ABSTRACT

OTTINGER, KEITH EVERETTE. A Radioactive Waste Cleanup Decision Making Framework. (Under the direction of Dr. Man-Sung Yim.)

Many sites in the United States and around the world are contaminated with radioactive waste. Because of the public perception of radiation, making economically feasible and socially acceptable radioactive waste cleanup decisions is often difficult. The current solution to this problem is to have the public participate in the decision making process. This allows for more disclosure to and input from the stakeholders which are supposed to make the decision more socially acceptable. The results of this approach have not always been good and meaningful public involvement takes time and resources. In this work a decision making framework is developed, which attempts to solve this problem by creating a method for quantifying the value of the public's perception of the contamination and cleanup options. These values are then input into a multi attribute value theory analysis which includes the direct costs of the cleanup and dose, and attempts to find socially viable solutions for the decision. The framework is complete, however, more research is need for to be able to adequately quantify the public's perception.

A case study based on the contamination at the Radioactive Waste Disposal Site (RWDS) of the Kurchatov Institute (KI) in Moscow, Russia was performed to illustrate how the framework works. This case study was primarily based on published data of the contamination levels and other important parameters as of 2003. However, some of the required data was not available or very uncertain and the site has since undergone remediation so the results may not represent the actual KI RWDS. In this case study the

optimum cleanup method was determined to be covering the site with clean fill for discount rates of 5 percent or less and no action for higher discount rates. This result seems reasonable but more research is needed to determine if it fully represents the public's perception of the contamination or the cleanup alternatives and if not how to improve the model.

A Radioactive Waste Cleanup Decision Making Framework

by
Keith Everette Ottinger

A thesis submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Master of Science

Nuclear Engineering

Raleigh, North Carolina

2009

APPROVED BY:

Peter Bloomfield	David McNelis

Man-Sung Yim
Chair of Advisory Committee

BIOGRAPHY

I was born in Nashville, TN in 1985. I latter moved to Lenoir City, TN were I attended Lenoir City High School. I graduated from this high school in 2003. I then went to college at the University of Tennessee: Knoxville where I studied Nuclear Engineering and graduated summa cum laude with my B.S. in Nuclear Engineering in May 2007. I am currently attending North Carolina State University.

ACKNOWLEDGMENTS

I would like to acknowledge my advisor Dr. Man-Sung Yim for continuous help and support throughout this research. I would also like to acknowledge Dr. McNelis for his many helpful comments and suggestions about this project. Dr. Terry Sullivan's help with the DUST-MS computer code was instrumental in the groundwater pathway analysis and he also provided information on the DECERNS framework. I would also like to acknowledge the George Russell Foundation for providing support for this work.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Chapter 1 Introduction

Cleanup decisions for sites contaminated with radioactive waste have become increasingly dependent on public involvement over the past several decades. This is because regulators have realized the benefit of public involvement in making socially acceptable decisions. There has, however, been some disagreement as to whether or not this has improved the quality of decisions made or, in some cases, if it has even improved the acceptability of the outcome for the concerned parties [1].

However, even if it is assumed that public involvement increases the quality of a decision, some decisions will still have to be made without public involvement, because it is not always possible to obtain public input. For example, public involvement in the decision-making process may not be practical for a site that requires only a limited remediation such that the cost of public involvement would be high relative to the cost of the cleanup, or a spill/site that is determined to be a significant immediate threat to the public which would require action before public involvement could be obtained. A model that does not require public involvement might also be used for scoping analyses prior to public involvement on larger problems. The purpose of this work is to design a framework for making radioactive waste cleanup decisions that are economical and socially acceptable without direct public involvement in the decision making process.

To do this the public's perception of the contamination and the cleanup alternatives must be quantified and valued. In this research a risk aversion factor based on the magnitude of the individual annual dose is used to calculate the value of the public's perception of the contamination. The risk aversion factor was modified by including factors related to the decision to calculate the change in value of the public's perception of the contamination for the cleanup alternatives.

The first step in the development of the framework was to perform a literature review of existing frameworks and methods for quantifying risk perception. The information gathered in the literature review is used to develop a decision making framework. After the decision making framework is presented a case study on the Radioactive Waste Disposal Site (RWDS) at the Kurchatov Institute (KI) in Moscow, Russia is performed. This case study is based on historical data from 2003, because the KI RWDS has been remediated since then, via removal of the contaminants to an offsite disposal site. Due to a lack of data and a desire to test the framework several assumptions were made and some scenarios were invented. These assumptions and scenarios altered the dose calculation results, and, consequently, the results in this work may not represent the expected dose from the actual conditions or probabilities of the occurrence of events at the KI RWDS.

Chapter 2 Literature Review

2.1 Introduction

The open literature was searched for decision frameworks dealing with the remediation of radioactively contaminated sites and also for methods of quantify the public's perception of radioactively contaminated site and possible cleanup alternatives for such sites. Several decision frameworks were found in the literature that were designed specifically to help make decisions involving radioactive waste, and two papers addressing the issue of how to quantify the publics perception to nuclear issue were reviewed. Three decision frameworks were examined as part of this research.

2.2 Decision Frameworks

2.2.1 Thomas Flueler's Framework

In the first of these, Thomas Flueler proposes a "sustainability triangle" specifically for dealing with the issue of spent nuclear fuel storage/disposal. In Flueler's framework tradeoffs are made between economy (costs), ecology (protection), and society (acceptance) to achieve a sustainable solution to the problem (one which meets all of these criteria in the present and future), as shown in Figure 2.1. Among other things Flueler recommends direct public involvement as a way to gain public trust and make better (more socially acceptable) decisions with regard to locating and building a spent fuel repository. [2]

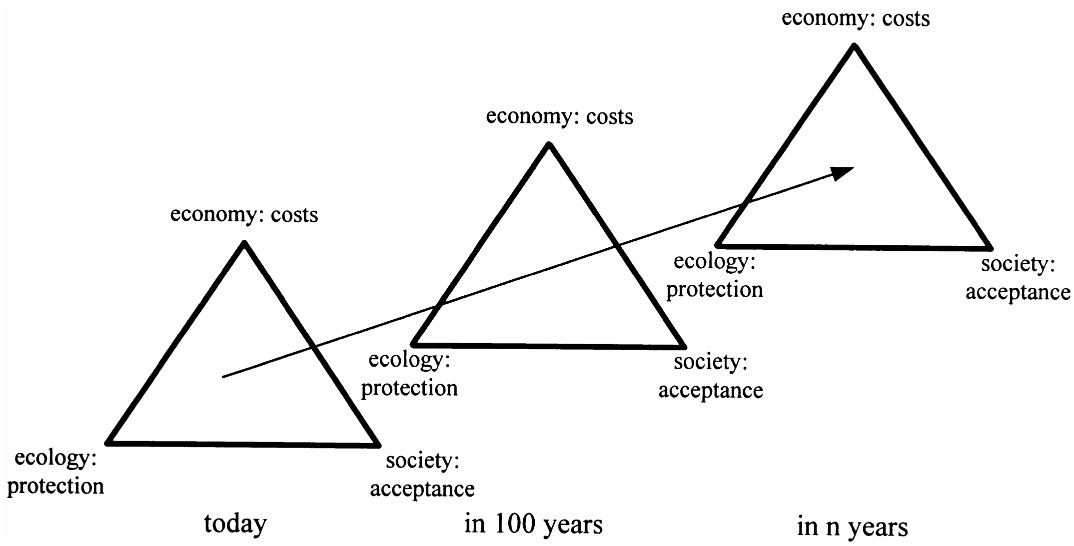


Figure 2.1 - A graphical representation of Thomas Flueler's "sustainability triangle" throughout time [2].

2.2.2 KONVERGENCE

The KONVERGENCE framework, which was developed at INL for making radioactive waste cleanup decisions, was also examined. Its basic premise "where ***Knowledge***, ***Values***, and ***Resources*** converge…you will find a sustainable solution" is similar to Flueler's, but the accompanying framework is more detailed. The bold/italicized terms in the above definition are defined in the framework as follows:

- Knowledge – What is known about the problem and possible solutions
- Values – What is important to those affected by the decision. Regulations are considered an imperfect overlay of values and are not used directly in the decision making process.

- Resources – What is available to implement possible solutions and improve knowledge

A basic outline of the framework and a Venn diagram representation of the mental model mentioned earlier is presented in Figure 2.2. In this figure, the "values" region has a participation arrow pointing to it indicating that public participation is the source of the values used in the framework. [3]

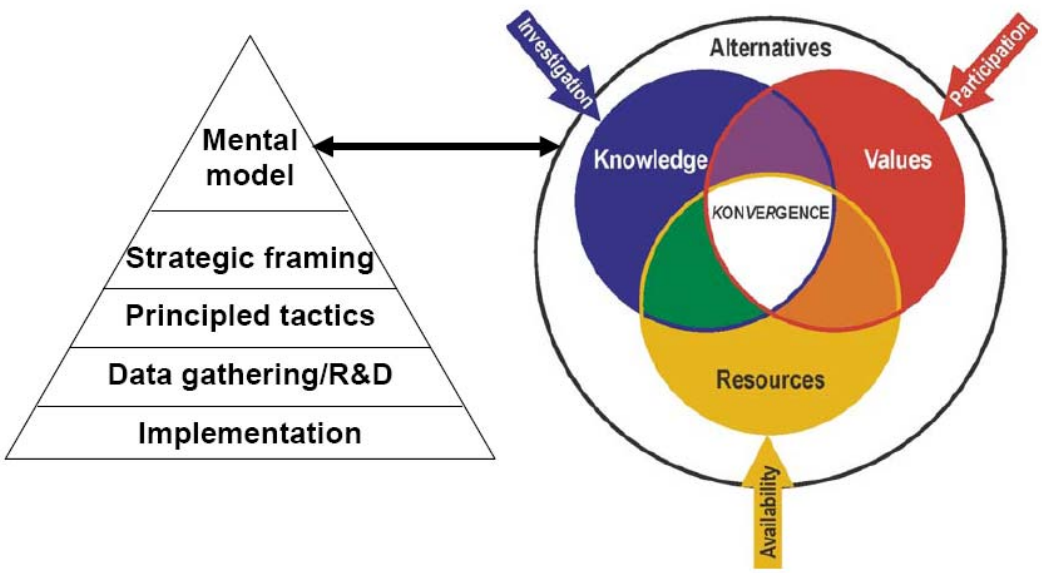


Figure 2.2 - KONVERGENCE framework and a graphical representation of the accompanying mental model [3].

