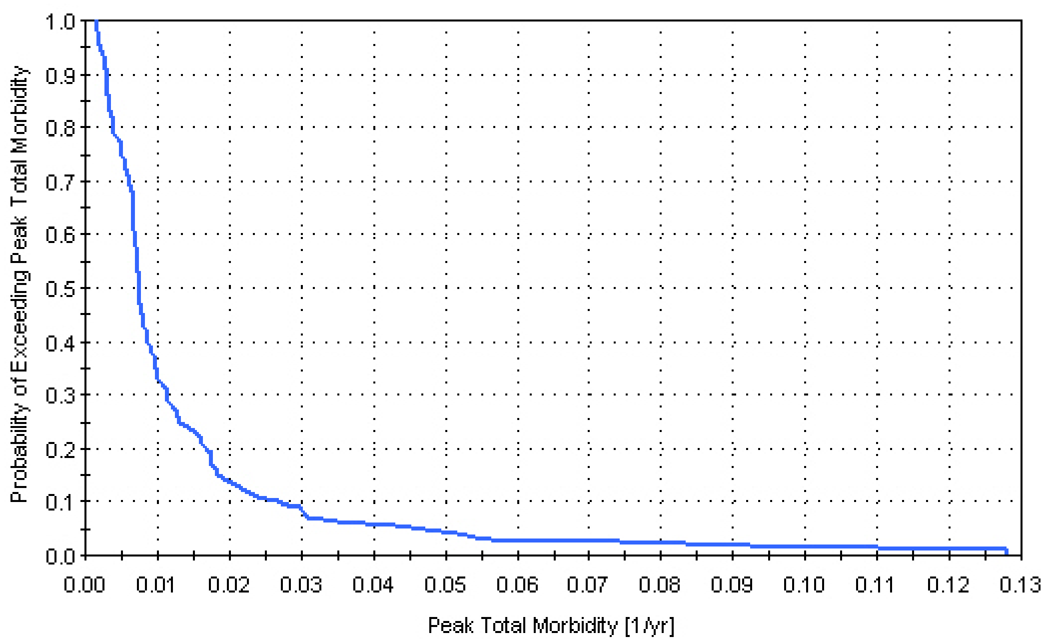

Figure 86. SDA On-Site Resident Scenario: Exceedance Curve in blue for the Annual Morbidity Rate for All Pathways and Radionuclides. (The morbidity values corresponding to $p = 0.05$ and $p = 0.5$ are $5x10^{-2}$ and $8x10^{-3}$ 1/yr, respectively compared to the EPA "action limit" of 10^{-4}/(9 yr) = $1.1x10^{-5}$ 1/yr.)

The uncertainties in morbidity predictions (like those for TEDE) were relatively small when compared to the uncertainties in predicted chemical cancer risks as shown in Figure 87 (which are shown on the same scale as the point-value results from Figure 60). The uncertainties shown on a graph with an ordinate expanded to cover the results (for 50 realizations) and the exceedance curve for the chemical cancer risk are provided in Figure 88 and Figure 89, respectively. From the exceedance curve, there was over a 60% chance that the peak chemical cancer risk exceeded the EPA "action limit." Thus the predicted cancer risks for the SDA chemicals appear less problematic than those for radionuclides. A significant number of the values even fell below the EPA *de minimus* of 10^{-6} cancer risk (as shown in Figure 88) unlike those for the radionuclide risk estimates.

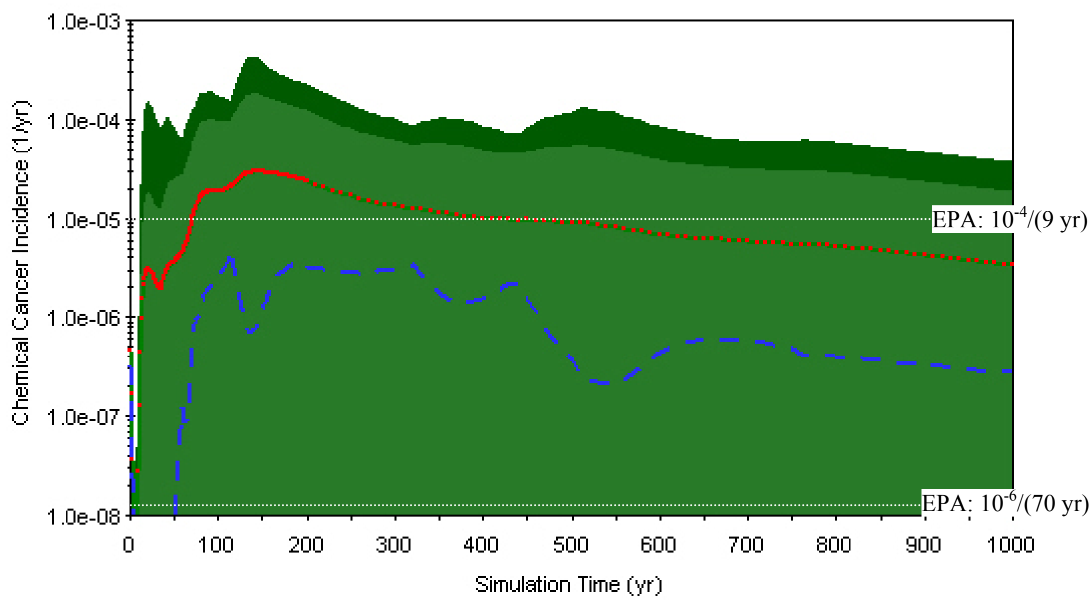

Figure 87. SDA On-Site Resident Scenario: Uncertainties in the Annual Cancer Risk for All Pathways and Chemicals on the Same Scale as Figure 60. The red dotted line is the median, blue hashed line the mean, and the other bounds represent 95% and the upper and lower bounds.

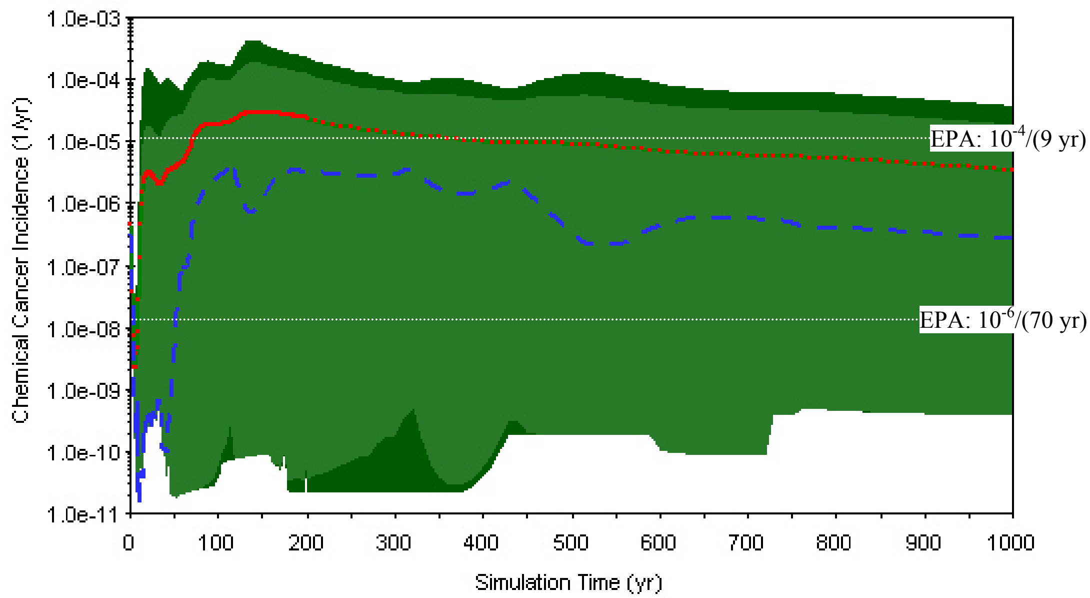

Figure 88. SDA On-Site Resident Scenario: Uncertainties in the Annual Cancer Risk for All Pathways and Chemicals from Figure 87 shown on an Expanded Scale. The red dotted line is the median, blue hashed line the mean, and the other bounds represent the 95% and the upper and lower bounds.

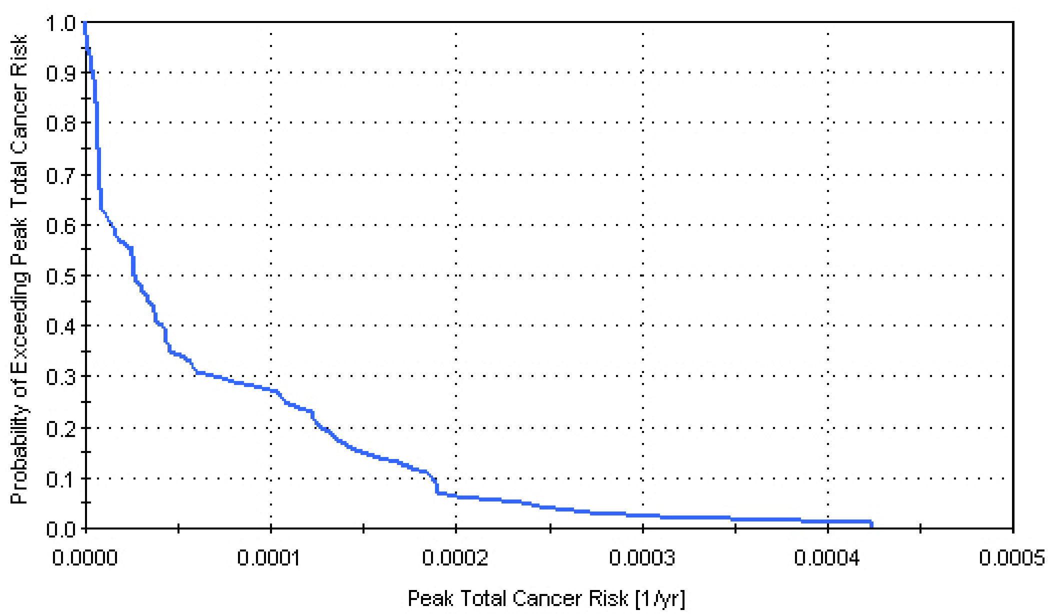

Figure 89. SDA On-Site Resident Scenario: Exceedance Curve in blue for the Annual Cancer Risk for All Pathways and Chemicals. (The cancer risk values corresponding to p = 0.05 and p = 0.5 are $2x10^{-4}$ and $3x10^{-5}$ 1/yr, respectively compared to the EPA "action limit" of 10^{-4}/(9 yr) = $1.1x10^{-5}$ 1/yr.)

Figure 82 through Figure 89 describe the uncertainties in the risk metrics predicted for on-site exposures to the wastes buried in the SDA over the 1,000-year assessment period. Despite large uncertainties in the predicted risk metrics, there is little doubt that site conditions would pose unacceptable risks to the general public. Remedial actions will be required to place the site in a final state protective of human health. The remedial alternatives proposed for the SDA were divided into those that manage the buried wastes in-place and those that retrieve buried wastes for treatment and disposal.

The uncertainties in the predicted morbidity rates for the SDA manage-in-place and targeted retrieval alternatives (both without employing *in situ* grouting (ISG) for easier comparison) are illustrated in Figure 90 and Figure 91, respectively. The corresponding exceedance curves are not shown because they changed little for the region near the limits from that for the SDA baseline conditions (i.e., Figure 86) because the largest peak morbidity values tended to be manifested before remedial actions began.

For the manage-in-place alternative, the majority of the post-closure predicted annual morbidity values fell between the EPA limits corresponding to 10^{-6} and 10^{-4} cancer risk. As shown in Figure 73 and Figure 91, the retrieval alternative produced significantly lower predicted morbidity risks than the manage-in-place option. This risk reduction was also manifested for other risk metrics and receptors. However, the predicted morbidity risks for retrieval were not significantly below the EPA *de minimus* limit (although the mean value did fall below the limit near Year 800). Thus the targeted retrieval alternative did not appear to provide a tangible benefit especially considering the likely increased worker and general public risks associated with excavation, retrieval, treatment, and off-site shipment activities.

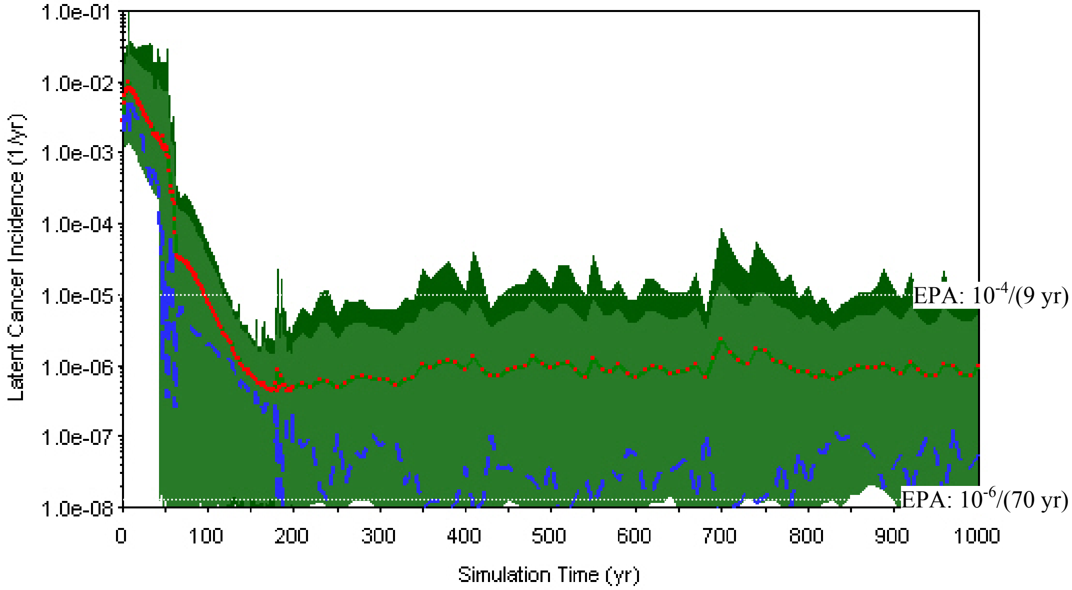

Figure 90. SDA On-Site Resident Scenario: Uncertainties in the Annual Morbidity Rate for All Pathways and Radionuclides for the Manage-In-Place Option (No *In Situ* Grouting) on the Same Scale as Figure 57 and Figure 85. The red dotted line is the median, blue hashed line the mean, and the other bounds represent the 95% and the upper and lower bounds.

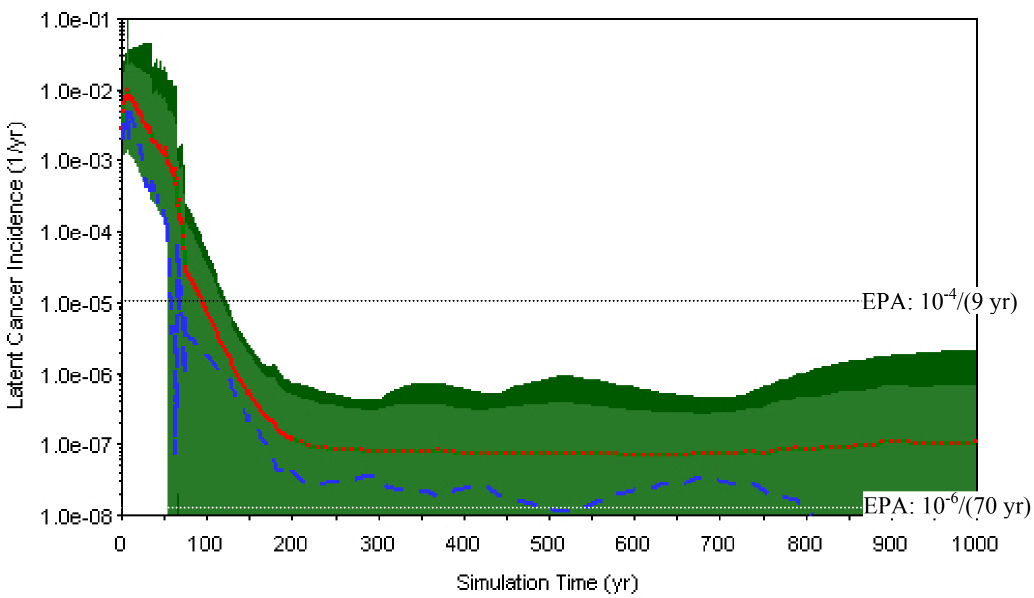

Figure 91. SDA On-Site Resident Scenario: Uncertainties in the Annual Morbidity Rate for the Targeted Retrieval Option (No *In Situ* Grouting) on the Same Scale as Figure 57, Figure 85, and Figure 90. The red dotted line is the median, blue hashed line the mean, and the other bounds represent the 95% and the upper and lower bounds.

SDA: Sensitivity Analyses

Large uncertainties were associated with the predicted exposure, dose, and risk estimates for both baseline conditions and the disposition of SDA wastes. Although these results were not unexpected, it is often useful to identify the uncertain parameters that have the largest impacts on the uncertainties in results. These influential uncertain parameters are those that should be investigated to determine if more representative (e.g., site-specific, accurate, etc.) information is available.

The sensitivity analysis for the SDA was executed in two stages using the "tornado" feature in the GoldSim software (GTG 2005b). This feature performs one-at-a-time sensitivity analyses for independent variables and presents the results graphically. While holding the other independent variables at their central values, three runs are made for the selected variable at the lower bound, central value and the upper bound. The results are arranged from top to bottom based on the range of the results.

Independent variables were selected to represent the transport pathways likely to impact the movement of contaminants in and around the SDA[185]. The sensitivity analysis results for selected transport parameters based on the total effective dose equivalent (TEDE) are shown in Figure 92. The next step was to keep the most influential transport parameters and then add variables representing human activities that impact exposure, dose, and risk predictions. The results for twenty of the most influential SDA parameters are shown in Figure 93. The eight most influential independent variables for the TEDE response analysis represented impacts from airborne, biotic, and waterborne transport pathways.

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[185] Partition coefficients, which are already known to impact exposure, dose, and risk results, were excluded from the sensitivity analysis for clarification.

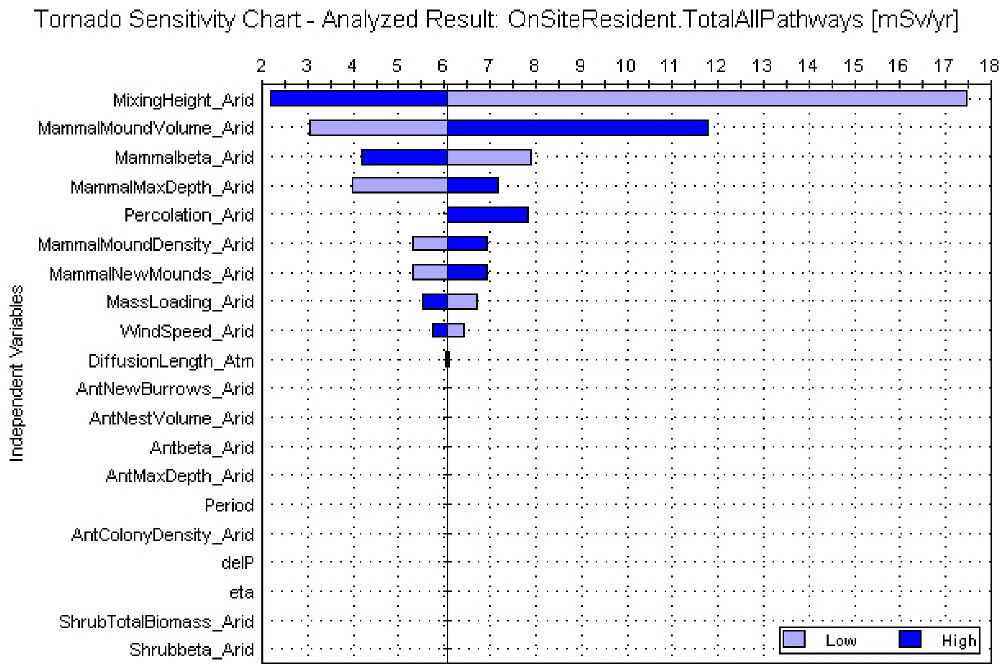

Figure 92. SDA Sensitivity Results for the Total Effective Dose Equivalent (TEDE) for Selected Variables representing Transport Pathways. The abscissa in the chart represents the TEDE for different values of the independent variables.

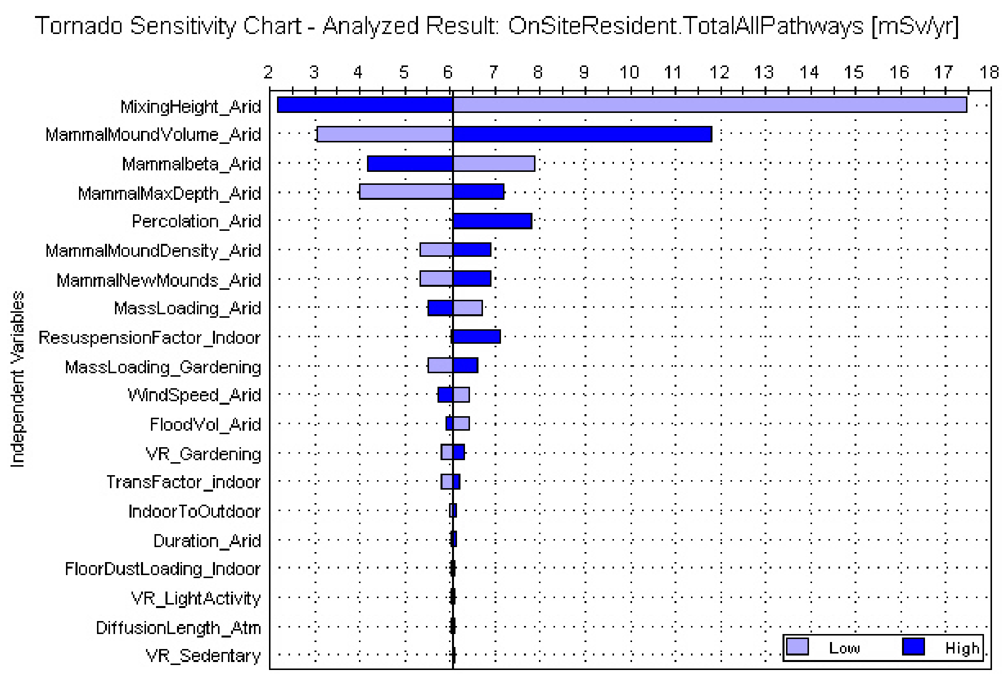

Figure 93. SDA Sensitivity Results for the Total Effective Dose Equivalent (TEDE) for Selected Independent Variables. The abscissa in the chart represents the TEDE for different values of the independent variables.

Although large uncertainties in dose estimates were not surprising, the parameters studied that had large impacts on the dose results are surprising. It might be expected that the movement of water through the burial site (i.e., percolation flow) would have one of the largest impacts on SDA dose results. However, the mixing height for the atmospheric pathway had the largest influence (for those evaluated) followed by parameters describing contaminant transport via burrowing mammals. These transport parameters and the models used to implement the pathways (as described in Chapter V and Chapter VI) should be reexamined if more representative dose and risk results would better support the remedial decision for the SDA. However, as indicated earlier in this chapter, more accurate estimates would unlikely alter significantly the risk-based information supplied from the screening tool for decision-making purposes.

SDA: Trade-offs in Exposure and Accident Risks

Figure 80 and Figure 81 illustrate the expected standard industrial injury and fatality risk estimates for remedial workers cleaning up the SDA in the manner preferred by many stakeholders in the State of Idaho (i.e., maximum retrieval). Legal decisions have been made that may force such aggressive remedial actions on the SDA regardless of the resulting risks posed to the general public and workers. In March 2008, the Ninth Circuit Court of Appeals affirmed an earlier decision requiring the DOE to remove all transuranic waste from the Idaho Site (Christensen 2008).

The purpose here is not to critique the maximum retrieval or other possible remedial actions for the SDA but instead to identify the risks to the general public and workers so that the decision made is, in fact, risk-informed. The SDA risks first described in Chapter IV and Appendix A based on expert judgment are expanded in this chapter using the results of the screening risk tool. However, the reality of any remedial decision must be made without caveat.

Whether made explicitly or implicitly, when deciding on remedial actions for a site, a trade-off is made between the risks posed by baseline conditions and the risks posed by remedial actions. Baseline risks tend to be long-term, exposure-based risks to the general public; whereas, remedial action risks are likely to be short-term, exposure-based and standard industrial risks to workers. However, remedial actions may also place members of the general public at risk from accidents involving contaminant releases but more likely due to more mundane events including traffic accidents. However, judgment is needed to assess whether or not the worker risks for a remedial action are unreasonable relative to the gains anticipated from its implementation.

The risks posed by the SDA appear unacceptably high and remedial actions are needed to place the site in a protective state. The results of this research indicate that, despite the remedial approach, the SDA will only be a candidate for *restricted* release (which limits the remedial benefit). Although maximum or targeted retrieval actions may reduce future exposure risks significantly more than managing the wastes in-place, these actions would also not allow the site to be released for *unrestricted* use. Large uncertainties are associated with any proposed remedial action. Thus, it may be argued that the trade-off between the maximum retrieval of SDA wastes and managing the wastes in-place may produce no demonstrable benefit on a *risk basis*.

The predicted morbidity risks for the *DBRA-Expected* and *DBRA-Maximum* cases are used to demonstrate the risk trade-off analysis. For illustrative purposes, the limit of the remedial action benefits is represented by the baseline risks (i.e., the maximum risk that can be reduced) and the corresponding costs are represented by the worker risks. The comparison of these risks then represents a trade-off associated with applying remedial actions to a site.

The trade-off is demonstrated graphically by showing the baseline exposure risks to the general public and the standard industrial injury risks for the workers during remedial actions on a single graph as shown in Figure 94. The primary message to take from Figure 94 is that standard industrial risks (based on injury statistics) for retrieval actions tended to be significantly larger that the predicted impacts for exposure to radionuclides from the SDA. This trade-off diagram is controversial because the risk metrics portrayed are fundamentally different; however, this comparison does indicate the type of risk trade-off that would be made if wastes are retrieved for treatment and disposal elsewhere instead of being managed in-place.

SDA: Screening Quantitative Comparison of Remedial Alternatives

The proposed retrieval alternatives—either maximum or targeted—may not be worthwhile in terms of the risks traded off, especially considering the potential benefit of installing a surface barrier on the SDA earlier than later. The primary drivers for reducing contaminant migration in the manage-in-place alternative are reducing biotic intrusion, barometric pumping, and water percolation primarily via the installation of a surface barrier. The predicted lack of effectiveness of the *in situ* grouting (ISG) process, used in both the manage-in-place and retrieval alternatives, to immobilize contaminants and reduce risks is a function of the assumptions made (e.g., rupturing remaining containers, grouting efficiency of less than 100%, etc.) and may be more effective in reality.

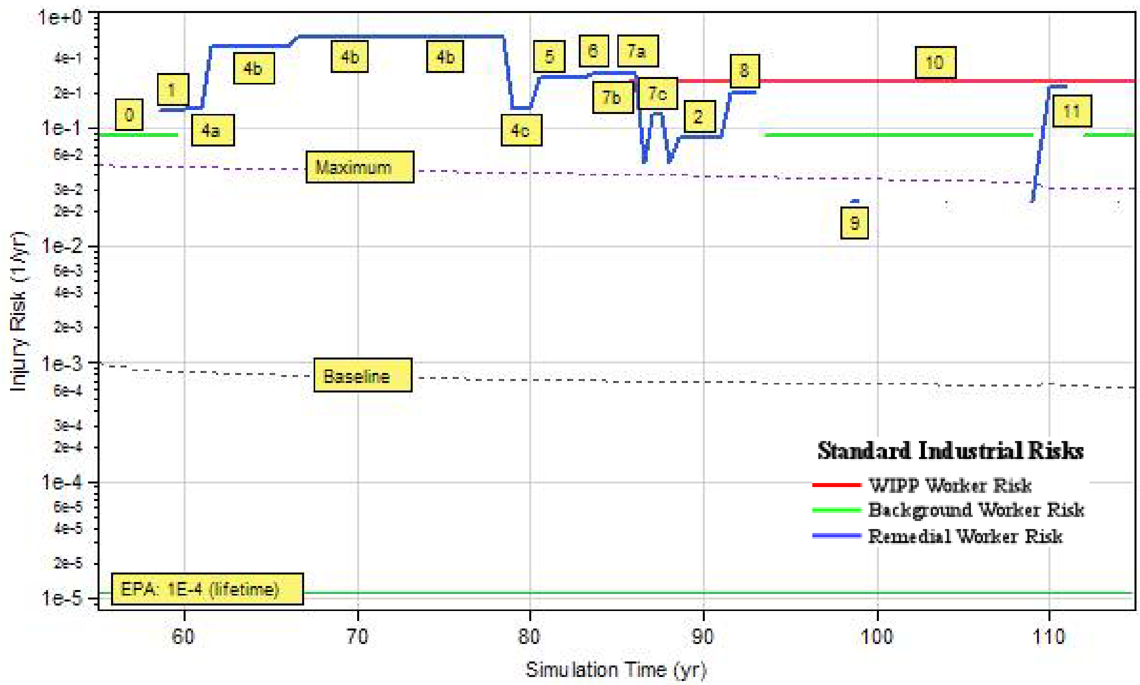

Figure 94. Solid Lines: *DBRA-Expected* Standard Industrial Injury Risks for the SDA Maximum Retrieval Alternative. Dotted Lines: Annual Morbidity Rates for All Pathways and Radionuclides. (Steps correspond to those in Table 51.)

The risk information summarized in this chapter represents just one input to the risk-informed decision-making process. There may be reasons other than risk that dictate remedial actions for a contaminated site. However, it is prudent to have an idea of the risks posed by potential exposures of workers and the public to contaminants as well as the worker and public risks likely for the remedial actions needed to disposition the wastes. Previous research used expert judgment to classify the risks posed by proposed remedial actions for the SDA as described in Chapter IV (Brown et al. 2005).

The results from applying the screening risk tool to SDA remedial alternatives are be used to examine the original classifications in Table 15 from Chapter IV. The information in Figure 94 and similar diagrams provide the predicted injury and fatality risks and probabilities for accidents during proposed remedial actions. The updated results are placed in Table 52 (in the shaded columns). The original classifications were based on the definitions in Exhibit 1 and Exhibit 3 from Chapter III to assess event probabilities and severities. The definitions were modified as described in Exhibit 4 (in Chapter III) for quantitative risk estimates.

When applying the classifications in Table 8 from Chapter III to the *quantitative* risk results in Table 52, only the step including off-site disposal at WIPP is classified as *severe*; however, all other steps are classified as *critical*. All steps but *in situ* grouting (ISG) for subsidence control and the long-term stewardship (LTS) activities pose injury risks that would be considered *probable*; whereas, the ISG and LTS activities pose injury risks that are deemed *possible*. The injury and fatality risks associated with the off-site shipment of transuranic wastes to the WIPP are deemed probable because of the very large number of shipments that would be required and the long duration of the activity.

When compared to the original, qualitative results, both ISG steps and LTS activities appear to have a lower contribution to remedial alternative risk than estimated in Table 15 from Chapter IV based on expert opinion. On the other hand, installation of the surface barrier and off-site transport and disposal of wastes at WIPP appear to have higher contributions to overall risk. In fact, none of the process steps are classified as *low-risk* based on the quantitative results. Some of these changes are due to focusing on different potential receptors (i.e., for LTS activities); however, the changes noted for the surface barrier and off-site disposal stem from a better idea of the potential risks involved with performing these steps.

Table 52. Summary of the Most Important Human Health Risks and Knowledge Gaps for the SDA Remedial Alternatives

Process Step	1A. Baseline/No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the conse-quences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C) Updated
1. Character-ization	√	√	√	√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Injury • Fatality	• 8E-1 • 5E-3	• 1E-1 • 5E-4	• Potential facilitated plutonium transport • Presence/location of high-rad material • Geospatial waste distribution	Significant (0,3,2,2) Significant
2. ISG for Stab.		√		√	√	• Failure of high-pressure grout system	• Probable	• Severe	• Worker	• Injury • Fatality	• 1E-1 • 8E-4	• 8E-2 • 5E-4	• No *high-priority* gaps	High (1,4,13,1) Significant
3. ISG for Both			√	√	√	• Failure of high-pressure grout system	• Probable	• Severe	• Worker	• Injury • Fatality	• 6E-1 • 6E-3	• 1E-1 • 9E-4	• Geospatial waste distribution	High (2,4,22,1) Significant
4. Excavate, Retrieve, Segregate				√	√	• Contaminated soil removal and exposure • Tote-bin dropped outside confinement • Traumatic injury during excavation	• Probable • Probable • Possible	• Critical • Critical • Severe	• Worker • Worker • Worker	• Injury • Fatality • Maximum Exposure	• 1E+0 • 2E-2 • (d)	• 5E-1 • 8E-4 • 5E+2	• Future legal decisions • Geospatial waste distribution	Significant[e] (0,11,37,3) Significant

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Appendix A.

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Appendix A.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The probability of this maximum radiation exposure event cannot be determined from available information.

e. The fact that there are three *high-risk* hazards associated with this process step makes it highly significant from a risk perspective so much so that is may need to be considered a *high-risk* step even though there are no events that would be considered both *probable* (in terms of likelihood) and *severe* (in terms of consequences).