The second step in the KONVERGENCE framework, strategic framing, is done by framing the problem in such a way that none of the cleanup alternatives has an advantage before any analysis is done. The principled-tactics step typically involves gathering the values of the stakeholders, but, in the absence of stakeholder specific, values a generic list is supplied (Table 2.1). Data gathering and R&D is the process of obtaining detailed data about the site, contamination, and possible cleanup alternatives. Implementation is the step in which the actual decision is made, applying all of the information obtained in the previous steps. [3]

Table 2.1 - Generic list of values used in the KONVEREGENCE framework [3].

Equality – the decisions are fair and just for current and future generations • **Trustee Principle** – "Every generation has obligations as trustee to protect the interests of future generations."(7) • **Sustainability Principle** – "No generation should deprive future generations of the opportunity for a quality of life comparable to its own."(7) • **Chain of Obligation Principle** – "Each generations' primary obligation is to provide for the needs of the living and succeeding generations."(7) • **Precautionary Principle 1**– "Actions that pose a realistic threat of irreversible harm or catastrophic consequences should not be pursued unless there is some compelling countervailing need to benefit either current or future generations."(7) • **Precautionary Principle 2** – "Where there are threats of serious or irreversible damage, scientific uncertainty shall not be used to postpone cost-effective measures to prevent environmental degradation."(8)
Democracy – the decision-making process is open with participation by all • **Involvement Principle** – The process should incorporate meaningful community and stakeholder involvement in all phases of decision-making now and in the future. • **Information Principle** – Complete, accurate and useable information should be provided to both current and future peoples. • **Invisible Man Principle** – The decision, the decision process, and supporting information must be transparent and understandable by interested parties now and in the future. • **Poisoning of the Well Principle** – Don't poison the "well" for future decisions. The process should make future decisions involving related problems and stakeholders easier by improving the decision environment. • **Tip of the Iceberg Principle (or Canary in the Coal Mine Principle)** – Without granting veto power to individual participants, concerns must be noted, addressed to the extent possible, and the risk of proceeding in the face of strong concerns considered before proceeding.
Truth – the decision should reflect the truth, the whole truth and nothing but the truth • **Uncertainty Principle** – There will be large uncertainties in the knowledge about the hazards, the facility and its environs especially their future behavior and performance. These uncertainties need to be acknowledged, documented and communicated with all involved. • **Faber College Principle** – Knowledge is good. To be able to make a sound decision, knowledge about the contaminated situation is essential. Research will be pursued if complete understanding is not possible. • **Forest and the Trees Principle** - Understand the characteristics and context of the land and facilities near the site or facility in question. Actions that might make sense in one location may not make sense in another. • **Price is Right or Fram Oil Filter Principle** – The stakeholders have a need and a right to know not only what the cleanup activity will cost but what the life cycle costs will be.
Reason – the decision should be real, practical, and meaningful • **What if You are Wrong Principle** – Decisions must withstand the test of time amid great uncertainty. • **Paul Masson Principle** – No decision should be made before its time. • **Perry Mason Principle** – Decisions must comply with the intent of environmental regulations regardless of current language or interpretation, e.g., protective of human health and the environment. • **Hippocratic Worker Principle** – Above all else, do no harm to the current worker especially when considering minimal hypothetical future risks. • **Little Engine that Could Principle** – The decision should lead to actions that are achievable, not necessarily easy, but doable with existing resources. • **Snicker Principle** – The decision should be able to pass a snicker test by participants before implementation.

Both of these frameworks rely on public input to increase the social acceptance and consequent sustainability of the chosen solution. This makes sense when the problem is large and a great number of people are affected by it, because the cost and delay required to get public input will be small and short, respectively, when compared to the cost and duration of the cleanup. However, for small problems or situations where serious harm is being done or about to be done, it may be better to skip direct public involvement and try to approximate the effects that public involvement would have had on the decision, by quantifying the public's perception of the problem and possible solutions. To do so, it will be necessary to take into account the public's thoughts about the risks posed by the site and the benefits of the cleanup alternatives. This could be accomplished by analyzing the contamination and cleanup options with respect to a list of the public's values such as the one provided in Table 2.1.

2.2.3 DECERNS

The third work reviewed was the Decision Evaluation in Complex Risk Network Systems (DECERNS) framework, which was developed to make decisions related to sustainable land use. DECERNS includes a decision making framework like the previous examples (Figure 2.3), but unlike the others it also contains tools that perform quantitative analysis, allow the user to run scenarios based on land use via a GIS system, create decision trees, and perform uncertainty/sensitivity analysis to aid in the decision making process. Except for these tools, the actual decision making framework is similar to the KONVERGENCE framework. [4]

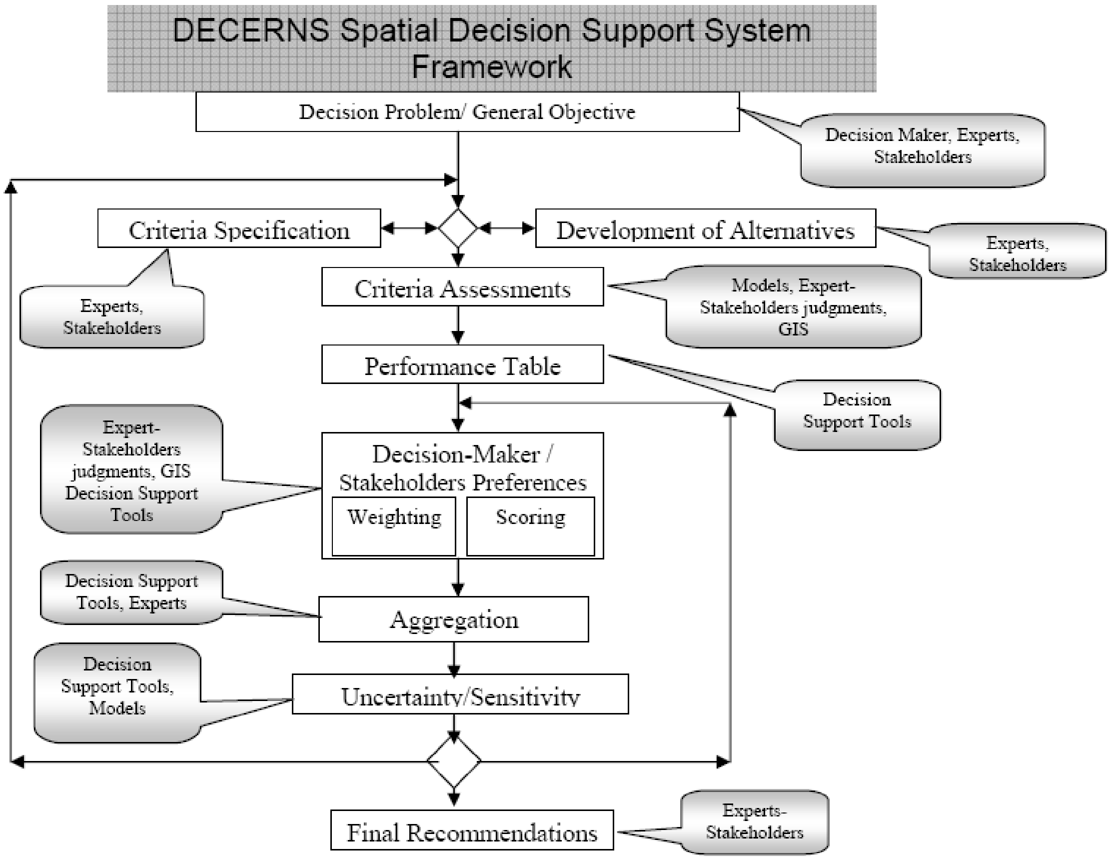
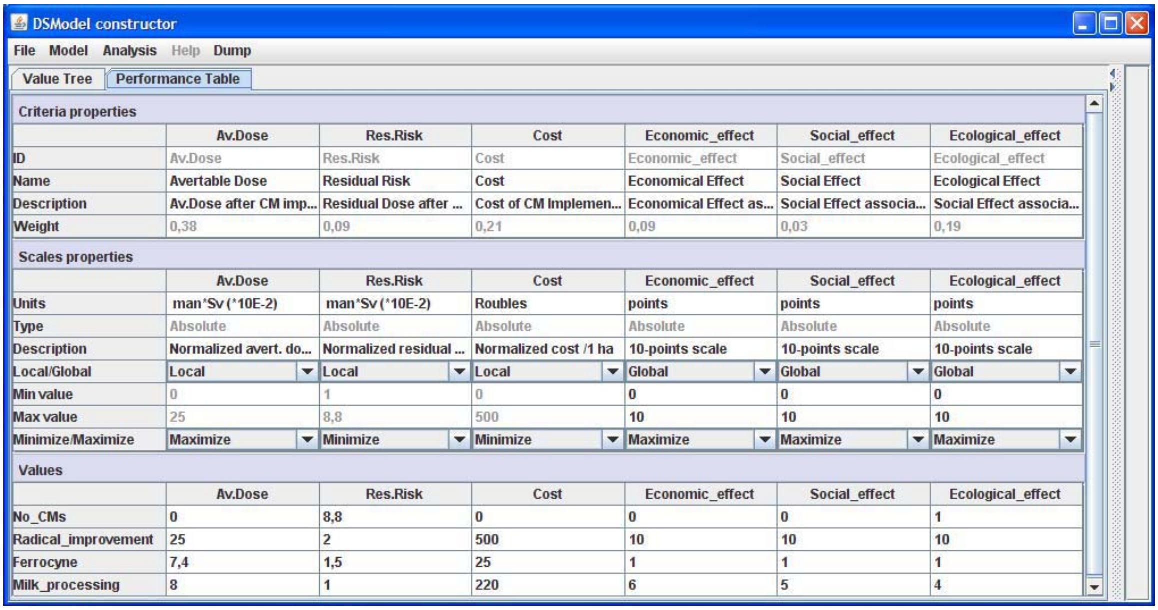


Figure 2.3 - DECERNS decision framework [4].