Table 52, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[b]	How likely is it?	What are the consequences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[c]	Overall Contribution to Risk[d] (H,S,L,N/C)
5. *Ex Situ* Treatment				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Injury • Fatality	• 1E+0 • 2E-3	• 3E-1 • 1E-4	• No *high-priority* gaps	Significant (0,5,13,1) Significant
6. Packaging				√	√	• Containment system failure and exposure	• Possible	• Severe	• Worker	• Injury • Fatality	• 1E+0 • 5E-3	• 3E-1 • 4E-4	• No *high-priority* gaps	Significant (0,2,17,0) Significant
7. Storage and Disposal				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Injury • Fatality	• 1E+0 • 8E-3	• 1E-1 • 2E-4	• No *high-priority* gaps	Low (0,0,5,0) Significant

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Appendix A.

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Appendix A.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The fact that there are three *high-risk* hazards associated with this process step makes it highly significant from a risk perspective so much so that is may need to be considered a *high-risk* step even though there are no events that would be considered both *probable* and *severe* (in terms of consequences).

Table 52, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[b]	How likely is it?	What are the conse-quences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[c]	Overall Contribution to Risk[d] (H,S,L,N/C)
8. Barrier Installation		√	√	√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Injury • Fatality	• 1E+0 • 2E-2	• 3E-1 • 5E-4	• No *high-priority* gaps	Low (0,0,8,3) Significant
9. LTS	√	√	√	√	√	• LTS Failure (MiP) • LTS Failure (RTD)	• Probable • Probable	• Severe • Critical	• Public • Public	• Injury • Fatality	• 2E-1 • 8E-4	• 2E-2 • 8E-5	• Geospatial waste distribution	High (1,7,8,1) Significant
10. Off-Site Disposal				√	√	• Injuries of operation of heavy equipment	• Probable	• Critical	• Worker	• Injury • Fatality	• (d) • (d)	• 3E-1 • 4E-2	• No *high-priority* gaps	Significant (0,3,11,1) High
11. Barrier Repair[e]		√	√	√	√	Same as LTS	Same as LTS	Same as LTS	Same as LTS	• Injury • Fatality	• 1E+0 • 3E-2	• 3E-1 • 6E-4	• Not determined	Not determined Significant

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Appendix A.

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Appendix A.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The probabilities of injury or fatality related to the shipment off-site of transuranic wastes is likely to approach unity because of the very large number of shipments likely to be required.

e. This step was added so that the monitoring and maintenance activities in Step 9 can be distinguished from the higher risks likely during major barrier repair activities.

From the information in Table 52, significant risks were associated with the remedial actions employed in either the manage-in-place or retrieval alternatives. However, these alternatives can be compared and ranked using the information developed in this research. For example, Figure 73, Figure 76, and Figure 80 illustrate that retrieval alternatives may be more effective at reducing exposure risks than the manage-in-place options; however, neither alternative appeared capable of cleaning up the site for unrestricted use. Furthermore, the reduction in exposure risks was made at the expense of increased worker risks. As illustrated in Figure 94, the trade-off was between exposure risks (and perhaps peace of mind) to future receptors and worker risks (and perhaps those to the general public). In terms of qualitative and screening quantitative risk estimates, there does not appear to an obvious remedial alternative that was preferable in terms of reducing both exposure risks to the general public and worker risks.

Employing the rationale from Chapter IV and considering both exposure and worker risks in Table 52, one rank-ordering of the remedial alternatives in terms of highest to lowest risk produces the following:[186]

No Action >> Maximum RTD > Targeted RTD >>
MIP (ISG for Immobilization and Subsidence Control) >
MIP (ISG for Subsidence Control) > Surface Barrier

The above rank-ordering, which essentially is the same as that from the qualitative analysis, was based on model predictions and value judgments as described in Chapter V

[186] The rank-ordering is based on the following assumptions: 1) risk increases with increased excavation and retrieval, 2) employing *in situ* grouting (ISG) for both subsurface stabilization and immobilization is higher risk than when ISG is used for only subsidence control (as illustrated in Table 52), 3) not containing the wastes using a surface barrier would have the potential to impact by far the greatest number the public, which would overwhelm any reduced worker risks, and 4) relative risk reductions that produce the same site release category (e.g., restricted, unrestricted, etc.) are essentially equivalent in terms of risk due to the large uncertainties involved (as illustrated in this chapter).

and Chapter VI. A more detailed and representative assessment of risks, investigation of assumptions (e.g., facilitated transport), different set of classifications (demonstrating different value judgments), or examination of remedial requirements (e.g., mandated retrieval of buried Rocky Flats Plant transuranic wastes) might produce a different rank-ordering or set of remedial alternatives than that evaluated in this research.

Thus a risk-informed remedial decision may include non-risk factors (e.g., social values, past legal agreements, etc.) that are deemed important by the decision-maker. For example, if waste retrieval is selected based on information other than risks, it is hoped that the information developed in this research can drive the retrieval process to the minimum retrieval volume possible based on risks, timing, and receptors. Furthermore, the results from this research can be used to identify those process steps most likely to be dangerous to remedial workers so steps can be taken during the planning stage.

SDA: Hypothesis Testing

Primary Research Hypothesis: For the Idaho Site Subsurface Disposal Area (SDA), the remedial alternative involving managing buried wastes using *in situ* techniques, barriers, etc. will result in lower life-cycle risks to potentially impacted receptors than the alternative whereby buried wastes are retrieved and treated for disposal off-site.

The results from both the previous qualitative analysis described in Chapter IV and the screening quantitative results in this chapter suggested that the life-cycle risks associated with the manage-in-place (MIP) alternative were significantly lower than those for the retrieval alternatives considered in this research. Although retrieval activities may have reduced exposure risks to the general public more than the MIP alternative, the risk reduction came at the price of increased risks to both workers and the general public. For example, it can be shown that the predicted exposure risk due to emissions from the trucks needed to transport the SDA wastes to WIPP was larger than the expected risk reduction for the site from RFP TRU waste retrieval activities. However, the truck emission risks paled in comparison to those from the more mundane traffic accidents expected to occur during the shipment of RFP TRU wastes to WIPP.

For the SDA, the life-cycle risks over the 1,000-year assessment period to both the general public from potential contaminant exposure and workers from both exposures and standard industrial risks were likely to be significantly lower for the MIP alternative than for those involving waste retrieval. The trade-off for selecting retrieval of wastes from the SDA was one of trading increased worker and general public risks over the short-term for an apparent (but highly uncertain) reduction in long-term exposure risks and the peace of mind from transferring wastes and their risks to a more stable disposal state. The primary hypothesis for the SDA would thus be accepted based on these results.

Research Hypothesis: The remedial alternative that results in the lowest life-cycle risks to potentially impacted receptors is a combination of *in situ* techniques and targeted retrieval actions taken, if possible, in different areas of the disposal site.

For the conditions and assumptions made in modeling the SDA, this hypothesis would be rejected because the lowest life-cycle risk remedial alternative was managing the wastes in-place by installing a surface barrier installation *over the entire SDA*. The impact of *in situ* grouting (ISG) for contaminant immobilization was small likely due to the amount of contamination already released into the vadose zone beneath the SDA.

Retrieval of wastes (including that targeted to the *highest-risk* wastes) did produce lower expected long-term exposure risks to the general public but at the expense of greatly increased worker and general public risks. Thus the life-cycle risks for retrieval activities could not be considered lower than those for the MIP alternative even on a local basis.

Although the targeted retrieval alternative would not likely result in the lowest life-cycle risks for the SDA, it might still be the selected alternative because non-risk factors may be given precedence in the remedial decision. If this is the case, then it would be desirable to use the results of this study to guide the implementation of the remedial decision. For example, because of the magnitude and types of risks involved with waste retrieval activities, it would be prudent to limit retrieval activities to the smallest possible volume of the *highest-risk* wastes considering effects, transport, timing, and potential receptors. The screening risk tool developed in this research could be put to advantage in deciding where to retrieve wastes.

Research Hypothesis: The significant sources of exposure and accident risks for both general public and workers (in addition to non-risk factors such as costs, technical feasibility, cultural and societal impacts, etc.) must be considered for each remedial alternative for the decision to be risk-informed.

For the conditions studied and the assumptions made in modeling the SDA, this hypothesis would be accepted. For example, if long-term exposure risks based on expected results were the only input to the remedial decision, then the retrieval alternatives (and specifically the targeted retrieval option) would be preferred to those involving managing the wastes in-place. However, if worker and general public risks were factored into the decision, then the manage-in-place remedial alternative would be the selected remedial alternative.

Perhaps the greatest contribution of the screening risk tool is elucidating the impact of uncertainty on the remedial decision. Retrieval actions appeared to pose lower expected long-term exposure risks to future generations than the MIP options. However, when uncertainties in predicted exposure risks were examined, neither alternative clearly provided lower long-term exposure risks nor produced a final state that could be released for unrestricted use. When factoring these considerations into the decision, the manage-in-place alternative appeared preferable from a life-cycle risk perspective. However, risks and uncertainties are just one set of inputs to the risk-informed decision-making process. The remedial alternative may be selected for reasons other than risk. The results of this research can be used to guide the implementation of the selected remedial alternative.

Screening Risk Analysis of the Oak Ridge Bear Creek Burial Grounds (BCBG)

The second DOE site to which the screening tool was applied is the Bear Creek Burial Grounds (BCBG) on the Oak Ridge Reservation. This site was selected for evaluation based on the fact that the BCBG and SDA tend to bracket the types of site conditions, contaminants, wastes, and hazards anticipated for DOE buried waste sites. For example, the SDA is in an arid region with a very deep vadose zone and the wastes buried there took many forms (e.g., resins, glass, metals, fuel-like elements, etc.)—often intermixed—contaminated with many different types of contaminants (e.g., volatile organic compounds, transuranic elements, fission products, etc.). The BCBG, on the other hand, is in a much more humid region with a very shallow saturated zone (i.e., some areas are perennially inundated with groundwater).

The wastes buried in the BCBG were primarily uranium and wastes contaminated with uranium from the Y-12 Plant. Unlike the SDA wastes, many BCBG areas contain unstable, explosive, or pyrophoric materials (e.g., uranium fines, cutting, chips, etc.) that present unique hazards to workers. Long-term exposure effects from waste buried at the SDA tend to be manifested over long times and via the groundwater pathway. Because of the shallow groundwater and abundant surface water in and near the BCBG waste areas, impacts tend to be more rapid and via the surface water pathway. CERCLA remedial investigation reports for the Bear Creek Valley (where the BCBG are located) are available for information and comparison purposes (SAIC 1996a; b; c; d; e; f).

BCBG: Screening Quantitative Baseline Risk Assessment

The initial step for evaluating a buried waste site is to determine whether or not remedial action is required for the site. A screening baseline risk assessment was performed using current site conditions to determine if risks posed from the BCBG contaminants are greater than appropriate concentration, risk, or other limits. The limits used in the BCBG baseline risk assessment are the same as those used for the SDA study.

For the baseline risk assessment *in this research*, the "reasonable" maximal exposure corresponded to that for the *on-site resident* as for the SDA. Although residents do not live on DOE sites, this scenario typically provided the largest risk predictions as shown in Figure 95 for predicted dose results and simplified the analysis for the purpose of illustration. This selection also allowed direct comparisons be made to the results obtained for the SDA. However, when applying the framework and screening risk tool to a site, the usual-and-customary receptor scenario should be adopted. A residential receptor more proximate to the BCBG may be warranted for more detailed analyses.

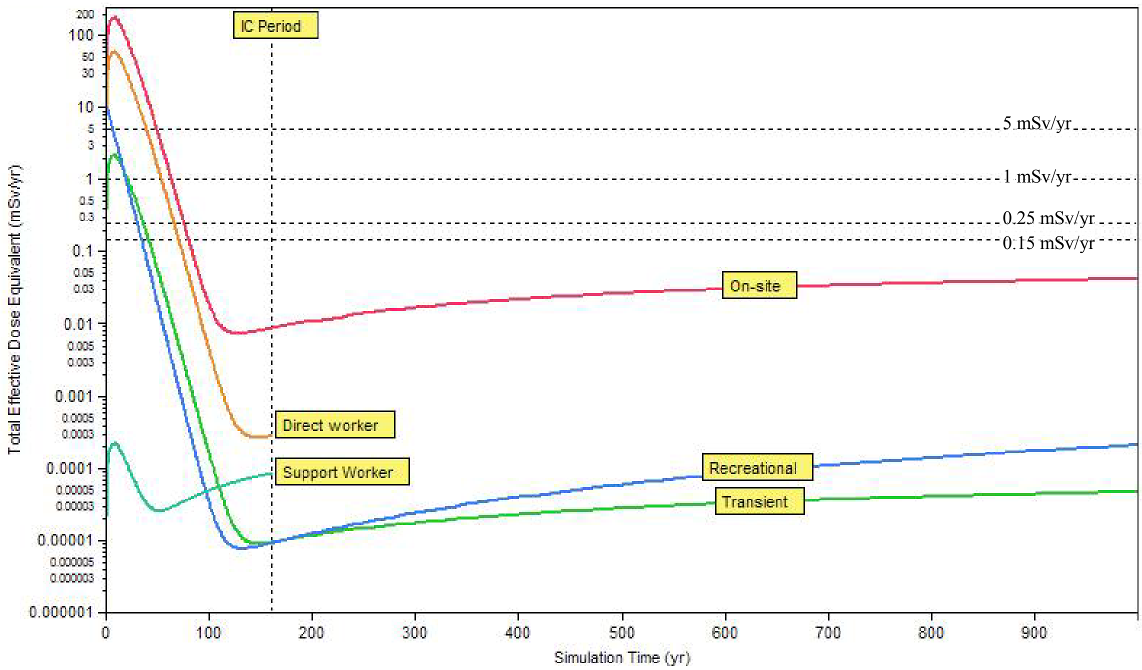

Figure 95. BCBG General Public and Worker Scenarios from Chapter VI: Baseline Annual Total Effective Dose Equivalent (TEDE) for All Pathways Summed over all Radionuclides and Compared to Various Dose Limits

One advantage of using the screening risk tool is the manner in which it can lend consistency and transparency to the site disposition analysis. For example, the screening risk tool can be used to determine whether a buried waste site poses excessive risk and, if so, which contaminants may be of potential concern from a risk perspective using either a point-value or probabilistic analysis. Proposed remedial actions can then be assessed for their potential effectiveness in reducing site risks as well as the concomitant exposure and accident risks to workers and the general public. These evaluations are performed using a consistent set of models, assumptions, exposure scenarios, etc. that can be updated if more accurate results are required to make a remedial decision.

A number of assumptions were made concerning the potential source release and transport pathways that also impacted the degree of exposure to BCBG receptors. The major source and transport pathway assumptions that impacted the results from the screening risk tool for the BCBG included:

- All wastes were buried at a single time instead of distributing burials over time.
- The contaminant source releases were controlled by the surface wash, dissolution, and diffusion mechanisms as modeled in Appendix E. Whereas, the release mechanisms had large impacts on the exposure predictions for the SDA, few of the BCBG contaminants appeared to be bound in waste matrices or in containers (as described in Appendix D). Therefore, the source release models had little impact on the risks predicted from the BCBG wastes.
- The complex interactions of contaminants in the subsurface could be described using a simple linear partitioning (i.e., K_d-based) retardation model. Because few BCBG wastes appeared to have been either bound in waste matrices or buried in containers, the primary source release and transport limitations were due to retardation and solubility.
- The maximum concentrations of contaminants in the aqueous phases were independent and could be described using one solubility value for each.
- The position of the drinking water well intake in relation to the direction of flow did not substantially change the concentration in the drinking water.
- The atmospheric and soil pathways including the vadose zone could be approximated using simple "box" models using GoldSim *Cell Pathway* elements.
- The surface water pathways could be described using a series of linked *Cell Pathway* elements.
- The primary transport pathway for contaminants to impact off-site receptors (at least, until the Institutional Control period has expired) was via the surface water pathway. There was neither loss due to air-water exchange of volatile contaminants nor deposition as contaminants are transported through the surface water.

The additional assumptions made in developing the screening risk tool that significantly impact risk predictions are described in the tool where they were made.

The assumptions have major impacts on predicted exposures and should be evaluated when considering whether a site poses an unacceptable risk. Because a baseline risk assessment (BRA) is used to determine if a site might pose an unacceptable risk, values are selected to represent the assumptions made in such a way as to maximize predicted exposure. The impacts of the assumptions made are explored in this chapter.

As illustrated for SDA, different calculations or simulations can be performed to determine whether or not site contaminants are likely to pose an unacceptable risk to potentially impacted receptors. The "deterministic" or point-value baseline risk assessment runs for the BCBG are described in Table 53[187]. If the BCBG poses unacceptable risks, then the results from the original analysis or more detailed analyses (e.g., stochastic simulations using the screening risk tool) can be used to generate the list of contaminants of potential concern (COPC) for potential remedial action.

Table 53. BCBG "Deterministic" Baseline Risk Assessment (DBRA) Simulations

Designation	Description[a]
DBRA-Expected	Baseline conditions with best inventory and stochastic elements set to expected (i.e., 50th-percentile) values. Inventories are segregated by waste form and containers (if applicable). All source release and transport mechanisms are enabled *except for organic degradation*.
DBRA-Maximum	Baseline conditions with inventory and stochastic elements set to their respective 95th-percentile upper or lower values depending on the estimated risk impact. Inventories are segregated by waste form and in containers (if applicable). All transport mechanisms are in effect *except for organic degradation*.

a. Maximum resuspension as described in Tauxe (2004) was not used for any simulation in this research. The flooding pathway and colloidal transport do not apply to the BCBG as described in Chapter VI. The *DBRA-ExpLoose* case evaluated for the SDA is not considered because few of the contaminants were either bound in matrices or buried in containers. The results of the SDA analysis indicated that, even for a site where wastes were either bound or contained, the impact is much smaller than the other effects represented in this table. The *DBRA-WorstCase* was not considered because the results are unrealistic and the impact of the assumptions involved was already described for the SDA and in Appendix G.

[187] Only the *DBRA-Expected* and *DBRA-Maximum* runs were evaluated for the BCBG based on earlier results. When compared to Table 49 for the SDA, the simulation for all loose wastes provided little useful information because most BCBG wastes were considered loose. The "worst case" run was not needed because the *DBRA-Maximum* case was previously shown to produce excessively high exposure and risk predictions.

The *DBRA-Expected* case for the BCBG provided predicted dose, risk, and hazards results for the best inventory and expected (i.e., 50th-percentile) transport parameters employing the waste forms, release mechanisms, and transport pathways as described in Table 53. The total latent cancer incidence (morbidity) rate summed over all pathways and radionuclides[188] is presented in Figure 96 and, as illustrated in this figure, the expected morbidity rate exceeded the EPA *de minimus* limit (i.e., 10^{-6} cancer risk) for the entire assessment period. However, the expected morbidity rate fell below the EPA "action limit" (i.e., 10^{-4} cancer risk) in less than 100 years when the present time is at Year 60. Based on morbidity predictions, the *radioactive* contaminants buried at the BCBG may pose unacceptable risks for only a few more decades.

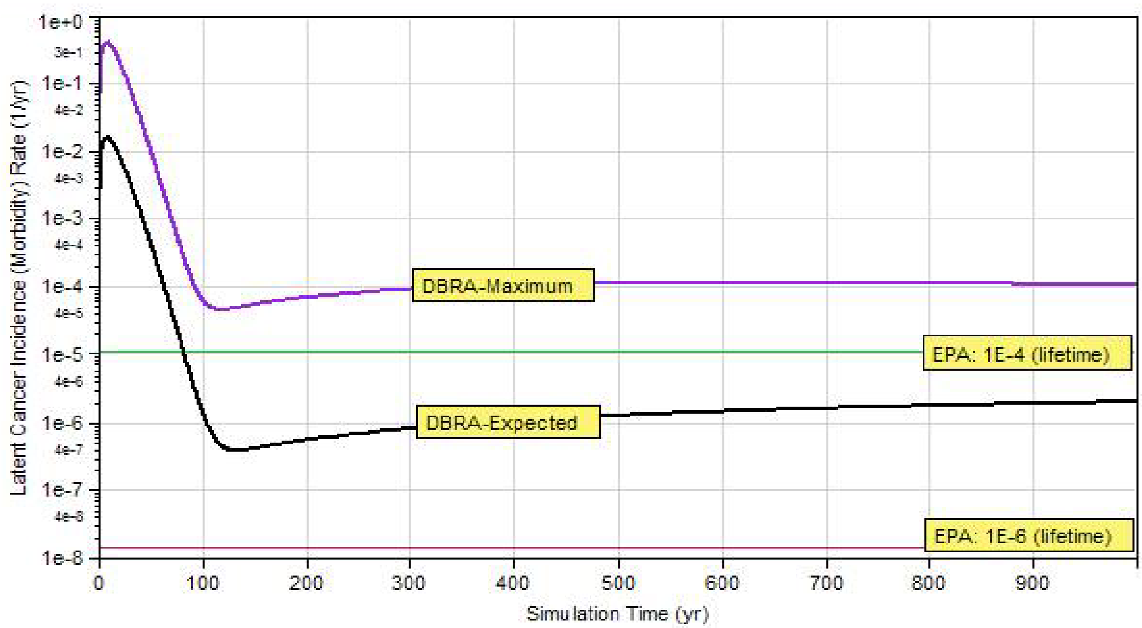

Figure 96. BCBG *DBRA-Expected* and *DBRA-Maximum* On-Site Resident Scenarios: Annual Latent Cancer Incidence (Morbidity) Rate for All Pathways and Radionuclides (compared to 10^{-4} and 10^{-6} limits converted to annual bases).

[188] The reasons for using the cancer incidence were described earlier in this chapter for the SDA results.

The *DBRA-Maximum* case in Figure 96 shows the impacts on the predicted annual morbidity rate when bounding (e.g., 95^{th}-percentile) inventories and transport parameters were used. The predicted morbidity results did not fall below the EPA "action limit" (i.e., 10^{-4} cancer risk) at any time during the assessment period. Similar results were found for predictions of chemical cancer incidence rates and non-cancer effects for the BCBG wastes as illustrated in Figure 97 and Figure 98, respectively. Therefore, as suggested in the BCBG remedial investigation (SAIC 1996a; e), there is little doubt that this buried waste site will require remedial action. Like the SDA, risks from both chemicals and radionuclides were predicted to be unacceptable; however, the radionuclide risks for the BCBG wastes may become acceptable (i.e., fall below the EPA "action limit" of 10^{-4} cancer risk) in a few decades.

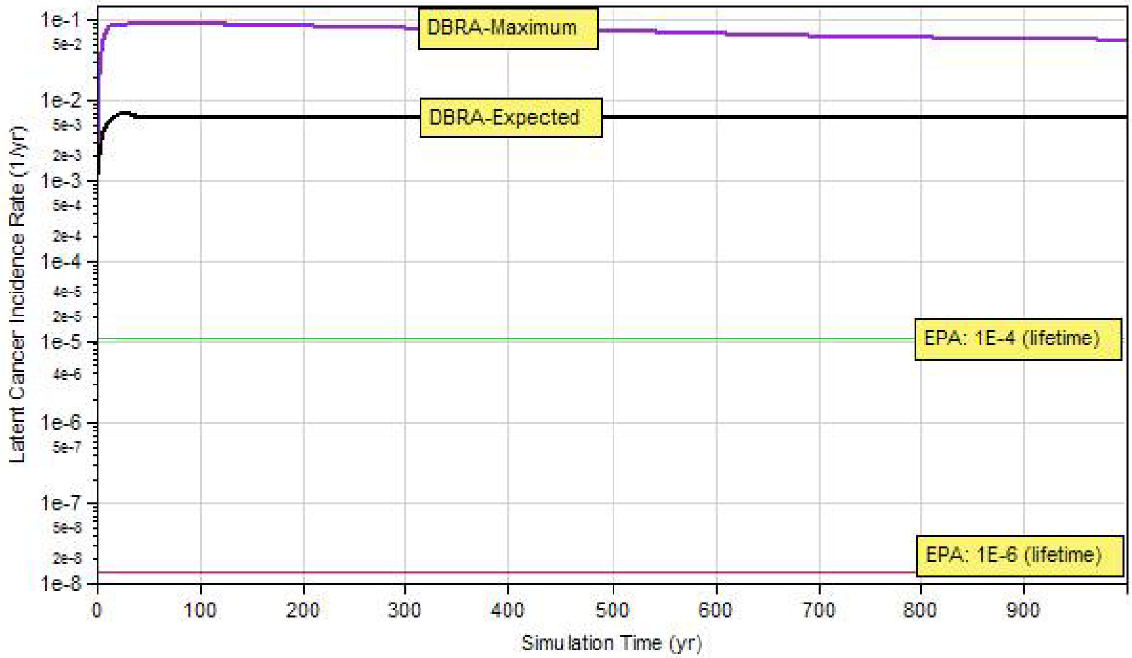

Figure 97. BCBG *DBRA-Expected* and *DBRA-Maximum* On-Site Resident Scenarios: Annual Latent Cancer Incidence Rate for All Pathways and Chemicals (compared to EPA cancer risk limits converted to annual bases).

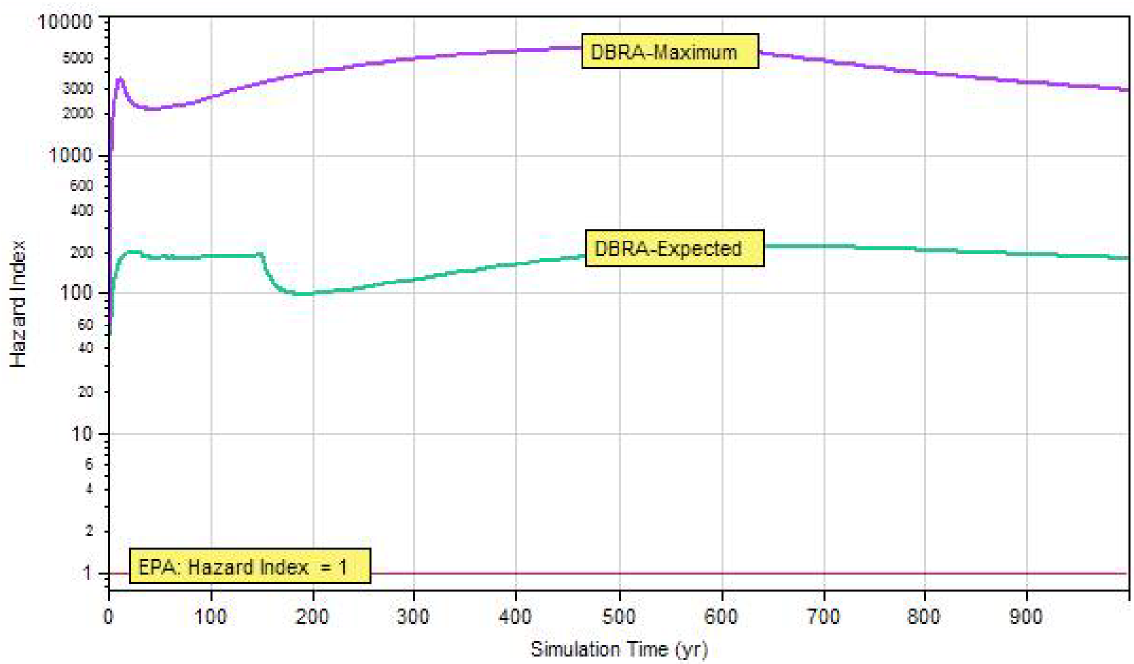

Figure 98. BCBG *DBRA-Expected* and *DBRA-Maximum* On-Site Resident Scenarios: Hazard Index for All Pathways and Chemicals (compared to EPA limit of 1).

BCBG: Probabilistic Assessment of Baseline Risks

A probabilistic approach using the results from a Monte Carlo (MC) simulation was defined for identifying the contaminants of potential concern (COPCs) for the SDA. MC results describe the uncertainties in the dose, risk, and hazard predictions. The complementary cumulative distribution function (CCDF) or exceedance curve illustrates the likelihood of exceeding a given dose, risk, or hazard. An example of the CCDF for the annual morbidity rates for BCBG radionuclides is provided in Figure 99. Depending on the limit used and the aversion to risk, the COPCs may include only Co-60 (or none accounting for rapid decay) or as many as ten radionuclides as shown in Figure 99.

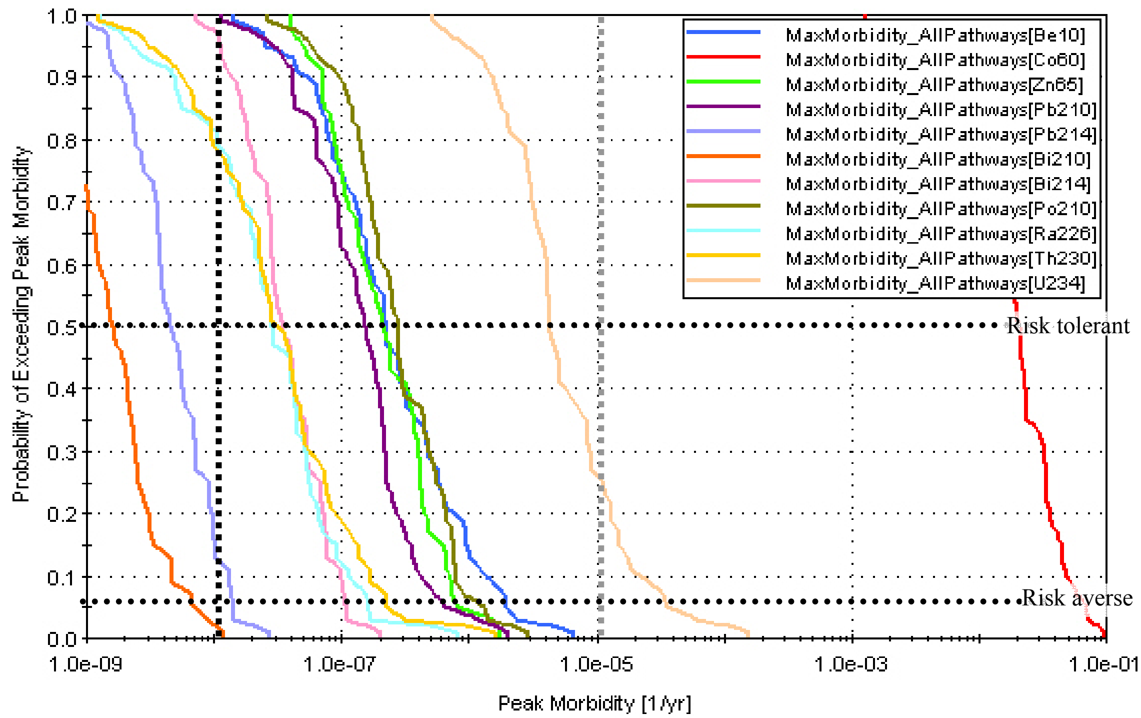

Figure 99. BCBG Complementary Cumulative Distribution Function (CCDF) for the Peak Morbidity Predictions for the On-Site Resident Scenario. (The gray and black dotted vertical lines represent the EPA limits corresponding to 10^{-4} and 10^{-6} cancer risk, respectively. The black, dotted horizontal lines indicate whether one is risk averse at $p = 0.05$ or risk tolerant at $p = 0.5$.)

The exceedance curve corresponding to the BCBG peak chemical cancer risks is provided in Figure 100. For this specific case, the limit selected to define acceptability had a much greater impact on the COPC list than the tolerance of risk. The predicted cancer risks for six of the seven chemicals identified in Figure 100 were likely to exceed the EPA *de minimus* limit. The risk for chloroform was less likely to exceed the EPA *de minimus* limit than the others; however, the probability is high enough that chloroform would likely be included in the COPC list for the BCBG. If the EPA "action limit" was instead used for the BCBG chemical cancer risks, the chloroform as well as Cr-53 and Cd-110 would be excluded from the BCBG COPC list.