Quantitative analysis in the DECERNS can be performed using many different types of multi attribute decision analysis methods. In the example shown in Figure 2.4, Multi Attribute Value Theory (MAVT) is used to determine the optimum method for cleaning up ^{137}Cs contamination from the Chernobyl nuclear accident in a rural part of Russia. In MAVT analysis, the alternatives are scored for each nonmonetary criterion on an arbitrary scale, and then converted to monetary values with a value function. In this example, the economic effect, social effect, and ecological effect are all modeled via scoring and value functions. The scoring and value functions used in the calculations were to be chosen by the stakeholders.



Criteria properties

	Av.Dose	Res.Risk	Cost	Economic_effect	Social_effect	Ecological_effect
ID	Av.Dose	Res.Risk	Cost	Economic_effect	Social_effect	Ecological_effect
Name	Avertable Dose	Residual Risk	Cost	Economical Effect	Social Effect	Ecological Effect
Description	Av.Dose after CM imp...	Residual Dose after ...	Cost of CM Implemen...	Economical Effect as...	Social Effect associa...	Social Effect associa...
Weight	0,38	0,09	0,21	0,09	0,03	0,19

Scales properties

	Av.Dose	Res.Risk	Cost	Economic_effect	Social_effect	Ecological_effect
Units	man*Sv (*10E-2)	man*Sv (*10E-2)	Roubles	points	points	points
Type	Absolute	Absolute	Absolute	Absolute	Absolute	Absolute
Description	Normalized avert. do...	Normalized residual ...	Normalized cost /1 ha	10-points scale	10-points scale	10-points scale
Local/Global	Local	Local	Local	Global	Global	Global
Min value	0	1	0	0	0	0
Max value	25	8,8	500	10	10	10
Minimize/Maximize	Maximize	Minimize	Minimize	Maximize	Maximize	Maximize

Values

	Av.Dose	Res.Risk	Cost	Economic_effect	Social_effect	Ecological_effect
No_CMs	0	8,8	0	0	0	1
Radical_improvement	25	2	500	10	10	10
Ferrocyne	7,4	1,5	25	1	1	1
Milk_processing	8	1	220	6	5	4

Figure 2.4 - Example input for a cleanup decision in DECERNS using multi-attribute value theory [4].

DECERNS is a very complete decision making tool that could be used to perform much of the decision analysis for this research. However, it does not include a method for quantifying the public's perception without direct public involvement, which is one of the main goals of this research.

2.3 Methods for Quantifying for Quantifying Risk Perception

2.3.1 Risk Aversion Factor

The first method of accounting for the public's perception of risk in decision making was found in reference [5]. Bohnenblust and Slovic define the monetary collective risk, R_m, from n scenarios/events as (Equation 2.1):

$$R_m = \sum_{i=1}^{n} p_i \cdot C_i \cdot \varphi(C_i) \cdot \omega_i \tag{2.1}$$

where

p_i = The probability of scenario *i* occurring per year;

C_i = The consequences (fatalities or equivalent) if scenario *i* occurs;

$\varphi(C_i)$ = The risk aversion factor for the consequences C_i;

ω_i = The marginal cost/willingness-to-pay of society to avoid a fatality due to scenario *i* occurring.

In this definition the risk aversion factor is typically greater than 1, and increases with consequences to account for societies' increased fear of larger disasters. The Willingness-To-Pay (WTP) is inversely related to how voluntary the risk is and the perceived benefit of the risk to those affected. So risk associated with an involuntary exposure to waste generated decades ago would have a larger WTP than choosing to work in a dangerous industry. [5]

2.3.2 Sandquist's Method

The second was proposed by Sandquist and is conceptually very similar, though more factors are directly considered to determine the risk multiplier used to calculate the perceived risk. The equation given for the calculation of the total perceived risk, R_p, from an event is: [6]

$$R_p = F \cdot C \cdot P_t$$

where

$$P_t = \prod_{i=1}^{n} P_i$$

and

F = The frequency of the event;

C = The actual risk from the event;

P_i = The perceived risk associated with the i^{th} risk factor.

The P_i values can be any number greater than 0 to allow risks to be over and under estimated by the public. Sandquist recommends and defines 6 risk factors for consideration when calculating P_t while acknowledging this is not a complete list (Table 1).

Table 2.2 - Sandquist's list of factors that contribute significantly to perceived risk [6].

Label	*Risk Factor*	*Description of Risk Factor*
P_c	Control	Risk is controllable or uncontrollable by the individual
P_b	Benefit	Exposure to risk provides benefit or disbenefit to the individual
P_v	Volition	Exposure to risk is voluntary or involuntary for the individual
P_s	Severity	Risk ranges from ordinary or familiar to catastrophic or unknown
P_m	Manifestation	Risk results in immediate or delayed effects to the individual
P_o	Origin	Risk is a consequence of natural or man made events

Both of the methods discussed above recommend a simple calculation to determine the perceived risk for a particular real risk, which consists of multiplying the actual risk by factors that represent the ratio of the perceived risk to the actual risk. The difficulty in the calculation is in determining appropriate values to use for the factors. Because creating a new equation for calculating the perceived risk would require vast amounts of data gathering to determine appropriate values for any new parameters, Equation 1 will be used to determine the baseline perceived risk in this research. However, a modified version will be developed to account for the change in perceived risk for each of the cleanup alternatives.

Chapter 3 Decision Framework

3.1 Decision Making Criteria

The basic framework used in this research is presented in Table 3.1. This framework is similar to those found in other research especially the KONVERGENCE and DECERNS frameworks examined above. Multi-attribute value theory was chosen as the comparison tool for selecting the optimum solution. This is shown by the decision criteria (Equation 3.1) cited in steps 4 and 5 which is to find the cleanup alternative *I* that minimizes the "cost" of the contamination. The total cost associated with a cleanup alternative *i* is the sum of the absolute value of the negative factors that can be associated with the existence of the contamination and cleanup costs plus the benefit of performing cleanup alternative *i* (Equation 3.2).

Table 3.1 - Decision framework.

Step	Processes
1	Identify and characterize the problem (hazard(s) present and in what amount) and the population at risk
2	Identify applicable cleanup technologies/methodologies
3	Determine risk (dose) to exposed population from hazard(s)
4	Narrow list of possible solutions based on the decision criteria (Equation 1) by eliminating choices that are clearly not economical also possibly create hybrid solutions utilizing the good aspects of various solutions or proposing the use of different solutions for different parts of the problem
5	Choose the optimum solution based on the decision criteria (Equation 1)

$$I = min_i(Cost_i) \quad (3.1)$$

$$\mathrm{Cost_i} = \mathrm{PW}\{\mathrm{D(t)} + \mathrm{PER(t)} + \mathrm{O(t)} + [\mathrm{CC(t)} - \Delta\mathrm{D(t)} - \Delta\mathrm{PER(t)} - \mathrm{CGL(t)}]_i\} \quad (3.2)$$

where

$PW\{X(t)\}$ = Present Worth of the time dependent quantity X(t) (see below for complete definition);

D(t) = The value of the dose as a function of time;

PER(t) = The value of the publics perception of the dose or contamination;

O(t) = Other costs associated with the contamination in general (i.e. characterization, risk analysis, public input sessions, etc);

$CC_i(t)$ = The time dependent cost of the cleanup alternative *i*;

$\Delta D_i(t)$ = The time dependent dose reduction expected from the completion of the cleanup alternative *i*;

$\Delta PER_i(t)$ = The time dependent change expected in the public's perception due to the completion of cleanup alternative *i*;

$CGL(t)$ = Time dependent value of any other cleanup related gains or losses resulting from the completion of the cleanup alternative *i* (i.e. land value after cleanup, road closings due to cleanup, etc).

3.2 Present Worth

In Equation 2.3 all of the values are expressed as the Present Worth (PW) of something. The PW of a discrete time dependent loss or gain X(t) is defined as the value of those losses or gains at the present time and is calculated using Equation 2.3. For continuous losses or gains an integral is used instead of a summation and n is replaced with t. [7]

$$PW\{X(t)\} = \sum_{n=0}^{N} \frac{X(t)}{(1+r)^n} \qquad (2.3)$$

where

r = The discount rate net of inflation per time period where the time period length is the unit of *t*;

n = The number of time periods before or after the present time at which a cost or benefit occurs;

N = The time period in which the last gain or loss occurs.

A positive discount rate means return on investment is expected to be greater than inflation (future harms worth less than present harms), a zero discount rate means that returns are expected to equal inflation (future harms and present harms have same weight), and a negative discount rate means that returns are expected to be less than inflation (future harms weighted more than present harms).

The discount rate used when analyzing intergenerational harms especially when radioactive waste is involved is the subject of debate. Some think it should be zero for radioactive waste decisions because they consider it unethical to value the lives of people in the future less than harm being done to people now. They also point to analyses involving very long time horizons in which huge catastrophes in the distant future have small PWs and consequently are not a major consideration in decision making. [8] For example, if an decision results in a .1 % chance of $1.6*10^{51}$ $ (current dollars the actual amount would be much greater due to inflation) of damage being done in 10000 years the PW of that decision using a 1 % discount rate is only 100000 $. This may seem absurd but the fact is that if 1 $ were invested today and consistently earned 1 % more than inflation for 10000 years it would be worth $1.6*10^{43}$ more than it is now. Some of those opposed to discounting acknowledge this but say that the problem is that the necessary money to cover future events is never actually set aside and consequently there is no compounding and if/when the event occurs there will be inadequate resources to fix it. This is a legitimate concern but does not suggest that discounting should not be used; rather it is a concern with how policy makers handle spending, and suggests that more should be done to set aside money for expected future expenditures. The other side of the argument is that radioactive waste decision making is, in its essence, an economic decision, and that ignoring the time value of money in an economic decision makes no sense [9]. The real problem with discounting over long time horizons is choosing a discount rate that is representative of the entire period. For this study the results will be presented as a function of the discount rate to avoid further entanglement in the previous argument.

3.3 Dose and Perceived Risk

In this work, "dose" refers to the effective dose equivalent which is a measure of the energy imparted to a person by radiation, weighted by factors representing the type of radiation and the part of the body the energy is absorbed in, as shown in Equation 3.4 from [10].

$$E(t) = \sum_T w_T \sum_R w_R D_{T,R}(t) \tag{3.4}$$

where

$E(t)$ = The effective dose equivalent as a function of time;

w_T = The tissue weighting factor for tissue T;

w_R = The radiation weighting factor for radiation type R;

$D_{T,R}(t)$ = The absorbed dose (energy/mass) in tissue T from radiation type R as a function of time.