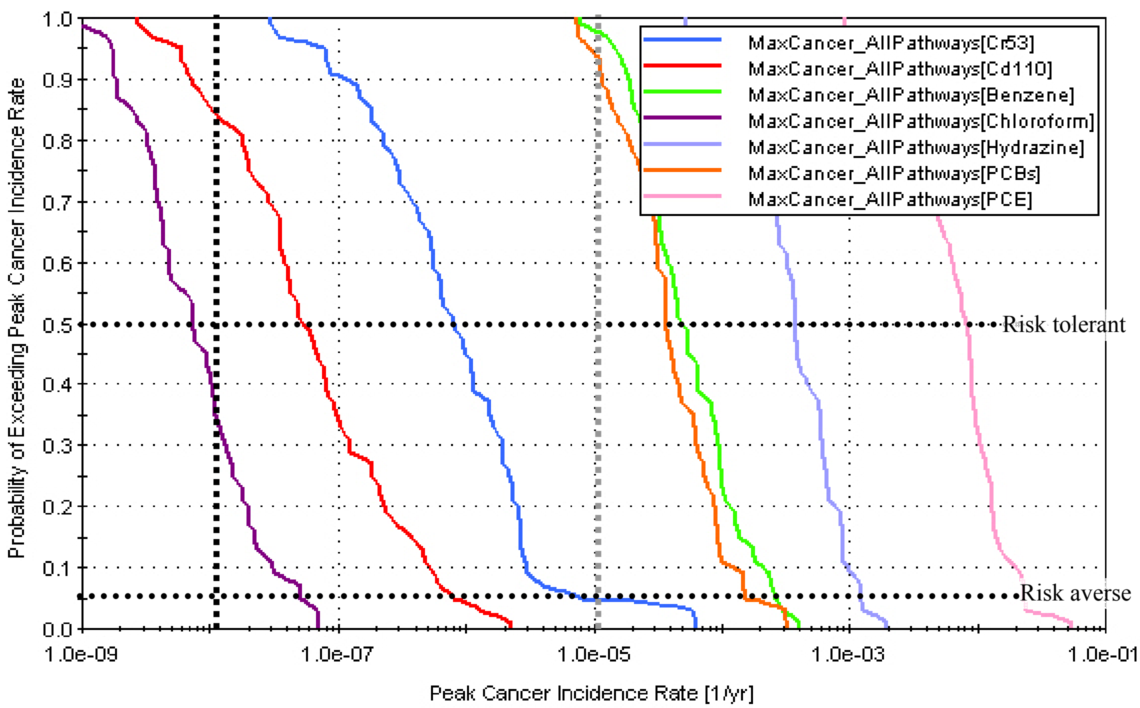

Figure 100. BCBG Complementary Cumulative Distribution Function (CCDF) for the Peak Chemical Cancer Predictions for the On-Site Resident Scenario. (The gray and black dotted vertical lines represent the EPA limits corresponding to 10^{-4} and 10^{-6} cancer risk, respectively. The black, dotted horizontal lines indicate whether one is risk averse at p = 0.05 or risk tolerant at p = 0.5.)

However, neither the level of risk tolerance nor the limit selected to define acceptability had a significant impact on the identities of the COPCs for non-carcinogen effects as illustrated in Figure 101. Nine of the ten contaminants would be included as COPCs despite the risk tolerance level or limit selected. Only Cr-53 would be excluded from the COPC list if one was tolerant of the risks. The identities in the COPC list often changed dramatically depending on the limit used. For example, a limit on the hazard quotient of 0.1 is often used because of the large uncertainties in the corresponding predictions and may result in many additional COPCs depending upon site conditions. For the BCBG, this was not the case.

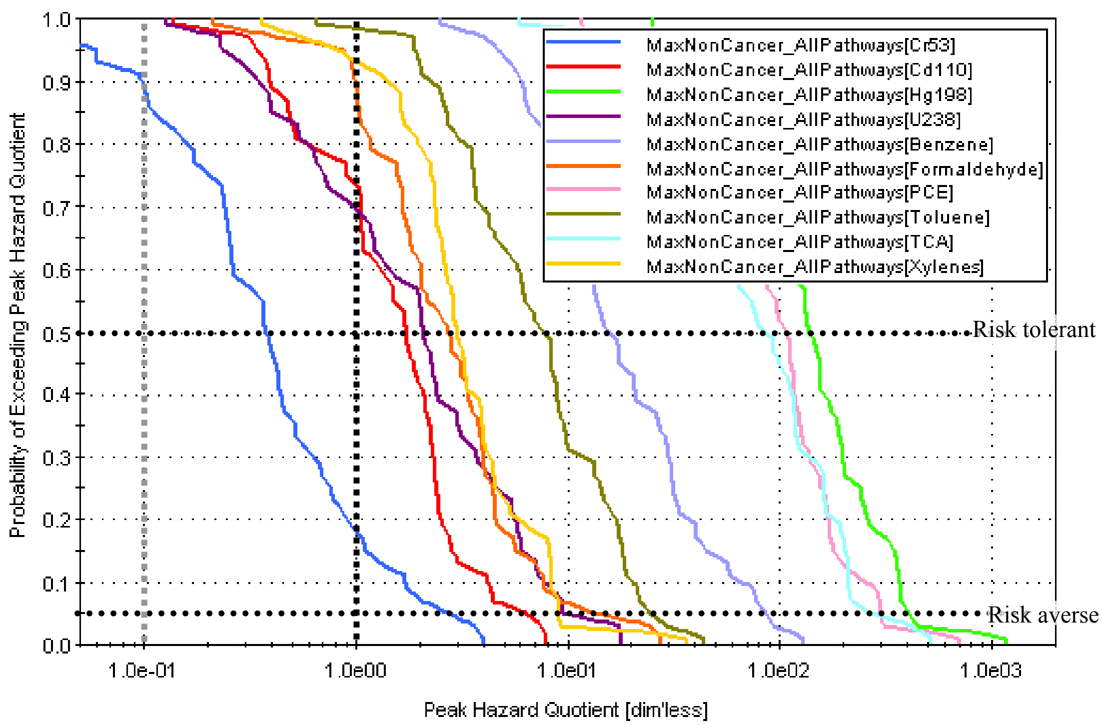

Figure 101. BCBG Complementary Cumulative Distribution Function (CCDF) for the Peak Hazard Quotients for the On-Site Resident Scenario. (The gray and black dotted vertical lines represent limits of 0.1 and 1, respectively. The black, dotted horizontal lines indicate whether one is risk averse at $p = 0.05$ or tolerant at $p = 0.5$.)

The proper use of probabilistic information to identify contaminants of potential concern (COPC) can be much more transparent and consistent than the usual and customary manner using point-value evaluations to identify COPCs. The assumptions (e.g., risk aversion, limits, etc.) for which COPCs were identified are clearly identified during the selection process. The information used to identify the COPCs agreed much more closely with the risk-triplet concept in that the risk curve and not a single value is needed to adequately describe the risk (Kaplan and Garrick 1981).

BCBG: Screening Quantitative Remedial Alternative Risk Evaluation

Because of the unacceptable risks posed by site conditions as illustrated in Figure 96 through Figure 98, the BCBG will likely require remedial action. Possible actions were assumed to involve either managing the wastes in-place or retrieving wastes for treatment and disposal. Other remedial actions were considered outside the scope of this research and the screening risk tool as currently developed.

Figure 102 illustrates the predicted impact on the radionuclide morbidity rate for the manage-in-place and retrieval remedial actions considered in this research. These results indicated that there is little difference in the alternatives based on predicted morbidity risks and that it would be possible to place the BCBG in a state that would be protective (for radionuclides) for restricted release. In fact, natural decay of the radionuclides would produce a protective state (for radionuclides) in a few decades without remedial action. However, a very different story emerged when the predicted cancer and non-cancer risks (in Figure 103 and Figure 104, respectively) for the hazardous chemicals buried in the BCBG were evaluated for both baseline conditions and after proposed remedial actions would be completed.

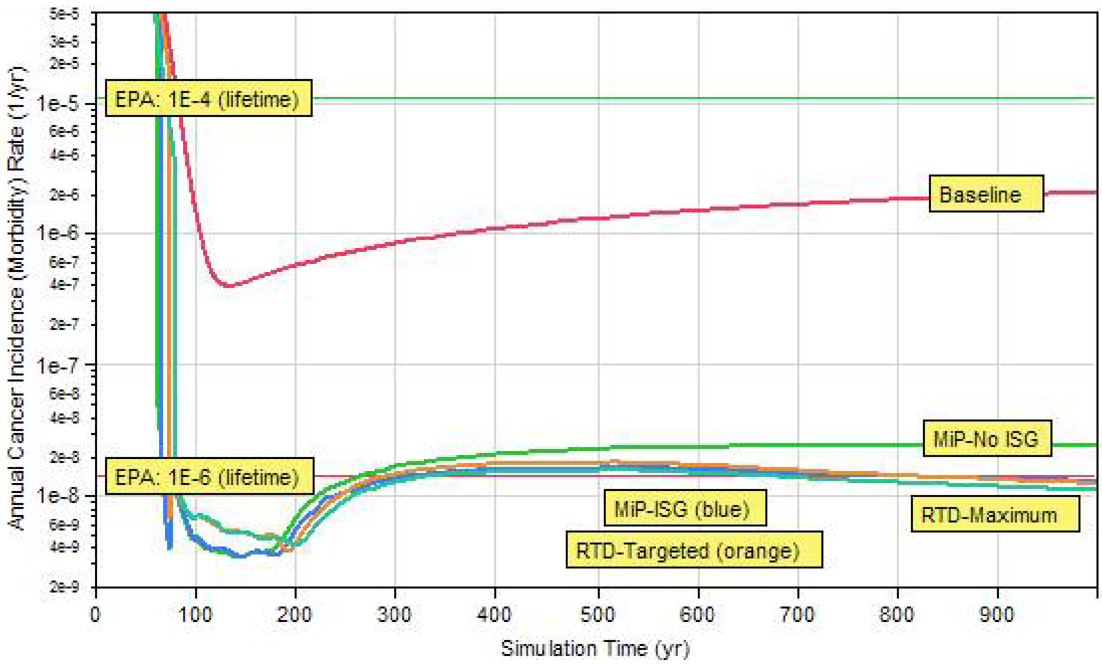

Figure 102. BCBG *DBRA-Expected* Case for On-Site Resident: Annual Radionuclide Cancer Incidence (Morbidity) Rate for Baseline, Manage-in-Place, and Retrieval Options (compared to EPA 10^{-4} and 10^{-6} cancer risk limits converted to annual bases).

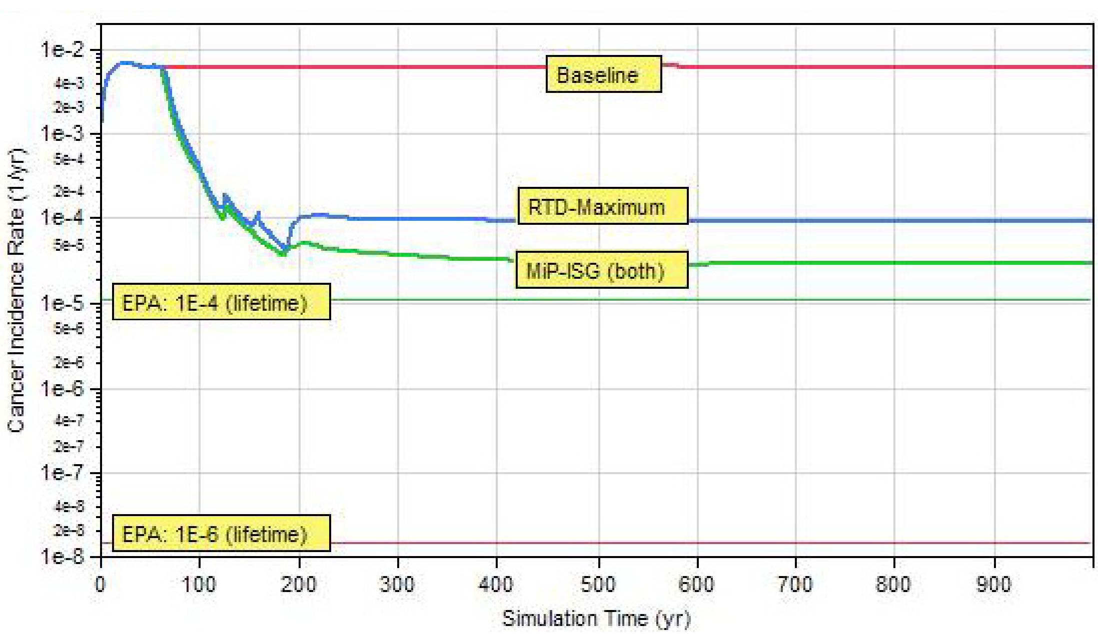

Figure 103. BCBG *DBRA-Expected* Case for On-Site Resident: Annual Chemical Cancer Incidence Rate for Baseline, Manage-in-Place, and Maximum Retrieval Options (compared to EPA 10^{-4} and 10^{-6} limits converted to annual bases).

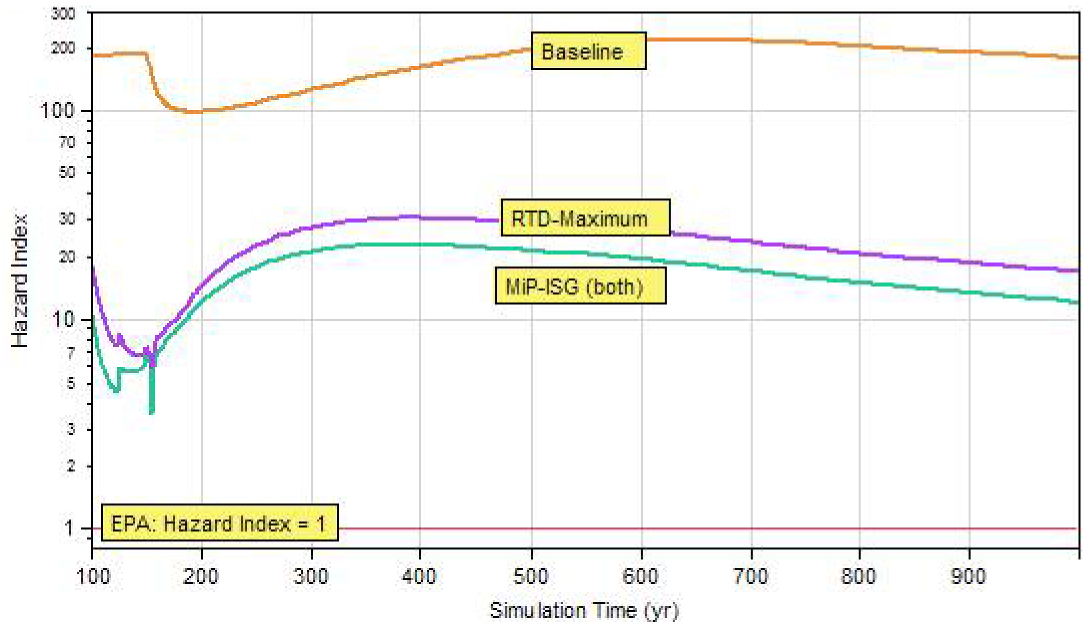

Figure 104. BCBG *DBRA-Expected* Case for On-Site Resident: Hazard Index for Baseline, Manage-in-Place, and Retrieval Options (compared to EPA limit of unity).

Figure 103 illustrates that the remedial alternatives considered in this research[189] are not expected to be capable of placing the BCBG in a protective state for even restricted release. The hazardous chemical that represents the majority of the post-closure chemical cancer risk for the BCBG was tetrachloroethylene (PCE). The results for non-cancer risks (as shown in Figure 104) also indicated that the proposed remedial actions would be ineffective based on model predictions. In this case, mercury was the primary, post-closure culprit. Furthermore, use of the maximum retrieval alternative results in increased predicted risk or hazard index compared to the manage-in-place alternative as illustrated in Figure 103 and Figure 104, respectively.

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[189] For the figures to follow, only the results for the manage-in-place (MIP) option employing *in situ* grouting (ISG) for contaminant immobilization are presented because this is the most effective MIP option. The maximum retrieval option is only shown for clarity because the targeted and maximum retrieval options are virtually identical for cancer risks and the targeted retrieval hazard indices are slightly lower than those for the maximum case but the differences are very small relative to the uncertainties.

However, even though the expected risk and hazard results indicated little chance of the proposed remedial actions placing the BCBG in a protective final state, the reasons for the ineffectiveness of the proposed actions may be important in defining new alternatives. The differences between the BCBG and the Idaho Site SDA that produced such different post-remedial results should be identified to provide input to the process of revising the remedial actions for the BCBG.

The likelihood of merely "tweaking" the remedial alternatives proposed for the BCBG to provide acceptable remedial solutions can be evaluated using the results of the stochastic evaluations. The chemical cancer risk results from the screening risk tool for the manage-in-place and maximum retrieval alternatives are illustrated in Figure 105 and Figure 106, respectively. For the predicted chemical cancer incidence rates, either remedial alternative appeared capable of producing a protective state (i.e., with a long-term risk less than the EPA "action limit") for some conditions. Therefore, there may be ways to apply either remedial alternative to the BCBG to reduce chemical cancer risks to acceptable levels for restricted release of the site.

The predicted non-cancer risk results for the manage-in-place and maximum retrieval alternatives are provided in Figure 107 and Figure 108, respectively. These results (especially for the mercury) did not show the same promise. The results in Figure 107 and Figure 108 for either remedial alternative were acceptable for some very limited sets of conditions, however, only for a brief interval and certainly not at the end of the assessment period. The non-cancer risks for the mercury exceeded the Hazard Index limit of unity in a relatively brief period of time. New remedial options must be defined for BCBG site clean up.