Many ways exist to estimate the dose to a group of people from a source of radioactive material including simple analytical models and computer codes utilizing more complex methods. Later in this work two computer codes (RESRAD Offsite and DUST-MS) will be discussed and used as well as an analytical model.

To calculate the PW of E(t) it must first be converted into risk and then to cost. For low dose rates the conversion to risk is done by multiply the dose by the total detriment probability per unit dose (most doses to the public from radioactive waste are low because they are regulated). The total detriment probability is the probability of fatal cancer development plus the probable detriment from nonfatal cancers and severe genetic effects. The total detriment probability per unit dose has been determined through epidemiological studies to be $7.3 * 10^{-7}$ per mrem [10]. The conversion to cost is done by multiplying the risk by the Value of a Statistical Life (VSL).

The time dependent perceived risk is calculated by substituting the time dependent risk for C_i in Equation 2.1 and subtracting 1 from the risk aversion factor to correct for the fact that the value of the actual risk is accounted for in D(t). The risk aversion factor used in this research was obtained from the National Radiation Protection Board (NRPB) of the UK. A plot of the risk aversion factor as a function of annual dose is provided below in Figure 3.1. The marginal cost/willingness to pay is the VSL quoted above.

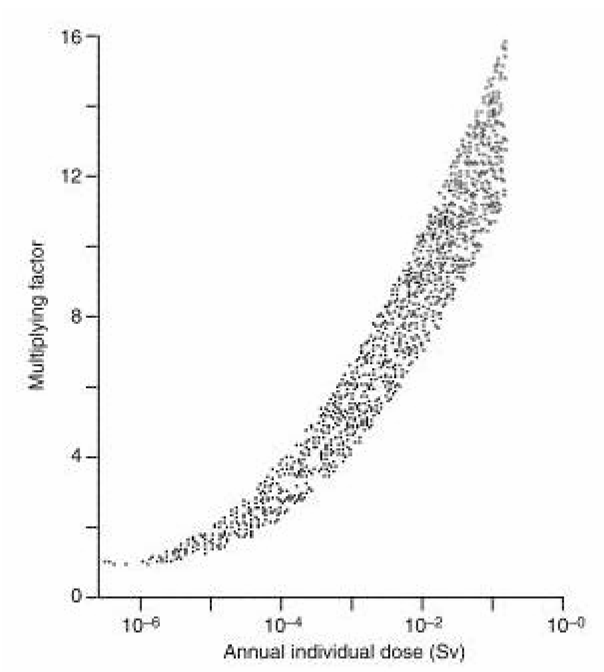


Figure 3.1 - NRPB's risk aversion factor for annual individual dose [11].

As can be seen in Figure 3.1, according to NRPB, there is some uncertainty in the value of the risk aversion factor. The average of the distribution is given in Figure 3.2 along with a functional fit of the data from a value of 1 at a dose of 1 mrem/yr to about 3 at a dose of 100 mrem/yr. The actual fit of the data is ($\varphi(E)$=.9869*$E^{.2365}$) but because this was less than 1 at 1 mrem/yr the fit was modified to ($\varphi(E)$=$E^{.2365}$) so it would always be greater than 1. For doses less than 1 mrem/yr $\varphi(E)$ is equal to 1. If doses higher than 100 mrem/ yr are to be considered a fit of the data up to the maximum dose considered would be required to obtain the appropriate $\varphi(E)$ for that region.

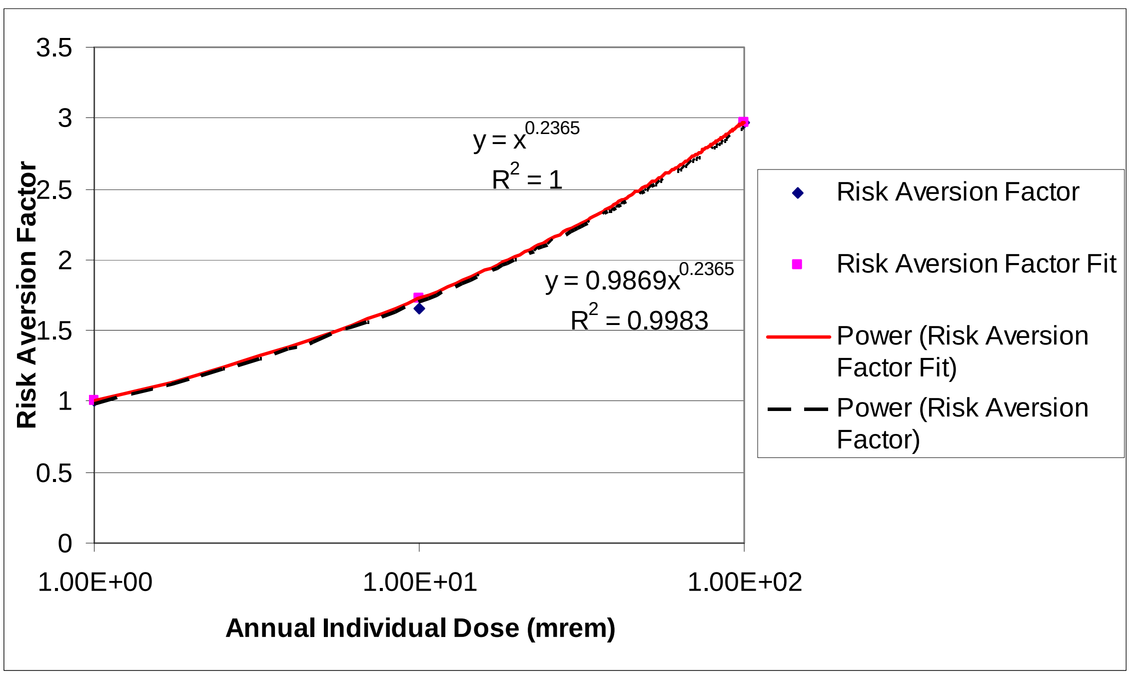


Figure 3.2 - Plot of the mean risk aversion factor obtained from Figure 3.1 as a function of annual individual dose with a functional fit.

3.4 Dose and Perceived Risk

The “other” costs are typically not very important to decision making because they have either already been incurred or never will be and consequently their inclusion will not affect the cleanup alternative selection process. However, they are important in determining the total cost of the contamination, and should be considered by decision makers in future decisions regarding the storage and disposal of hazardous materials.

3.5 Cleanup Alternative Specific Costs

For step 3 of the decision framework the cost of each cleanup alternative will be estimated using rough unit cost data from the literature. This will not result in the greatest accuracy, but should provide some idea of what the relative cost of each of the cleanup alternatives will be. If it was not clear which alternative is the best after this scoping calculation; more accurate cost estimates will be necessary in step 4 of the decision framework.

3.5.1 Change in Dose from Cleanup

The change in dose that results from the completion of each cleanup alternative will be calculated by determining the time dependent dose that will occur if the cleanup alternative is performed, and adding any worker dose due to the cleanup effort. The population dose calculation will be carried out using the same methods used for the initial dose estimates; only the source term and altered parameters will be changed to reflect the cleanup alternatives effectiveness. An estimate for the worker dose, if expected to be significant, could be obtained through similar methods to the population dose. The change in dose is then calculated by subtracting the initial dose estimate from the cleanup alternative specific estimate. This should result in a negative value for all feasible alternatives (except the no action alternative) which will reduce the total cost of the cleanup. The conversion to value is made the same way as for the original dose.

3.5.2 Change in Perceived Risk from Cleanup

The change in perception is more complicated, because perception is not easy to measure or calculate to begin with, so determining how certain events will affect it is even more difficult. This will be done using Equation 1 with a modified risk aversion factor (Equation 3.5). For this research a method for calculating a risk multiplier (φ^*) from the original risk aversion factor based on the decision outcome, the amount of public involvement in the decision, and the risk reduction expected from the remediation is developed. This method is given below in Equation 3.6.

$$R_m = \sum_{i=1}^{n} p_i \cdot C_i \cdot \varphi_i^* \cdot \omega \qquad (3.5)$$

where

$$\varphi^* = \frac{a \cdot (1 - rr) \cdot \varphi}{p} \qquad (3.6)$$

and

$$a = \text{action factor} = \begin{cases} 1 & \text{if action is taken} \\ 2 & \text{if action is not taken} \end{cases};$$

rr = fractional risk reduction achieved by the cleanup;

$$p = \text{public involvement factor} = \begin{cases} 1 & \text{no public involvement} \\ 2 & \text{some public involvement} \\ 3 & \text{full public involement} \end{cases}.$$

The reasoning behind this risk multiplier is based on Sandquist's list of factors that affect the public's risk perception (Table 1 above). The action factor and public involvement attempt to capture the effects of the public's control over the risk and ability to avoid it. The risk reduction factor accounts for the severity of the risk, and is used instead of just recalculating the risk aversion factor for the new risk to account for the fact that once a risk exists, getting people to forget about is much harder than reducing the risk in most cases.

3.5.3 Change in Other Costs from Cleanup

The other cleanup related gains and losses (CGL) can have a large impact on the decision or very little impact depending on the site. The CGL includes factors such as the value of the land after remediation versus the land's initial value and charges for events such as road closures or other public inconveniences. For instance, land values in cities are typically much higher than rural land values, so a completely cleaned reusable site would generally be worth much more in a city.

Chapter 4 Application of Decision Framework to the Kurchatov Institute's Radioactive Waste Disposal Site

4.1 Brief History of the Institute

The Russian Research Center: Kurchatov Institute (KI) was founded in 1943 as part of the former Soviet Union's nuclear weapons program. It is located about 12 km NNW of the center of Moscow, Russia in what is now a densely populated area. There are several operating reactors at the institute and many more that have been shut down. The waste generated during the operation of these reactors was, at least in part, stored at a designated Radioactive Waste Disposal Site (RWDS) within the institute's border. Between 1943 and 1973 this waste was placed in "repositories" which varied from a backfilled trench to concrete monoliths. These repositories have either partially failed or waste was spilled while being placed into or removed from them, because the soil at the RWDS is (as of 2003 when the last available data was taken) heavily contaminated with ^{137}Cs and ^{90}Sr as well as traces of other radioactive isotopes. [12]

Because most of the available data is from 2003 this is the time at which the case study will be based. Also the RWDS was remediated by excavation with offsite disposal beginning in late 2001; remediation is almost complete according to published studies [13].

4.2 Current (2003) Contamination Information

Maps of the ^{137}Cs and ^{90}Sr soil concentrations are provided below in Figures 4.1 and 4.2, respectively. Inspection of these maps reveals that the profiles are exactly the same but that the ^{137}Cs activity concentration is always about 5 times greater than the ^{90}Sr activity concentration. This is because the ^{137}Cs concentration was determined from exposure rates observed during a complete mapping of the site with a portable radiometer (Figure 4.3), while the ^{90}Sr concentration was determined by taking seven soil samples from various locations at the site, measuring the ^{137}Cs to ^{90}Sr ratio in these samples, and assuming the average was representative of the entire site. [12] The six boxes drawn on top of the exposure map in Figure 4.3 are the six different regions used later in the dose calculation. These were selected because of the relative homogeneity of the contamination within them, and to better approximate the shape of the contamination. The average exposure rate and the concentrations for each radionuclide in each of the regions are presented below in Table 4.1.

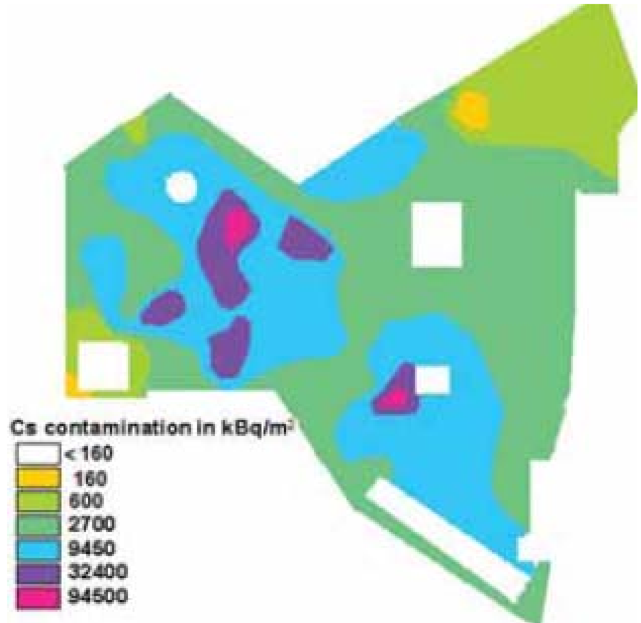


Figure 4.1 - A map of the ^{137}Cs concentration in the soil at the Kurchatov Institute radioactive waste disposal site [12].

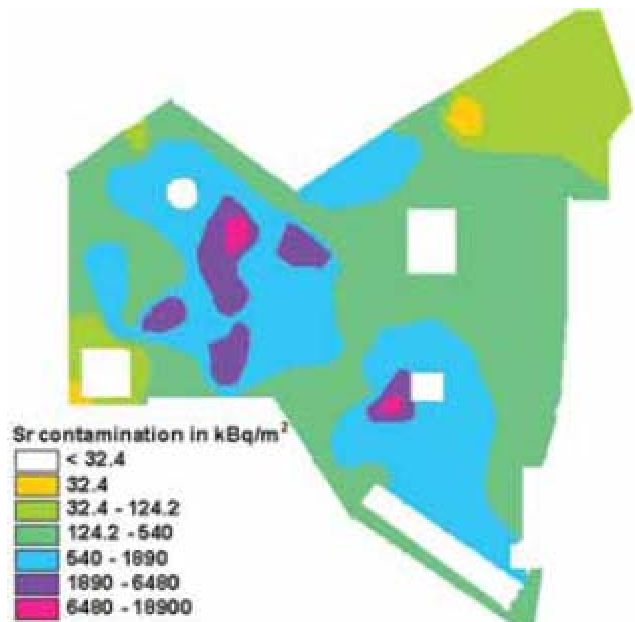


Figure 4.2 - A map of the ^{90}Sr concentration in the soil at the Kurchatov Institute radioactive waste disposal site [12].

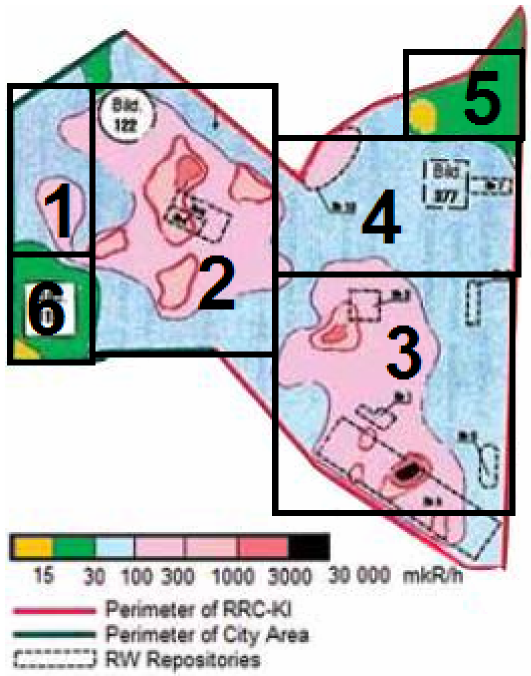


Figure 4.3 - Map of the exposure rate measured 1 m from the soil surface at the Kurchatov Institute radioactive waste disposal site [12].

The concentration of ^{137}Cs and ^{90}Sr were also determined as a function of depth (Figure 4.4), though the source does not reveal if this profile is representative of the site or just a measurement at a specific location. [12]

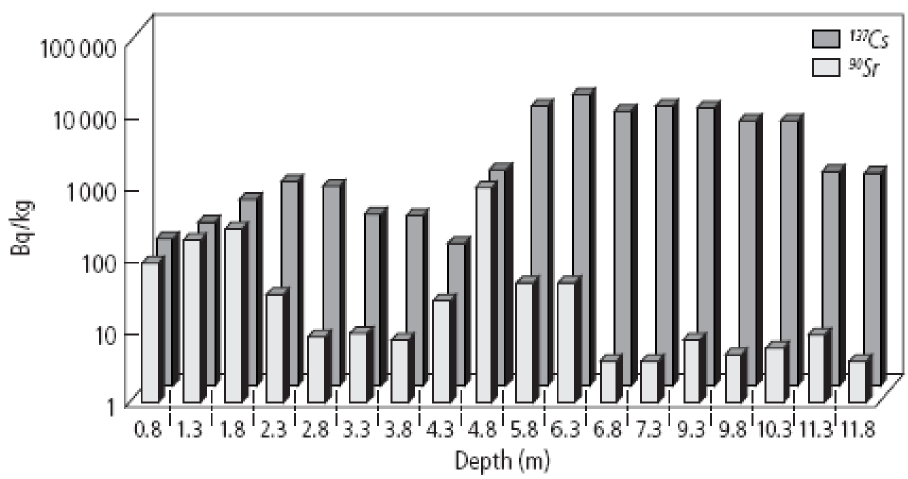


Figure 4.4 - The ^{137}Cs and ^{90}Sr concentration in the soil as a function of depth at the Kurchatov Institute radioactive waste disposal site [12].

In addition to the soil contamination, the groundwater at the site is also contaminated with ^{90}Sr [13] and probably ^{137}Cs and other nuclides, but no data is given on any other nuclides. ^{137}Cs is not a threat via the groundwater pathway, because its hydrological properties limit its mobility in the aquifer; however other more mobile nuclides might pose a threat if present. Figure 4.5 shows the distribution of ^{90}Sr in the groundwater at the site as of 2003.

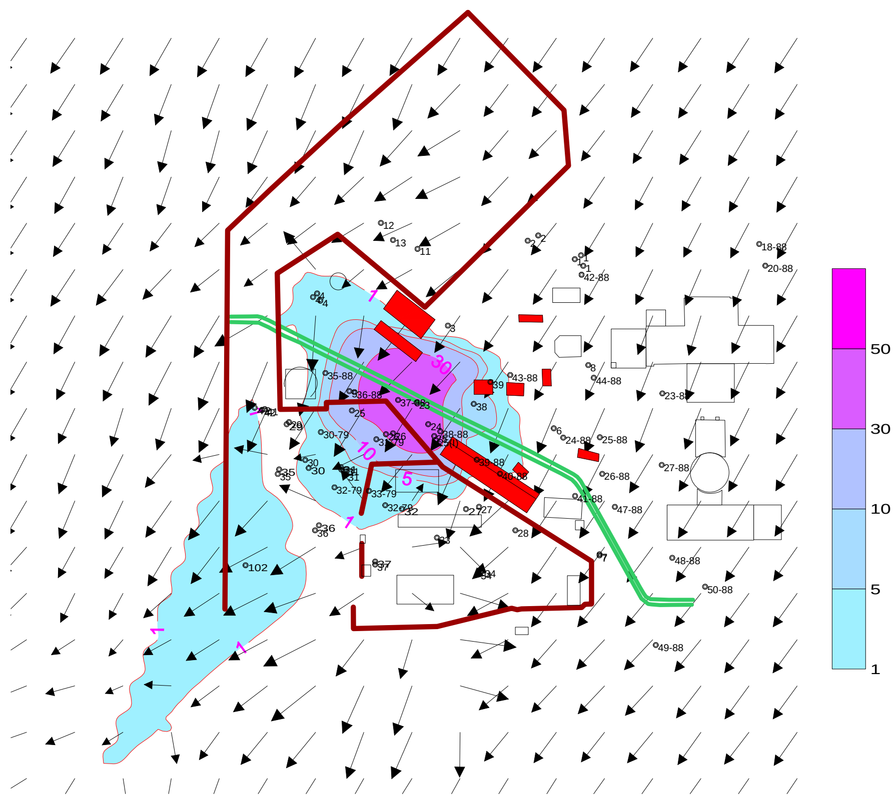


Figure 4.5 - A map of the ^{90}Sr concentration in the unconfined aquifer at the Kurchatov Institute radioactive waste disposal site in 2003 [13].

4.3 Uncertainties in the Current Contamination Information

There is a lot of uncertainty in the previously mentioned soil concentration data. The exposure rate measurement has an accuracy of ±20%. The conversion factor, a, used to convert the measured exposure rates into ^{137}Cs soil concentrations is 135 (Bq/kg)/ (μR/h) and has an accuracy of ±60% (it is assumed in this conversion that all of the exposure is caused by ^{137}Cs). [12] Both of these errors are assumed to be standard deviations. Equation 4.1 was used to estimate the standard deviation for the concentration of ^{137}Cs in pCi/g and the results are presented in Table 4.1.

$$\sigma_{\bar{f}} = \sqrt{\sum_{i=1}^{M}\left(\frac{\partial f}{\partial \bar{x}_i}\right)^2 \sigma_i^2} \qquad (4.1)$$

The average ^{137}Cs concentration is calculated with the following equation:

$$[Cs] = C\dot{E}a$$

where,

C = a conversion factor = .027 (pCi/g)/(Bq/kg).