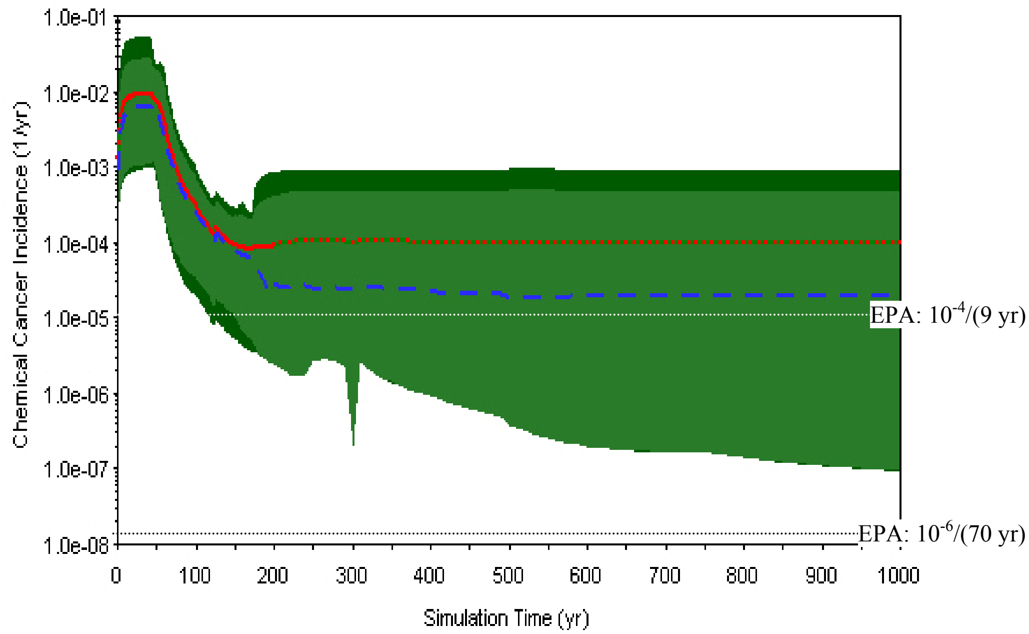

Figure 105. BCBG On-Site Resident Scenario: Uncertainties in Cancer Incidence Rate for All Pathways and Chemical for the Manage-In-Place Alternative. The red dotted line is the median, blue hashed line the mean, and the other bounds are the 95% and the upper and lower bounds.

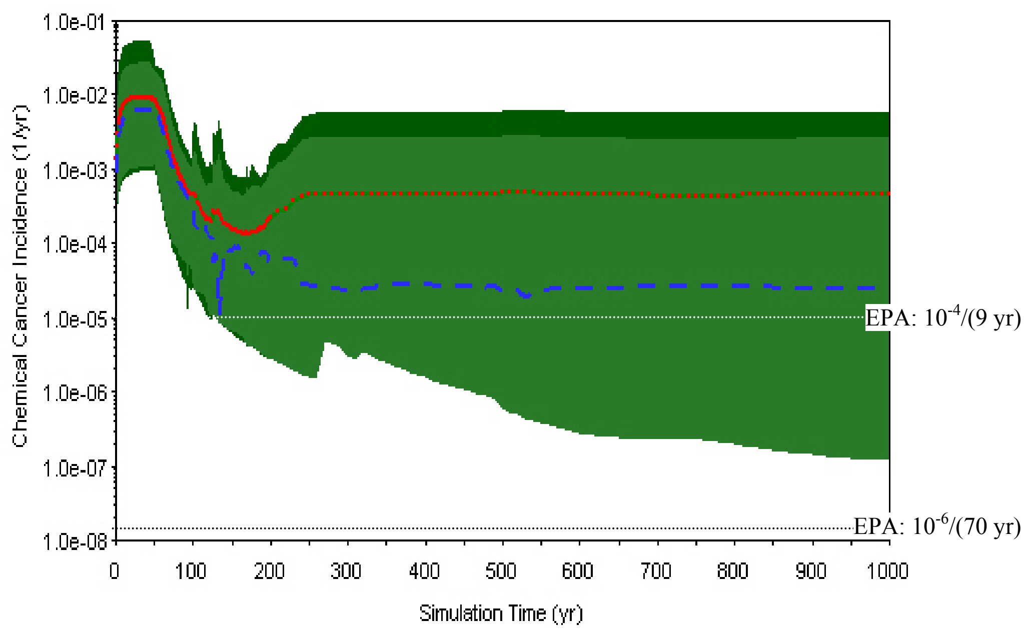

Figure 106. BCBG On-Site Resident Scenario: Uncertainties in Cancer Incidence Rate for All Pathways and Chemical for the Retrieval Alternative. The red dotted line is the median, blue hashed line the mean, and the other bounds are the 95% and the upper and lower bounds.

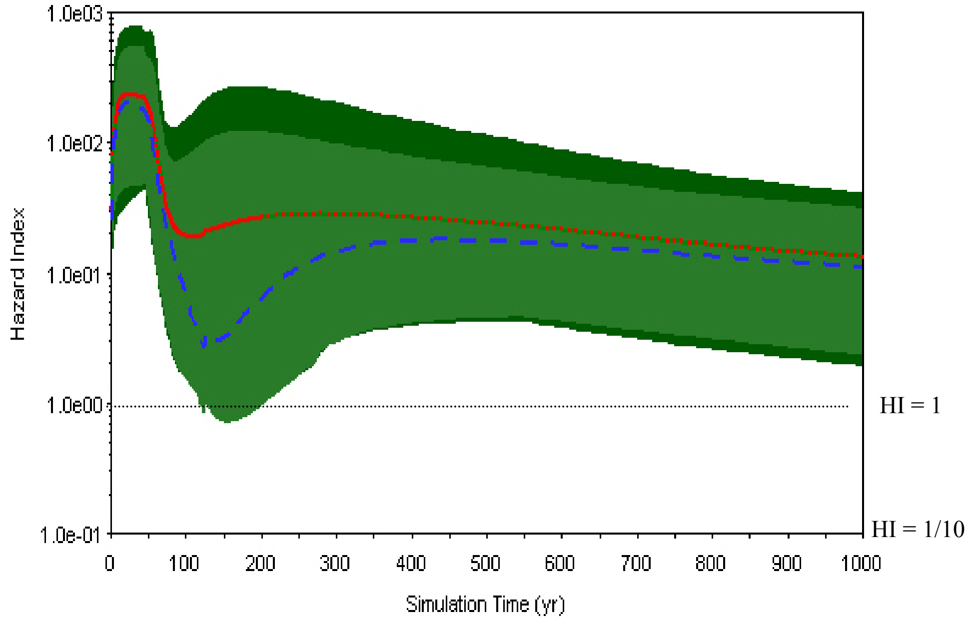

Figure 107. BCBG On-Site Resident Scenario: Uncertainties in Hazard Index for All Pathways and Chemical for the Manage-In-Place Alternative. The red dotted line is the median, blue hashed line the mean, and the other bounds are the 95% and the upper and lower bounds.

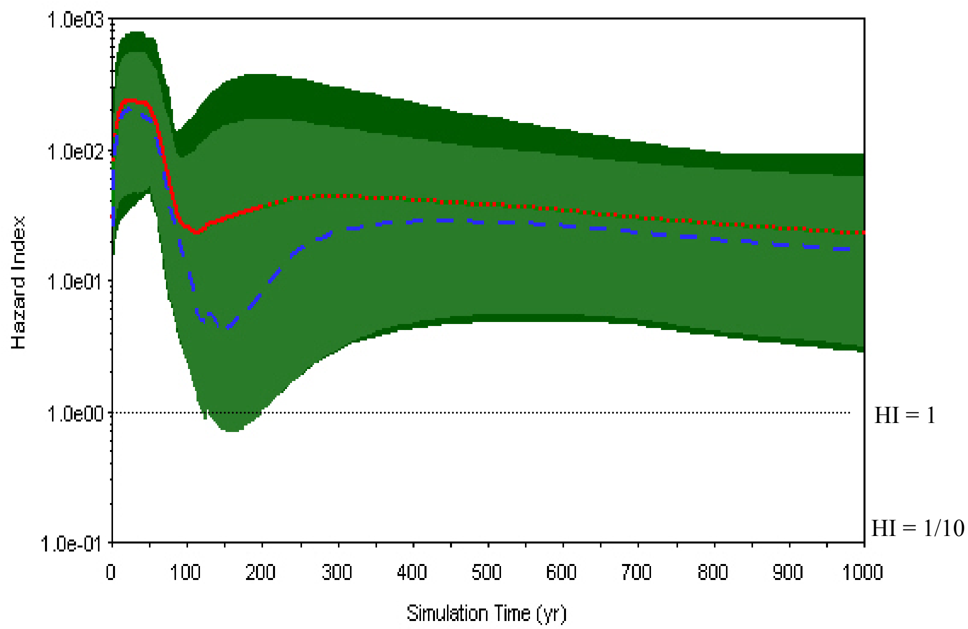

Figure 108. BCBG On-Site Resident Scenario: Uncertainties in Hazard Index for All Pathways and Chemical for the Retrieval Alternative. The red dotted line is the median, blue hashed line the mean, and the other bounds are the 95% and the upper and lower bounds.

BCBG: Impact of the Conceptual Model on Predicted Risk Results

The persistence of tetrachloroethylene (PCE) predicted by the screening risk tool might be impacted by enabling the organic degradation mechanism. However, PCE appears to be persistent in the BCBG environment based on sampling (SAIC 1996a) and thus degradation of PCE (or lack thereof) in the areas around the BCBG would not likely allow the remedial options considered to produce a final state that is protective for even restricted use. Organic degradation appears to have little impact on PCE and cannot affect mercury; therefore, the remedial actions considered will likely be ineffective in addressing all the contaminants of potential concern for the BCBG.

Furthermore, as indicated in Table 99 from Appendix D, PCE was buried loose in the areas from which wastes would be retrieved. The ineffectiveness of the proposed remedial for PCE might also be due to the fact that the material was buried loose and is assumed mobile in the environment. On the other hand, mercury was also buried loose but in areas not considered for retrieval because of the highly hazardous nature (e.g., unstable, explosive, or pyrophoric) of many wastes collocated with mercury. Additional remedial actions must be investigated for managing the risks related to PCE and mercury.

The predicted fluxes of PCE and mercury from the BCBG wastes to the environment were examined to identify what makes these contaminants problematic. The concentration and flux of PCE from the wastes to the surrounding media was relatively constant suggesting solubility limitations. When solubility limits were disabled, the risk contribution from PCE decreased over time as anticipated. Other factors may also be involved; however, this provides an area to begin examining how to improve the remedial effectiveness for PCE.

The predicted fluxes of mercury throughout the media were also examined to identify factors contributing to the unacceptable non-cancer risks posed to the on-site resident even after closure. The mercury appeared to be accumulating in the saturated zone which impacts the on-site receptor via both drinking water and dermal exposure from showering. When retardation was disabled, then the non-cancer risk contribution from mercury began to decrease significantly over time. However, because adsorption and solubility cannot be disabled in the actual environment, it would appear that the only way to assure that unacceptable cancer and non-cancer risks are not posed to on-site receptors would be to impose groundwater use restrictions. It is not known how this would be accomplished *in perpetuity*.

BCBG: Exposure and Standard Industrial Risks for Workers

An informed decision is made when all aspects of the buried waste disposition are considered and the risks posed to all potentially impacted receptors by the remedial actions that would be taken are judged in light of the risks averted or transferred. The screening risk tool can be used to estimate baseline risks and those associated with proposed remedial actions and the residual risks presented by the contaminants remaining on the site.

Because the remedial options considered in this research are not likely to produce a final state that is protective for even unrestricted release, an exhaustive analysis of remedial action worker risks similar to that for the SDA will not be made. As illustrated in the risk analysis framework in Chapter III, this is the time when new or revised remedial options would be investigated, if necessary. However, the development of new remedial alternatives is considered outside the scope of this research, and the development of new or revised remedial alternatives for the BCBG is recommended as additional research. However, because useful risk information was obtained in evaluating the adequacy of the remedial alternatives for the BCBG[190], the results are summarized for comparison to the qualitative results developed for the BCBG in Appendix B and Chapter IV as well as those quantitative results obtained for the SDA earlier in this chapter.

BCBG: Screening Quantitative Comparison of Remedial Alternatives

Based on the results presented in this chapter, neither managing the wastes in-place nor retrieving the wastes for treatment and disposal produces a protective final state for the BCBG. The ineffectiveness of these remedial actions appears to be due to the refractory nature of PCE (and its predicted carcinogenic impact) and the mobility of the mercury (and its non-carcinogenic effect) as modeled in the screening risk tool. Many constituents appeared to have been placed in the BCBG neither bound in a matrix nor contained in drums or other containers.

The risk information summarized in this chapter represents just one input to the risk-informed decision-making process. Factors other than risk may dictate the retrieval of wastes from the BCBG, perhaps even from areas where unstable, explosive, and pyrophoric materials were buried (which may require surface barrier removal). It is important to have a general idea of the risks posed both by potential exposures to workers and the general public to contaminants buried in the BCBG as well as by the remedial actions that would be required. Appendix B provides a detailed analysis of the risks

[190] It is possible that PCE will eventually be found to not be refractory in the areas around the BCBG and that wastes can be retrieved from the areas containing mercury collocated with highly hazardous materials including those that are unstable, explosive, or pyrophoric. Certain areas in question have already been capped under a previous RCRA closure action (SAIC 1996a; b).

posed by potential BCBG remedial actions. The injury and fatality risks to remedial workers who might be involved in retrieval activities (which are the most aggressive studied here) are presented in Figure 109 and Figure 110, respectively.

When compared to the worker risks for proposed SDA remedial activities, the BCBG worker risks are considerably higher primarily because of the unstable and pyrophoric nature of wastes that would be handled. Both the injury and fatality risks for the BCBG retrieval activities approach a value of one per year with a probability also approaching unity. Thus the risks related to possible excavation and retrieval activities in the BCBG are very high both in comparison to SDA worker risks and in absolute terms.

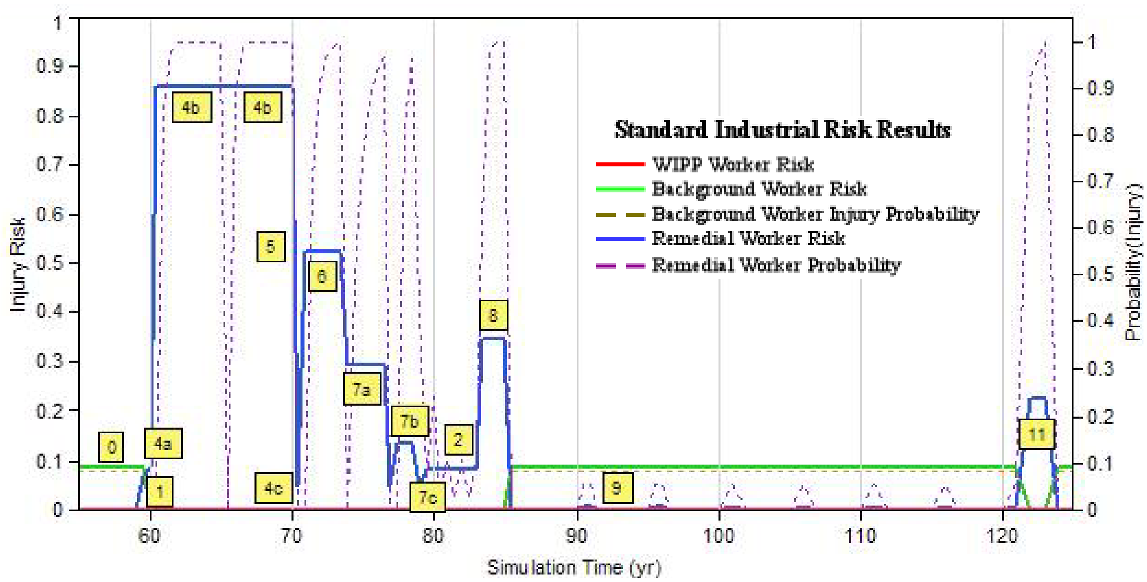

Figure 109. DBRA-Expected for Remedial Workers: Standard Industrial Injury Risks and Probabilities for Maximum Retrieve, Treat, Dispose (RTD) Scenario. (Steps correspond to those in Table 51.)

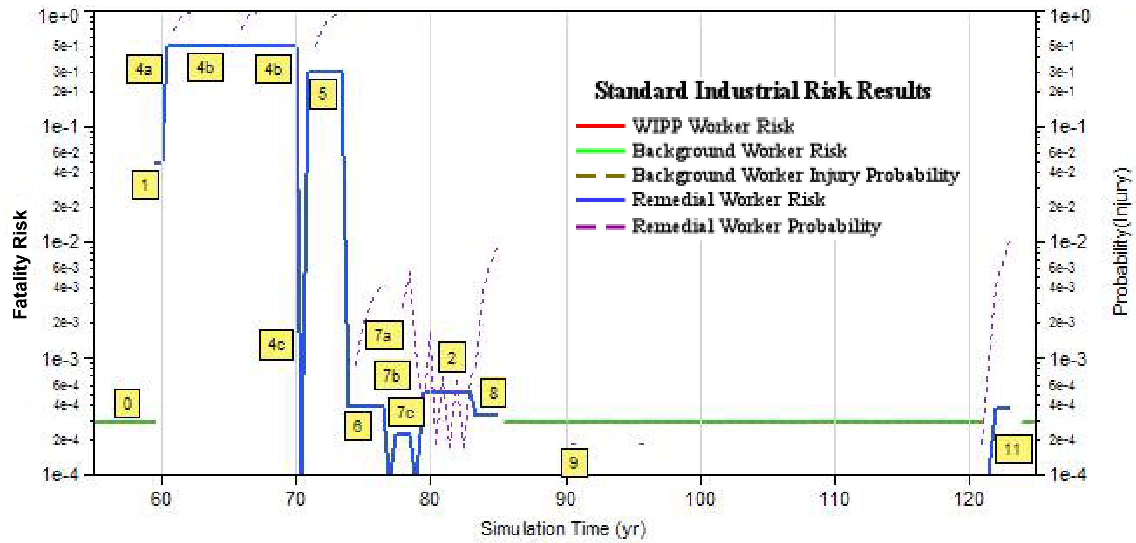

Figure 110. DBRA-Expected for Remedial Workers: Standard Industrial Fatality Risks and Probabilities for Maximum Retrieve, Treat, Dispose (RTD) Scenario. The semi-log nature of the diagram causes the breaks in lines representing predicted risks. (Steps correspond to those in Table 51.)

The results from the screening risk tool for proposed remedial alternatives were used to examine the original classifications summarized in Chapter IV. Figure 109 and Figure 110 provide indications of the injury and fatality risks and probabilities relating to accidents during proposed remedial actions. These remedial action risk and probability results are placed in the Table 54 (in the shaded columns). The definitions used to classify risks and uncertainties are described in the exhibits in Chapter III.

The maximum exposure fatality risk described in Chapter VI was estimated for the excavation and retrieval step as indicated in Table 54. Because this step involved potentially disturbing unstable and pyrophoric materials, the fatality risk estimate was very high (i.e., 0.8/yr) although the estimate was meant to be bounding. The intrinsically dangerous nature of the retrieval process was demonstrated by the fact that the fatality risk was estimated to be 0.5/yr for the routine case with a probability approaching unity.

Table 54. Summary of the Most Important Human Health Risks and Knowledge Gaps for the BCBG Remedial Alternatives

Process Step	1A. Baseline/No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the conse-quences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C) Updated
1. Character-ization	√	√	√	√	√	• Disturbing pyrophoric materials during sampling	• Probable	• Severe	• Worker	• Injury • Fatality	• Low • Low	• 7E-2 • 5E-2	• Presence/location of pyrophoric materials • Geospatial waste distribution	High (1,3,2,2) Low
2. ISG for Stab.		√		√	√	• Failure of high-pressure grout system	• Probable	• Severe	• Worker	• Injury • Fatality	• 2E-1 • 2E-3	• 8E-2 • 5E-4	• No *high-priority* gaps	High (1,4,13,1) Significant
3. ISG for Both			√	√	√	• Failure of high-pressure grout system	• Probable	• Severe	• Worker	• Injury • Fatality	• 2E-1 • 2E-3	• 8E-2 • 5E-4	• Geospatial waste distribution	High (2,4,20,1) Significant
4. Excavate, Retrieve, Segregate				√	√	• Uncovering pyrophoric materials • Contaminated soil removal and exposure • Loaded tote-bin dropped • Cave-in occurs during excavation operations • Exposure or injury from handling pyrophoric materials	• Probable • Probable • Probable • Possible • Probable	• Severe • Critical • Critical • Severe • Severe	• Worker • Worker • Worker • Worker • Worker	• Injury • Fatality • Maximum Exposure (Fatality)	• 1E+0 • 1E+0 • (d)	• 9E-1 • 5E-1 • 8E-1	• Future legal decisions • Geospatial waste distribution	High (3,11,35,3) High

a. *High-risk* hazards are 1) *probable* with either *critical* or *severe* consequences or 2) *possible* with *severe* consequences based on the definitions in Appendix B.

b. *High-priority* gaps are *critical* (in terms of safety) and *large* (meaning little or no information is available) as indicated in Appendix B.

c. The overall contribution for a process step is based on the hazard information provided in Appendix B using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

d. The probability of this maximum radiation exposure event cannot be determined from available information.

Table 54, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the conse-quences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C)
5. *Ex Situ* Treatment				√	√	• Containment system failure and injury • Fire or explosion during operations	• Probable • Probable	• Critical • Critical	• Worker • Worker	• Injury • Fatality	• 1E+0 • 1E+0	• 5E-1 • 3E-1	• No *high-priority* gaps	Significant (0,7,15,1) High
6. Packaging				√	√	• Containment system failure and exposure • Explosion during operations	• Possible • Possible	• Severe • Severe	• Worker • Worker	• Injury • Fatality	• 1E+0 • 5E-3	• 3E-1 • 4E-4	• Presence/location of unstable, explosive, and shock-sensitive materials	Significant (0,3,14,0) Significant
7. Storage and Disposal				√	√	• No *high-risk* hazards	• Not applicable	• Not applicable	• Not applicable	• Injury • Fatality	• 1E+0 • 8E-3	• 1E-1 • 2E-4	• No *high-priority* gaps	Low (0,0,11,1) Significant

a. *High-risk* hazards are 1) probable with either critical or severe consequences or 2) possible with severe consequences based on the definitions in Appendix A.

b. *High-priority* gaps are critical (in terms of safety) and large (meaning little or no information is available) as indicated in Appendix A.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

Table 54, Continued

Process Step	1A. Baseline or No Action	1B.Surface Barrier	1C. *In Situ* Grouting	2A Targeted Retrieval	2B. Maximum Retrieval	What can go wrong?[a]	How likely is it?	What are the consequences?	Who is impacted?	Risk Type	P(Risk)	Annual Risk (1/yr)	Highest Priority Information Gap(s)[b]	Overall Contribution to Risk[c] (H,S,L,N/C)
8. Barrier Installation		√	√	√	√	• Uncovering pyrophoric materials	• Probable	• Severe	• Worker	• Injury • Fatality	• 1E+0 • 9E-3	• 3E-1 • 3E-4	• No *high-priority* gaps	High (1,0,7,3) Significant
9. LTS	√	√	√	√	√	• LTS Failure (MiP) • LTS Failure (RTD)	• Probable • Probable	• Severe • Critical	• Public • Public	• Injury • Fatality	• 5E-2 • 2E-4	• 8E-3 • 3E-5	• Geospatial waste distribution	• High (1,7,8,1) Low • Significant (0,8,8,1) Low
11. Barrier Repair[e]		√	√	√	√	Same as LTS	Same as LTS	Same as LTS	Same as LTS	• Injury • Fatality	• 1E+0 • 4E-3	• 2E-1 • 4E-4	• Not determined	Not determined Significant

a. *High-risk* hazards are 1) probable with either critical or severe consequences or 2) possible with severe consequences based on the definitions in Appendix A.

b. *High-priority* gaps are critical (in terms of safety) and large (meaning little or no information is available) as indicated in Appendix A.

c. The overall contribution for a process step is based on the hazard information provided in Appendix A using the methodology described in Chapter III for "rolling up" hazard contributions to a single metric. Numbers in parentheses indicate number of failure mode events in a process step that are (*High*, *Significant*, *Low*, *Not Considered*).

When applying the classifications from Chapter III to the *quantitative* risk results in Table 54, all steps were deemed either *significant* or *high* in terms of their contributions to overall risk (where major barrier repair is considered a long-term stewardship activity). Although six of the process steps evaluated for the BCBG wastes were originally considered *high-risk* (as indicated in Table 20 in Chapter IV), only the excavation and *ex situ* treatment steps had quantitative risks that would be classified in a like manner. The risk for the treatment step was higher than that based on the original classification.

From the qualitative and quantitative information in Table 54, significant risks were associated with the actions employed in either the manage-in-place or retrieval alternatives. However, this fact does not mean that the alternatives cannot be compared based on the information developed in this research if one or both are later selected for partial treatment of the BCBG wastes. However, neither alternative would be capable of placing the site in a protective state without the ability to enforce groundwater use restrictions *in perpetuity*.

Employing the same rationale as in Chapter IV and for the SDA alternatives, it is very likely that any excavation and waste retrieval activities in the BCBG will be very risky due to the unstable, explosive, and pyrophoric nature of the wastes buried there. However, simply capping the BCBG will not effectively limit water from reaching those areas inundated with shallow groundwater. A more detailed and accurate assessment of risks, investigation of assumptions (e.g., organic degradation), classifications (demonstrating different value judgments), and examination of remedial requirements and alternatives may produce a set of remedial alternatives that would effectively treat the BCBG wastes without the necessary imposition of groundwater restrictions.

A risk-informed remedial decision may include non-risk factors (e.g., social values, past legal agreements, uncertainties, etc.) that are deemed significant by the decision-maker. As a result, either managing wastes in-place or retrieval as described in this research may be selected for partial treatment. If this is the case, it is hoped that the information developed in this research can drive the selected remedial actions to the minimum life-cycle risks possible. Furthermore, the process described in this research can be used to identify those additional process steps most likely to be dangerous to the remedial worker so steps can be taken during the planning stage.

BCBG: Hypothesis Testing

Primary Research Hypothesis: The retrieve, treat, and dispose alternative for the Bear Creek Burial Grounds (BCBG) located in the humid conditions on the Oak Ridge Reservation (ORR) will result in lower life-cycle risks than managing the wastes using *in situ* techniques.

The analysis of the predicted exposure risks to future on-site receptors indicated that neither the manage-in-place (MIP) nor RTD alternative would produce a final state that could be released for even restricted use without the imposition of groundwater use restrictions *in perpetuity*. Much of this had to do with the fact that wastes were originally buried loose and thus large amounts of contaminants have migrated into the environment significant distances. There are large contaminant plumes (especially organic) around and beneath the BCBG. However, despite the fact that contaminants have migrated, there remains a source that will add to the contamination without treatment. Because of the large risks associated with retrieval activities for the BCBG wastes, additional *in situ* treatment actions should be considered.

However, the primary hypothesis concerns relative life-cycle risks to receptors and not the protectiveness of the final state for on-site receptors. Putting the final state considerations aside for the moment, the possible excavation and retrieval of wastes in areas with large quantities of unstable and pyrophoric materials would pose unacceptably high worker risks as indicated in Table 54. Furthermore, the proposed retrieval actions would result in lower exposure risks for only radionuclides as illustrated in Figure 102, which may attenuate naturally to an acceptable state in a few more decades. The MIP actions resulted in lower predicted chemical cancer and hazard effects (as shown in Figure 103 and Figure 104, respectively), which drove the primary exposure risks associated with BCBG wastes. Because of the mixed results obtained for the RTD alternative, the primary hypothesis would be *rejected* for the BCBG.

Research Hypothesis: The remedial alternative that results in the lowest life-cycle risks to potentially impacted receptors is a combination of *in situ* techniques and targeted retrieval actions taken, if possible, in different areas of the disposal site.

For the targeted retrieval option, *in situ* grouting (ISG) was employed to immobilize contaminants, and thus this option represented a combination of techniques. However, the radionuclide and chemical cancer risks were virtually identical for the targeted and maximum retrieval options. The hazard indices were slightly lower for the targeted retrieval option than for the maximum case. Differences among predicted risks for the targeted and maximum retrieval cases were small relative to the uncertainties.

Furthermore, worker risks for retrieval actions in the BCBG tended to be large when compared to those for the *in situ* techniques. Therefore, the combination of *in situ* techniques and targeted retrieval actions did not appear to provide the lowest life-cycle risks. Therefore, this hypothesis would be *rejected*.

Research Hypothesis: The significant sources of exposure and accident risks for both general public and workers (in addition to non-risk factors such as costs, technical feasibility, cultural and societal impacts, etc.) must be considered for each remedial alternative for the decision to be risk-informed.

For the conditions and assumptions made in modeling the BCBG, this hypothesis could not be adequately tested. Because of the wide-spread contamination in the environment around the BCBG and the predicted ineffectiveness of the remedial activities studied, there was not sufficient information to judge this hypothesis. Additional research is required to provide the information needed to test this hypothesis. Different conclusions can be drawn because of incomplete information concerning potential remedial actions. The framework and screening risk tool developed in this research could be put to great advantage in deciding what additional information would be required.

Conclusions and the Consideration of Uncertainty in Site Analysis

The greatest contribution of applying the framework and screening risk tool to the prototypic sites was in demonstrating the impact that uncertainty could have on the remedial decision-making process. The results of the probabilistic analyses were used to reinforce the information illustrating that both sites pose unacceptable risks and were then used to focus attention on those risks and contaminants of potential concern (COPCs) that should receive the greatest initial attention. A probabilistic method for defining COPCs was provided and applied to both sites; the results were compared to those from the more traditional approaches.

The screening risk tool was used to demonstrate the potential effectiveness of the proposed remedial alternatives to the buried waste sites and estimate residual risks. When a decision is made to perform remedial actions at a contaminated site, the cost in terms of one set of risks or risk metrics is traded off against the benefits in terms of the same or different set of risks or risk metrics. For illustrative purposes, the upper limit of the benefits of applying remedial actions to a contaminated site was represented by the baseline risks (representing the maximum risk that could be reduced). The corresponding costs were represented by the worker risks. The analysis of these benefits and costs then represented one version of the risk trade-off associated with applying proposed remedial actions to a site.

The results of the screening risk tool indicated that the SDA might only be cleaned up to a restricted release status (regardless of the aggressiveness of the remedial approach). Furthermore, the manage-in-place and retrieval alternatives considered in this research appeared ineffective in placing the BCBG in a protective state for all contaminants (without permanent groundwater-use restrictions); new remedial options are needed for the BCBG.

The screening risk tool was used to develop estimates of the worker risks for the remedial actions likely to be performed on the prototype sites. Both injury and fatality risks (and probabilities) were assessed using the screening risk tool based on recent U.S. Bureau of Labor statistics.

For a contaminated site, the screening risk tool can assess whether the risks posed are unacceptable and, if so, identify the contaminants of potential concern based on dose, risk, and hazard metrics either deterministically or stochastically. Proposed remedial alternatives can be assessed for their potential effectiveness including uncertainties in the remedial endpoints and residual risks. The injury and fatality risks to workers performing cleanup activities can be estimated. All these predictions are made in an integrated platform using a consistent set of models, assumptions, parameters, etc. to provide both consistency and transparency to the various analytical steps required to provide the risk information needed for a remedial decision.

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CHAPTER VIII

CONCLUSIONS AND RECOMMENDATIONS

Before 1970, hundreds of thousands of cubic meters of transuranic (TRU), low-level, and mixed low-level wastes generated from nuclear materials production were buried at sites across the U.S. Department of Energy (DOE) Complex. Most of these wastes were buried in shallow unlined pits and trenches and covered with soil creating the potential for contaminant migration and exposure with concomitant safety and health concerns. The buried waste inventory is large and variable making assessment, retrieval, treatment, and disposal highly problematic. Inconsistency in regulatory approach and agreements concerning disposal alternatives (i.e., manage the wastes in-place or retrieve the wastes for treatment and disposal either on- or off-site) provides neither a consistent basis for site remediation nor transparency to a range of stakeholders.

To provide a foundation for risk-informed decision-making, a framework and methodology were developed as part of this research for the transparent and consistent technical evaluation of the life-cycle risks and risk trade-offs (both to the general public and workers) associated with buried waste disposition and site remediation. A screening risk tool was also developed in the GoldSim Monte Carlo simulation software as an integrated platform for estimating many of the risks needed for decision-making. Risk is just one of the inputs (along with costs, technical feasibility, cultural and societal impacts, etc.) needed to make a risk-informed decision. Use of this framework and screening risk tool to provide the risk information needed will differ from existing approaches by providing a basis for evaluating relevant risk tradeoffs involving the general public and workers over time in a consistent and transparent manner. The framework is tested using two DOE sites with very different climatic and subsurface conditions, the Idaho Site Subsurface Disposal Area (SDA) and the Oak Ridge Bear Creek Burial Grounds (BCBG).

Risk Analysis Framework

The goal of this research was the development and demonstration of a general life-cycle risk analysis framework for assessing the life-cycle risks and risk trade-offs associated with DOE buried waste disposition. The conceptual, graphical framework described in Chapter III outlines the general process for estimating and comparing the risks and risk trade-offs involved with either 1) managing buried wastes in-place or 2) retrieving, treating, and disposing wastes for disposal elsewhere[191]. The risk analysis framework is iterative and tiered so that each successive assessment phase builds on preceding phases and represents an increase in accuracy (e.g., for the parameters and models used) and site-specific information to better represent the uncertainties in the risk inputs to the decision-making process.