The results of the measurements used to determine the ratio between the ^{137}Cs and ^{90}Sr concentrations are not provided; however the result inferred from the above figures is about 5. The errors for the measurement of the concentrations of each of these nuclides in a soil sample were assumed to be the same as errors cited for similar measurements performed during the determination of the K_d value for the radionuclides.

These errors are ±20% for ^{137}Cs and ±30% for ^{90}Sr [12] and were also assumed to be standard deviations. Equation 4.1 was used to determine the standard deviation for the ^{90}Sr concentration in pCi/g from the following equation for the average ^{90}Sr concentration in the soil (Table 4.1).

$$[Sr] = C\dot{E}a\left(\frac{[Sr]}{[Cs]}\right)_{avg}$$

There was also no data provided on the accuracy of the ^{90}Sr concentration in the groundwater so it was assumed to have an error of ±30% like the ^{90}Sr in the soil.

Table 4.1 - The average exposure rate and the mean and standard deviation of the concentration for ^{137}Cs and ^{90}Sr in each of the 6 regions shown in Figure 4.3.

Contamination Region	Mean Exposure Rate (μR/h)	Average ^{137}Cs Concentration (pCi/g)	Average ^{90}Sr Concentration (pCi/g)	Standard Deviation of ^{137}Cs Concentration (pCi/g)	Standard Deviation of ^{90}Sr Concentration (pCi/g)
1	166	605	121	383	88
2	354	1291	258	817	188
3	474	1727	345	1092	251
4	121	441	88	279	64
5	28	103	21	65	15
6	22	80	16	51	12

4.4 Exposure Scenarios Considered

The following exposure pathways were considered in the analysis of the dose to the public from the RWDS: external gamma, inhalation, soil ingestion, surface water (external gamma), and groundwater consumption. The first three scenarios are straight forward. External gamma exposure occurs when a radionuclide decays and releases a gamma ray or a gamma ray is created by the interaction of other particles outside of a person's body and the resulting gamma ray strikes them. The entire radionuclide inventory (onsite, suspended, and deposited offsite) is a possible source of external gamma exposure. Inhalation exposure occurs when a person breathes in air containing suspended radionuclides. Only the currently suspended (airborne) fraction of the inventory is a possible source for inhalation exposure. Soil ingestion exposure occurs when someone eats soil (typically inadvertently) that contains radionuclides. All contaminated soil is a possible source of soil ingestion exposure, though in many cases, including the current case, public access of the site is not allowed, so only the deposited (offsite) fraction of the inventory is available for ingestion exposure.

The surface water external gamma pathway results in exposure through the same process described above for external gamma. The differences are in how the radionuclides get to a point where they can affect that exposure and in the duration of the exposure. The surface water considered in this study is puddles that could potentially contain contaminated water/sediment in a nearby public parking lot because of runoff from the RWDS [12]. The consumption of groundwater contaminated with radionuclides is a possible exposure pathway if drinking water is obtained from wells that draw water from a contaminated aquifer. No data is available on current well use, except that the institute utilizes artesian wells within its boundaries to obtain water for its work [12]. Consequently, it will be necessary to predict the likelihood of future generations using water from the contaminated aquifers to get an estimate of the expected dose from this pathway.

The residents living within 1 km of the site were assumed to be the only affected humans for this study because preliminary data showed doses beyond 1 km are very low. A plot of the exposed population's distribution with respect to the site was created from data in [12] and is provided below as Figure 4.6. The exact center of the population distribution data from which this figure was created was not provided in the reference, but by fitting the data to a map of the area the center was found to be near the center of the RWDS which makes sense because the RWDS was the subject of the paper from which it was obtained.

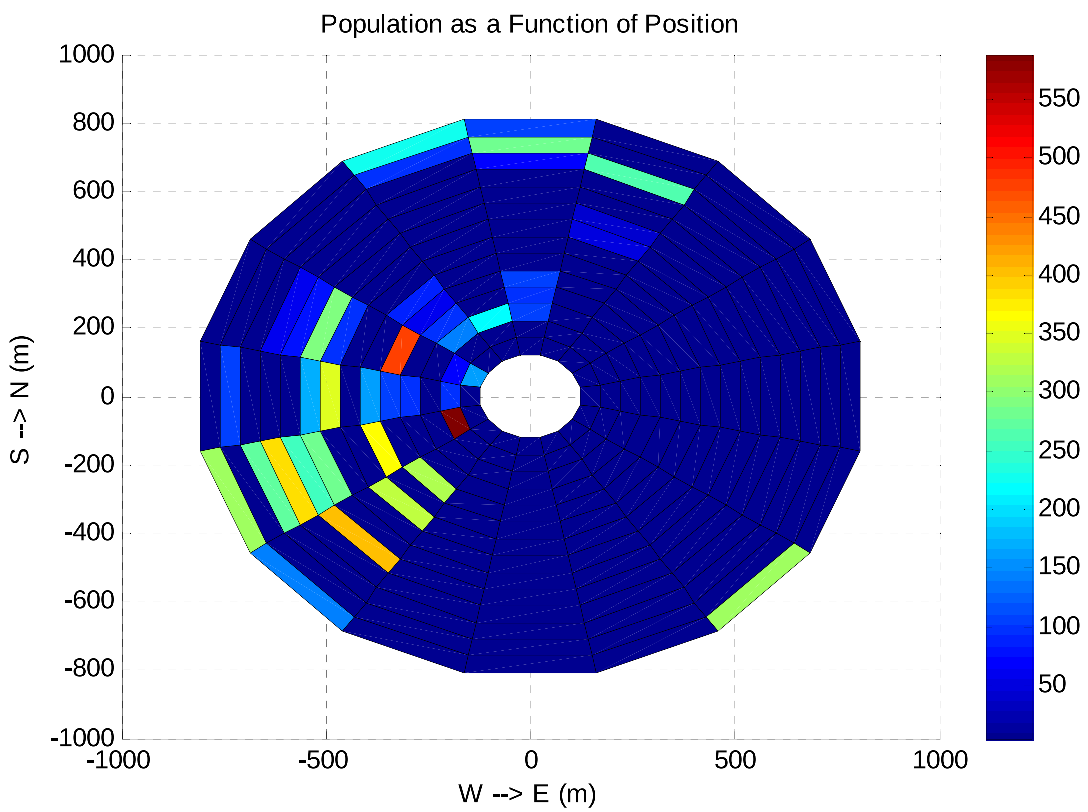


Figure 4.6 - Population distribution near the Kurchatov Institute's radioactive waste disposal site, base on data from [12].

4.5 Cleanup Alternatives Considered

In the previous sections, step 1 of the decision framework was completed by identifying the hazards and at risk population. In this section potential cleanup alternatives for this contamination will be described, completing step 2 of the framework.

The cleanup alternatives evaluated in this research for use at the KIRWDS are: no remediation, cover contamination with clean fill dirt (hereafter clean fill), grout/cement the soil at the site and cover with clean fill, In-Situ Vitrification (ISV) of the site and cover with clean fill, and excavation and removal of the contamination with offsite disposal and backfilling the site with clean fill. No remediation and covering the site with clean fill are pretty self explanatory. The depth of the clean fill for all of the alternatives involving it will be 1 m. Covering the site with clean fill will shield against gamma radiation and limit atmospheric contact with the radionuclides thus virtually eliminating the direct, inhalation, and soil ingestion pathways.

Grouting/cementing the soil is done by injecting cement into the soil while it is being mixed with an auger. This process results in a very stable mass which if done right greatly reduces access of water and air to the soil contaminates. When the monolith is covered with soil to protect it and also shield the gamma radiation from it there is very little dose to the public via the atmosphere. The groundwater should also be reduced because of the leach resilience of the grouted soil.

In-situ vitrification is done by melting the soil with resistance heated graphite electrodes so that it forms a very strong glass. A hood is placed over the soil to collect for treatment any harmful gases released during the process. The effect of ISV on the contamination is similar to that of grouting only the ISV generated glass is much more resilient than the grouted soil.

Excavation of the contamination with offsite disposal should eliminate the hazard at the current site entirely once completed. The problem is that it just moves it to another location (though the new location is expected to be much safer than the current location). There is the potential for increased total dose during the remediation process because of the amount of dust generated during the cleanup and the fact that workers performing the cleanup will be receiving elevated doses.

4.6 Dose Calculation Inputs

4.6.1 External Gamma, Inhalation, and Soil Ingestion

The dose from the external gamma, inhalation, and soil ingestion pathways was computed using the RESRAD Offsite computer code developed at ANL. As stated above, the RWDS was divided into 6 regions for the purpose of dose modeling to allow for better spatial characterization of the site and to account for the non homogeneity of the contamination (see Figure 4.3 for the location of the regions). The dose from each of theses regions was calculated at 72 different locations outside of the site to accurately map the dose to the exposed population. A plot of these 72 locations with respect to the center of the RWDS is given below in Figure 4.7, along with the location of the population data points. Linear interpolation was used to estimate the dose at points where it was not calculated.

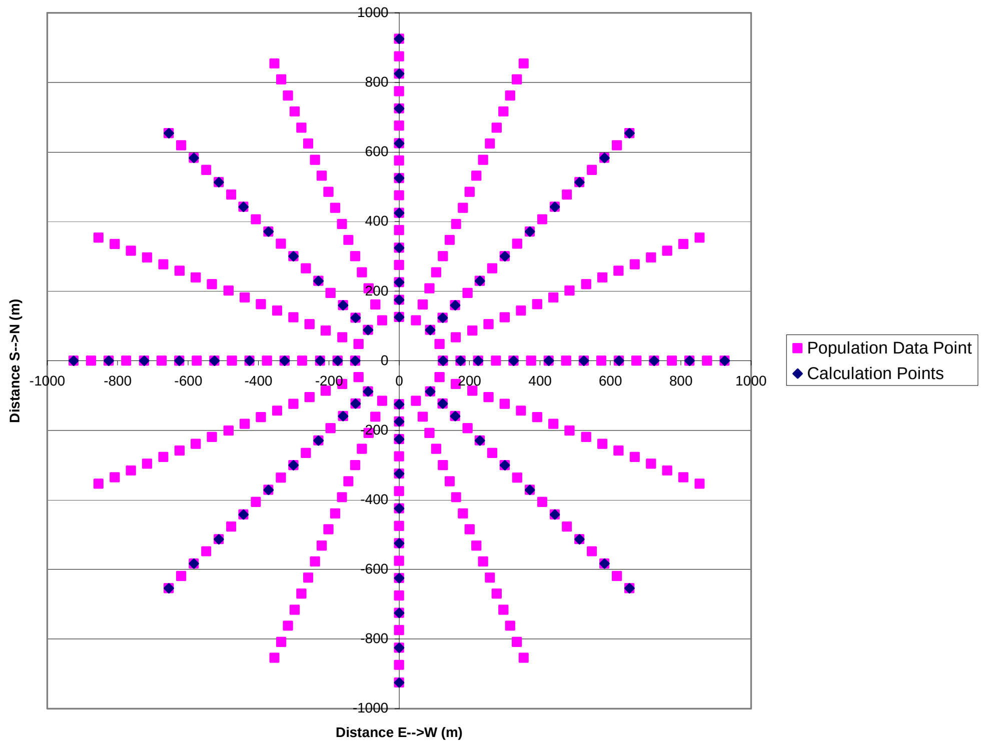


Figure 4.7 - Plot of the population data points and the calculation points used to characterize the dose distribution around the site.

The possible error in the source term identified previously was accounted for by running the RESRAD calculation for different source terms, based on the standard deviation of each contaminant's concentration and also a hypothetical situation. The hypothetical situation is that there is 10% chance that the ^{90}Sr concentration is roughly half the ^{137}Cs concentration instead of one fifth. This scenario is possible given the method used to determine the concentration of ^{90}Sr and the fact that ^{137}Cs and ^{90}Sr are typically produced in roughly the same quantities in a nuclear reactor. These different source terms and the probability of occurrence assigned to each are given in Table 4.2.

Table 4.2 - Source terms used in the RESRAD dose calculation.

	Amount to be added to each of the baseline concentrations	
Probability	Cs-137 (pCi/g)	Sr-90 (pCi/g)
.2777	$-\sigma$	$-\sigma$
.3446	0	0
.2777	$+\sigma$	$+\sigma$
.1	0	$+2.5\sigma$

It was also necessary to analyze the variation in dose as function of age. The age distribution of the affected population is given in Figure 4.8. Several important parameters can vary from one age group to another including: behavior, inhalation rate, and ingestion rate. The most important behaviors for this calculation are time spent indoors at home and time spent outdoors near the site. Data on the time the local population spends outdoors is provided in Table 4.3; because no data was available on the time spent indoors at home this parameter was considered constant with a value of 12 hours per day. Variations in the inhalation dose due to different dose conversion factors and inhalation rates for different age groups were not modeled, because the inhalation dose was negligible compared to the total dose. Ingestion of food grown on contaminated soil was not considered; consequently, variations in ingestion rates were not considered. These data were used to calculate the age dependent dose for the direct, inhalation, and soil ingestion pathways.

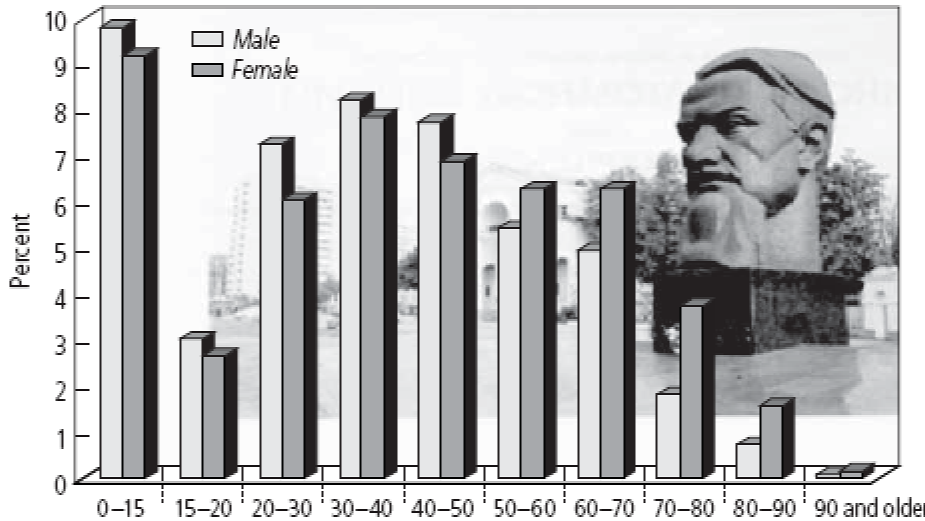


Figure 4.8 - Age distribution of population near the KI site [12].

Table 4.3 - Average time spent outside per day for various age groups [12].

Age (years)	Younger than 1	1–2	2–7	7–12	12–17	17–60	Over 60
Time spent outdoors (hours)	1.5	2	3	1	0.5	0.3	1.5

Because RESRAD is able to handle only one value for each input per run (including source and receptor location), it was necessary to write a code (inputcreator; see Appendix A) to generate the 5184 (6*72*4*3) input files. For the most part this code produced the same results as RESRAD Offsite's Graphical User Interface (GUI), because most of the inputs were not changed from one scenario to another. However, there were small differences (< 5%) in the calculation of shape factors (the area of a circular disc intersected with the region of interest) with inputcreator and the RESRAD Offsite GUI. It is not clear why these errors occur but the suspected reason is that

RESRAD calculates the values “graphically” while inputcreator uses integrals to find the areas.

Some of the more important inputs that were constant for all of the input files were the wind and stability class probability data, and the external gamma penetration factor. The stability class, wind speed, and wind direction probability data were obtained from the data in Figure 4.9 and Table 4.4 below.

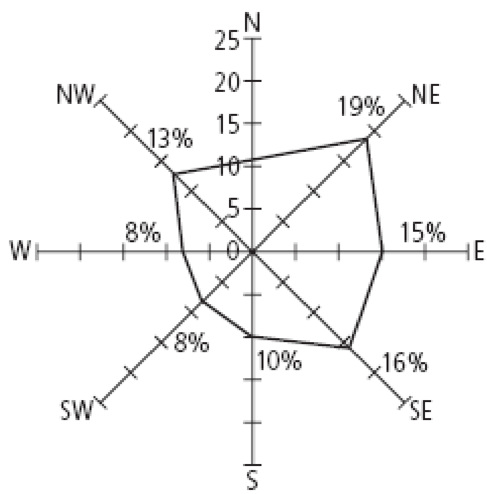


Figure 4.9 - Wind rose for Moscow [12].

Table 4.4 - Stability class and wind speed probability data [12].

	Categories of atmospheric stability*						
Parameter	A	B	C	D	E	F	G
Recurrence of weather conditions (%)	0.86	7.37	12.72	48.76	3.74	1.61	3.13
Average wind velocity (m/s)	1.2	2.35	2.58	2.54	1.54	1.2	0.91
Calm (%)	1.36	1.31	1.51	10.86	2.09	1.02	3.63

The external gamma penetration factor was changed from RESRAD's default value of .7 to a less conservative value of .5, which was determined to be conservative for modern buildings from reference [14]. The concentration of the contaminants was assumed to be constant to a depth of 5 m, which, based on Figure 4.4, is close to true for ^{137}Cs but not for ^{90}Sr, though, because groundwater dose is not considered in the RESRAD Offsite calculation, only the surface concentration is important, and the concentration for ^{90}Sr is roughly constant for the first 2 meters.

RESRAD Offsite default values were used for inputs not mentioned.

4.6.2 Groundwater

The groundwater dose calculations were performed using DUST-MS, a computer code developed by Terry Sullivan at BNL. DUST-MS is a very versatile code that is able to model releases from many types of waste, and perform contaminant transport of waste in the vadose zone and aquifer. Three soil conditions were used to model the groundwater pathway for the KI site: contaminated soil, grouted/cemented soil, and vitrified soil. The contaminated soil scenario represents the no remediation and clean cover alternatives. The other two scenarios represent their respective cleanup alternative.

The different site conditions were modeled using different input files for the vadose zone and the same input file transport in the aquifer. The contaminated soil was modeled as rinse release which releases material from the soil into the surrounding water until either equilibrium partitioning or the solubility limit is reached. The grouted/cemented soil was modeled as diffusion release which allows the contaminant to gradually diffuse out of the waste form subject to a diffusion coefficient. The vitrified soil was modeled as dissolution release which allows a certain fraction of the waste form and the associated contamination to dissolve each year and become available for transport to the aquifer.

The waste is assumed to be in the vadose zone (above the water table) for all three scenarios. For the contaminated soil scenario, the entire site was modeled (11360 m^2); the other scenarios were modeled as being one tenth this size to account for the possibility of gaps between the treatment regions and stress cracking of monoliths in the future (smaller sections yield higher releases due to increased surface area to volume ratio). The contaminated region was assumed to be 5 m thick with 1 m of clean cover on top of it. The source term for the groundwater calculations was assumed to be the soil contamination used in the RESRAD Offsite calculation, 74 Ci, plus 380 Ci that was removed from the repositories during the RWDS remediation [13]. Two scenarios were used for the groundwater source term: the first was a 90% probability the ^{137}Cs to ^{90}Sr ratio is 5 to 1 (all nuclide ratios are activity based) and a 10% probability the ^{137}Cs to ^{90}Sr ratio is 2 to 1. The second scenario was a 100 % probability of ^{99}Tc, ^{129}I, and ^{90}Sr being in equilibrium with the average ^{137}Cs concentration according to the thermal fission yield of ^{235}U for each nuclide (note: there is no concentration data available for nuclides other than ^{137}Cs and ^{90}Sr so this scenario is hypothetical). The ^{235}U thermal fission yield and halflife (used to calculated the decay adjusted concentration as of 2003) of each of these nuclides is given in Table 4.5. The ^{137}Cs concentration for this scenario was based on a 5 to 1 ^{137}Cs to ^{90}Sr ratio and a total activity of 454 Ci. The fission yields and half lives used to derive the concentrations for this scenario are given in Table 4.5. For this scenario it was assumed that all of the waste was created 50 years prior to 2003. The source terms for both of these scenarios are presented below in Table 4.6. Because ^{137}Cs moves slowly in groundwater systems due to its high K_d value (~70) it was not modeled in the groundwater dose calculations [12].