Risk Analysis Methodology

The methodology for the consistent application of the risk analysis framework to evaluate and compare the risks and risk trade-offs for the two general disposition alternatives for buried waste sites was developed in concert with the risk analysis framework as shown. The steps and types of information needed (e.g., diagrams,

[191] The risks and risk trade-offs associated with the needed transportation and final disposal of retrieved wastes are included to provide a comprehensive and life-cycle-based basis for comparison.

screening tools, etc.) to produce meaningful and transparent risk evaluations for proposed remedial actions for buried waste sites were defined and described in detail in Chapter III. A rational approach was provided for managing uncertainties and missing information so that this critical dimension of risk could be incorporated when comparing risks and risk trade-offs for proposed remedial alternatives. Guidelines for defining reasonable comparison metrics were provided and tested. The information developed in this research was demonstrated to provide an excellent basis for the evaluation and comparison of buried waste sites. However, the specific conditions and stakeholder input for any site to be evaluated must be considered; the framework and methodology developed in this research provide an excellent basis for adapting the risk analysis to a given buried waste site.

Framework and Methodology Application

The framework and methodology were applied to two prototypic sites to illustrate the effectiveness, flexibility, and value of the approach to promote consistency in planning for the disposition of buried waste in the DOE Complex. Previous experience indicated that the Oak Ridge Reservation (ORR) Bear Creek Burial Ground (BCBG) and Idaho Site Subsurface Disposal Area (SDA) were appropriate prototype sites. These sites appear to bracket the types of contaminants, hazards, and conditions expected from the various DOE buried waste sites.

The framework was applied in stages to the prototype sites. The initial, qualitative phase resulted in the critical components needed to guide the further analysis of risks for a given buried waste site. The critical components included

- *conceptual site models* which link sources of contaminants to potential receptors via transport pathways and exposure routes,
- *task lists* and *management flow diagrams* which show the steps needed to perform the possible remedial actions and the logical sequence of the steps,
- *risk flow diagrams* which, based on the framework provided by the management flow diagrams, indicate those process steps that are most likely to present significant human health risk,
- *detailed hazard and gaps analyses* describing the major hazards and knowledge gaps and uncertainties for the major process steps, and
- *summary tables* integrating the results from the detailed hazard and uncertainty analyses that are likely to impact the remedial decision.

Application of the risk analysis framework to two prototype DOE sites illustrated the effectiveness, flexibility, and value of the approach in providing the risk information needed to make an informed remedial decision. The effectiveness of the approach was demonstrated by showing that appropriate risk and uncertainty information was provided to decision-makers through application of the framework to DOE buried waste sites. The flexibility of the approach was demonstrated by applying the framework to two different DOE buried waste sites in very different climatic and geologic settings. These results implicitly demonstrated the value of the risk analysis framework to risk-informed decision-making.

Site-Specific Conclusions

Based on the risk and uncertainty information for the Idaho Site Subsurface Disposal Area (SDA) using the framework and methodology, either managing the waste in-place or retrieving wastes for treatment and disposal elsewhere would satisfy regulatory evaluation criteria for restricted use. Waste retrieval actions appeared to provide lower exposure risks from radionuclides and chemicals during the assessment period than the manage-in-place options; however, retrieval actions could not create post-closure conditions in the SDA for unrestricted use and posed significantly increased worker risks from exposures and accidents. Installation of a surface barrier appeared to provide the minimum life-cycle risk remedial option for the SDA buried waste site. One major reason for this result is that aggressive actions like excavation and retrieval posed significant additional worker and general public risks both from contaminant exposure and accidents. However, because factors other than risk must be considered to make a risk-informed decision, retrieval actions targeted on the highly mobile wastes that represent the highest, short-term risks[192] may be the risk-informed decision that would be selected by regulators.

For the Oak Ridge Bear Creek Burial Grounds (BCBG), neither managing the wastes in-place nor retrieving the wastes for treatment and disposal would produce acceptable post-closure conditions for even restricted use. Because many of the wastes buried in the BCBG were loose or in liquid form, contaminants have been migrating from the site for decades and large plumes have been identified. Contaminant migration makes restoration of the site very difficult and contributes to the predicted ineffectiveness of the remedial actions studied. Retrieval actions appeared prohibitively hazardous for remedial workers, especially in those areas that contain unstable and pyrophoric materials. If retrieval actions are selected for other BCBG wastes, assurances are needed that unstable materials will not be disturbed or pyrophoric materials exposed to air. The comparatively

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[192] One issue that should be considered is whether, by the time remedial actions can be taken to influence highly mobile contaminants, it is too late to make a significant difference in the risks posed by site wastes. A recent legal decision indicates that all transuranic wastes in the SDA originally from the Rocky Flats Plant must be retrieved for disposal at the Waste Isolation Pilot Plant (WIPP) despite the risks involved.

simple action of characterizing BCBG wastes in these areas may be highly hazardous; the uncertainty in locating these wastes makes excavation and retrieval operations highly hazardous. It is foreseen that remedial actions employing *in situ* techniques in these areas will pose minimum life-cycle risks.

This research was developed to promote a broader discussion among DOE, regulators, public representatives, and the general public on the most appropriate path forward for disposition of DOE buried wastes. Risk is but one of several important aspects that must be considered in decisions impacting public welfare. Imperfect and incomplete information, inherent variability and uncertainty, and differences in individual values and perspectives will undoubtedly lead to differing views on the appropriate path forward. These differences highlight the necessity for a clearly defined and engaged stakeholder participation process as an integral part of the on-going decision and management process for wastes buried in the DOE Complex.

Screening Risk Tool

Evaluation of the life-cycle risks for the disposition of a buried waste site involves many evaluations including baseline, remedial action, and residual risks to possibly different types of receptors including workers and the general public. A first-of-a-kind integrated risk screening tool was developed using the GoldSim Monte Carlo simulation software to provide screening estimates of the exposure and standard industrial risks associated with remedial actions for DOE buried waste sites. This tool integrates the ability to estimate and evaluate exposure and standard industrial risks for the baseline, remedial action, and residual conditions that constitute the life-cycle of the buried waste site management process. The broad nature of the source term, fate and transport, exposure, and receptor implementations lends itself to typical baseline risk evaluations but is extended to consider life-cycle and remedial actions to a similar level of detail.

The screening risk tool was applied to both prototypic sites to evaluate the risks and risk trade-offs expected from either managing wastes in-place or retrieving wastes for treatment and disposal. For the SDA, retrieval operations would likely trade large additional remedial worker risks for little obvious benefit in terms of reduced long-term exposure risks to the general public. The manage-in-place alternative appeared preferable in terms of the predicted risk trade-offs. For the BCBG, the results indicated that neither alternative would be effective in cleaning up the site to even restricted use because of the large contaminant plumes already in the environment. However, results from the screening risk evaluation suggested that *in situ* techniques for residual source control should be investigated for the BCBG because of the highly hazardous nature of many wastes buried there.

For both sites, the results from the screening risk tool as described in Chapter VII compared favorably to those from applying the framework in Chapter IV using expert judgment, especially in terms of identifying *high-risk* tasks in the remedial process[193]. The initial results in Chapter IV were very useful in describing the major hazards and knowledge gaps in an understandable fashion. The results from the screening risk tool for both sites provided quantitative estimates of exposure doses and risks from radionuclides and chemicals and accident risks from performing proposed remedial actions. However, because of the large uncertainties involved with the exposure and risk estimates, both sets of information are seen as critical inputs to risk-informed remedial decisions.

[193] The screening risk tool essentially implements three quantitative parts (i.e., Phases 2A, C, and D) of Phase 2 (*Screening Quantitative Baseline and Remedial Action Risk Analysis*) of the framework and methodology defined in Chapter III.

Significance and Contribution of this Research

The primary goal of this research was to develop and demonstrate a comprehensive, life-cycle risk analysis framework and methodology for the disposition of DOE buried waste sites that is straightforward and efficient to apply and results in a *consistent* and *transparent* evaluation of the life-cycle disposition risks. To truly improve the risk assessment process and its acceptance requires focusing efforts on consistency, transparency, and trust in the process and reducing uncertainty in the technical components that will likely never be fully understood (nor perhaps trusted) by some regulators and many stakeholders alike. The risk analysis framework and methodology developed in this research provides a mechanism for providing the consistency and transparency that has not been delivered by other frameworks.

The methodology for applying the framework to the disposition of buried waste sites promotes consistency and transparency in developing the risk information needed for informed decision-making. The methodology required development of the following elements critical to risk communication:

- *Improved site conceptual models* (CSM) were developed for baseline conditions linking contaminant sources to potentially impacted receptors (both general public and worker) and describing graphically why remedial action is likely either to be required or not. The improvements in the CSM resulted from generalization of common elements for buried waste sites, indication of the temporal nature of the risks, and more obvious presentation of the risk drivers.

- *Comparison metrics* were defined that formed a reasonable basis for how exposure and corresponding information (e.g., dose, risk, hazard quotient, etc.) obtained from the analysis could be compared. A method for moving to risk-based comparisons for specific conditions was outlined.

- For each acceptable remedial alternative[194], the following information was required:
    - *Task list* and corresponding *management flow diagram* were needed to outline the process steps and tasks required to perform the remedial alternative. The management flow diagrams represent the flow of remedial tasks and can be further conceptualized in terms of "pinch-points" for the remedial alternative.
    - *Novel conceptual site models* were generated relating the natures of the hazards and risks during remedial activities to potentially impacted receptors. These models extended the CSM concept to remedial actions to better characterize the risks associated with the remedial actions.
    - *Hazard analysis* was performed that identified (for each process step) the task frequency, elements of risk, potentially impacted population, basis for characterizing the risk, and contribution of the remedial task to overall risk.
    - *Novel risk flow diagram* was developed to indicate the sequence of remedial and stewardship activities with potential to pose significant human health risks. This novel diagram was based on the corresponding management flow diagram but illustrated graphically which steps were most likely to present significant human health risks.
    - *Gap analysis* was performed describing the key knowledge barriers, missing information, variabilities, and uncertainties involved in assessing risks for the remedial alternative.
    - *Novel integrated hazard and gap analysis* was generated summarizing the most important potential risks and information gaps for the remedial alternative. The initial version of this table was based upon expert opinion and is updated or supplemented when quantitative results are obtained.
- *Life-cycle risk breakdown and comparison* was produced indicating the life-cycle risks for proposed remedial alternatives as they relate to types of risks and potential receptors. An explicit declaration of the value judgments and simplifying assumptions made by the risk assessor must be made as well as the likely impact of significant uncertainties on the risk estimates.

The above components of the risk analysis information help to focus the risk analysis process during subsequent phases and provide a basis for the consistent and transparent comparison of potential remedial alternatives.

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[194] A usual and customary way to screen potential remedial alternatives is using three (i.e., effectiveness, implamentability, and cost) of the nine CERCLA evaluation criteria per the National Oil and Hazardous Substances Pollution Contingency Plan (CFR 1994).

There are a number of additional qualities of the risk analysis framework and methodology that lend consistency and transparency and ultimately trust to the risk results obtained. Early and continued stakeholder input is required. The risk assessment process is *tiered* so that the level of detail in the analysis of both risk and uncertainty and the types of simplifying assumptions tolerated are commensurate with the importance, complexity, and stage of the buried waste site disposition. The risk analysis framework is *iterative* so that the risk assessment can be updated as new information is obtained, new questions are asked, or regulations are changed. Risk assessment should be thought of as a journey much more than a goal. This journey, to which all interested parties are enjoined to early participation, addresses all relevant types of risk and considers the impacts of uncertainty consistently and transparency so that trust can be engendered. Explicit consideration is given to diverse populations over both immediate and long-term time frames.

The risk assessment framework and methodology also integrates the concepts of exposure *and* standard industrial risks to all potentially impacted receptors—both in the general public and workers. Despite the advances in risk assessment techniques, there is often a conspicuous absence of the consideration of standard industrial risks[195] in many risk assessment approaches despite indications that the predominant source of risk in site cleanup is industrial or occupational in nature (Applegate and Wesloh 1998; Gerrard and Goldberg 1995). However, as important as the question of who is at risk is the question of when they are at risk. Workers tend to be most exposed and thus at risk during remedial

[195] Standard industrial risks are non-exposure risks associated with falls, explosions, transportation accidents, etc.

actions; whereas, general public exposures may last for millennia. The temporal aspects of risk are integrated into the evaluations of risk in the framework developed here.

A novel risk screening tool was developed in the GoldSim Monte Carlo simulation software that integrates many of the concepts in the life-cycle risk analysis framework and methodology. Although the primary product of this research is *not* software, the screening risk tool was developed to incorporate these basic concepts (i.e., integrating exposure and industrial risks, public and occupational receptors, temporal variation in risks, sensitivity and probabilistic capabilities, etc.). The screening risk tool can be used to apply the concepts of the risk analysis framework and methodology to a buried wastes site in one integrated platform to evaluate the risks for proposed remedial alternatives for a buried waste site. An example of the usefulness of the screening risk tool was the development of a more rigorous, probabilistic basis (i.e., using exceedance curves) for identifying the contaminants of potential concern as part of the baseline risk assessment process.

No risk analysis framework or software tool can decide what should be done with a contaminated site. However, the life-cycle risk analysis framework and methodology and the results from applying the risk screening tool can effectively organize the evaluation process and assure that the evaluation is performed in a consistent and transparent manner. The risk and uncertainty results obtained from the evaluation can then be used as the risk input to the risk-informed decision-making process.

Recommendations for Future Research

The risk analysis framework and methodology developed in this research has been established as a mechanism to bring consistency and transparency to the evaluation of risks associated with DOE buried waste sites. However, additional research could improve both the bases for risk evaluation and comparison and acceptance of the methodology for input to the risk-informed decision-making process. Additional research should include:

- providing a more rigorous and consistent methodology and bases for the *comparison* of the risks and risk trade-offs for potential remedial alternatives,
- applying the framework to additional DOE and other buried waste sites to better reflect its usefulness and flexibility,
- expanding the description of the more detailed phase (i.e., Phase 3) of the risk analysis framework,
- defining a more rigorous basis for estimating the probability of exposure by leveraging the methods (e.g., fault tree, event tree, etc.) used in probabilistic or quantitative risk assessment (Garrick 2007),
- developing a screening conceptual site model (CSM) development tool[196] based on the general CSM's defined in this research,
- examining the impacts of "early remedial actions" (e.g., vacuum vapor extraction, capping, etc.) on resulting life-cycle risks,
- examining the impact on the soil resuspension model when a shallow surface soil layer (e.g., 15 cm) is used to represent the resuspension layer instead of the entire surface soil layer,
- examining changes in how the depth of the layer accessible to biota is defined in relation to the maximum depth for the plant and animal species,
- treating workload and durations for injury and fatality risk stochastically,
- examining impacts of alternative exposure scenarios,
- examining impacts of organic degradation on risk results and develop a more accurate model of degradation in the environment,
- presenting a more refined analysis of risk versus pathway for receptors,

[196] An example of one such tool is the U.S. Department of Energy (DOE) Site Conceptual Exposure Model (SCEM) Builder available at http://homer.ornl.gov/nuclearsafety/nsea/oepa/tools/scem.html (Accessed November 10, 2007).

- expanding the receptor analysis to be less focused on typical receptor scenarios (e.g., on-site resident, etc.) and more on the distribution of receptors neighboring the site, and

- exploring alternative risk metrics (e.g., years of potential life lost, expected loss of lifetime, etc.) to better describe the temporal impacts on receptors.

A screening risk tool was developed in the GoldSim Monte Carlo simulation software to generate the screening quantitative information called for in the risk analysis framework and methodology. Further enhancements to this tool could include:

- evaluating sensitive population effects and providing a better representation of the potentially impacted population over time,

- implementing additional transport pathways (e.g., DNAPL movement through the subsurface, atmospheric pathway dispersion, deposition, etc.) and media including both porous and fractured subsurface capabilities,

- providing a more general implementation of remedial alternatives,

- improving the methodology for evaluating standard industrial risks including breaking down work load by activity instead of worker type and industry,

- providing a more facile and easier-to-understand risk comparison information,

- implementing a two-dimensional Monte Carlo simulation, when needed, by treating those parameters that are most sensitive to variability versus uncertainty, and

- providing additional remedial options (e.g., slurry wall).

Final Thoughts

It is the hope of the author that the elements of this research can be folded into the site assessment process when and where needed. The elements of the risk analysis framework and methodology developed in this research are not meant to be prescriptive but instead to generate new ideas for lending consistency and transparency to the remedial decision process. The remedial process will not ultimately be successful without a renewed trust between stakeholders and the U.S. Department of Energy (DOE). However, this must be a two-way street. The DOE must trust that stakeholders will listen if the information communicated to them is consistent and the bases for decisions transparent. Stakeholders must be willing to listen with "new ears" and with trust until trust is no longer warranted. The process must be restarted.

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