Table 4.5 – Thermal ^{235}U fission yields and half lives for ^{137}Cs, ^{90}Sr, ^{99}Tc, and ^{129}I from KAERI [15].

Nuclide	Fission Yield (#/fission)	Half life (yrs)
^{137}Cs	.0627	30.07
^{90}Sr	.0590	28.79
^{99}Tc	.0611	$2.11*10^5$
^{129}I	.00718	$1.57*10^7$

Table 4.6 – Radionuclide concentrations for ground water contamination scenarios.

Scenario	Concentration (Ci/m^3)		
	^{90}Sr	^{99}Tc	^{129}I
1a (90 %)	.00133	0	0
1b (10 %)	.00266	0	0
2	.00622	$2.93*10^{-6}$	$4.63*10^{-9}$

The aquifer was modeled to about 400 m from the KI RWDS boundary. The initial ^{90}Sr concentration in the aquifer was taken from Figure 5.5. The direction of groundwater flow and the hydraulic gradient were determined from Figure 18. The exposure scenario for the groundwater pathway was a 1 % chance of a well being dug at the nearest possible point (for a private well this is across the street from the KI) to the site, and that all of the 593 people living in the nearest dwelling will get their drinking water from the well if it is dug [12]. The average individual dose from this scenario is calculated using the following equation:

$$D_{groundwater} = \sum_j C_{groundwater,j} \cdot \sum_i w_i \cdot CR_i \cdot DCF_{i,j}$$

where

w_i = The fraction of the exposed population belonging to age group *i*;

CR_i = The consumption rate of drinking water for age group *i*;

DCF_i = The Dose Conversion Factor (DCF) for age group *i* for consumption of nuclide *j*;

$C_{groundwater,j}$ = Concentration of nuclide *j* where the well is located.

The age dependent consumption rates and the ^{90}Sr, ^{99}Tc, and ^{129}I DCFs used for this calculation are provided in Table 4.7.

Table 4.7 - Annual age dependent consumption rate of water and the DCFs for the consumption of ^{90}Sr, ^{99}Tc, and ^{129}I.

Age Range (yrs)	Annual Consumption of Water (L/yr) [16]	^{90}Sr Ingestion DCF (Sv/Bq) [17]	^{99}Tc Ingestion DCF (Sv/Bq) [17]	^{129}I Ingestion DCF (Sv/Bq) [17]
0-1	260	$2.3*10^{-7}$	$1.0*10^{-8}$	$1.8*10^{-7}$
1-2	260	$7.3*10^{-8}$	$4.8*10^{-9}$	$2.2*10^{-7}$
2-7	260	$4.7*10^{-8}$	$2.3*10^{-9}$	$1.7*10^{-7}$
7-12	260	$6.0*10^{-8}$	$1.3*10^{-9}$	$1.9*10^{-7}$
12-17	260	$8.0*10^{-8}$	$8.2*10^{-10}$	$1.4*10^{-7}$
17+	370	$2.8*10^{-8}$	$6.4*10^{-10}$	$1.1*10^{-7}$

Some of the important parameters for the DUST-MS calculations are given in Tables 4.8 and 4.9. The effective diffusion coefficient of the contaminant in the hydrological system was assumed to be 10^{-5} cm/s which is a conservative value given in the DUST-MS manual for use when data is unavailable [18]. The solubility limit used for ^{129}I was $2.28*10^{-7}$ Ci/cm^3 (for the other nuclides solubility does not limit transport) [19]. These parameters are conservative and should yield concentrations at the receptor location greater than or equal the actual concentrations that would result from the contamination scenarios considered. Copies of the input files used in the DUST-MS calculations for scenario 2 are provided in Appendix B.

Table 4.8 - Important nuclide specific parameters for DUST-MS calculations.

Parameter	Units	^{90}Sr	^{99}Tc	^{129}I
K_d	cm^3/g	5.1 [20]	.1 [21]	1 [21]
Diffusion Coefficient in Cement	cm^2/s	$5.2*10^{-10}$ [19]	$8.0*10^{-12}$ [22]	$1.3*10^{-12}$ [22]

Table 4.9 - Important general parameters for DUST-MS calculations.

Parameter	Units	Value	Description	Source
Infiltration Rate	cm/s	$1.598*10^{-6}$	Rate at which water flows through the vadose zone to the aquifer	See Below
Longitudinal Dispersivity	cm	.1*L		[23]
Vitrified Soil Dissolution Rate	yr^{-1}	$5.312*10^{-6}$	Fraction of waste dissolved each year (site specific)	[24]
Hydraulic Conductivity	cm/s	.004167	High estimate	[12]
Hydraulic Gradient	cm/cm	.00263	Calculated from Figure 4.10 (2003 data)	[25]

The infiltration rate was calculated from the following equation recommended in the RESRAD Offsite manual [26]:

$$V_d = I = (1 - C_e) \cdot [(1 - C_r) \cdot P_r + I_{rr}]$$

where

I = Infiltration rate = Darcy velocity in vadose zone;

C_e = Evapotranspiration coefficient = .5 (RESRAD default) [26];

C_r = Runoff coefficient = .2 (RESRAD default) [26];

P_r = Precipitation rate = 59.9 cm /yr [27];

I_{rr} = Annual irrigation applied = 0 (Assumed because no reason for irrigation).

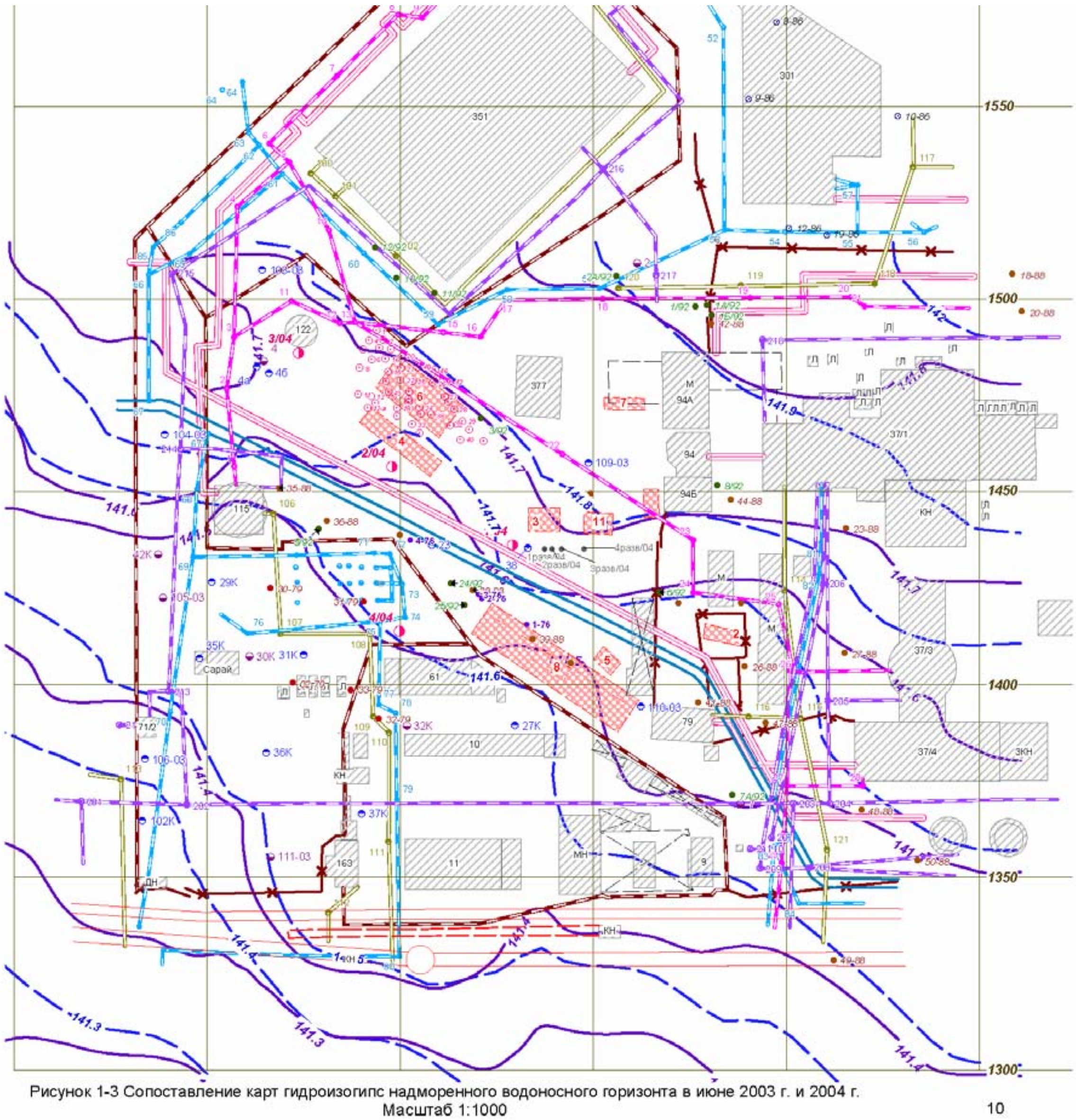


Figure 4.10 - An equipotential map of the KI RWDS for 2003 (blue dashed lines) and 2004 (solid purple lines) [25].

4.6.3 Surface Water

The surface water pathway analysis in this research is based on a study performed by IIASA in [12] to determine the dose due to runoff from the KI RWDS. In their work, they determined that contaminant redistribution resulting from a precipitation event of average rainfall (40mm/hr) would lead to no dose to population because such an event would not cause runoff out of the RWDS. They also analyzed the runoff from a precipitation event of historically maximum intensity (100 mm/hr). For this scenario they determined there would be some material redistributed to a municipal car park and street located adjacent to the RWDS if the soil was wet prior to the precipitation event. The probability of this occurring was determined to be .000041 per year based on a .000082 probability of rainfall ≥ 80 mm and an assumed 50 percent chance of the soil being wet prior to such an occurrence. A plot of the time dependent rainfall rates for both scenarios is provided in Figure 4.11. If this event occurs, the study estimates that it would result in ^{137}Cs and ^{90}Sr being deposited in the municipal car park and street in the amounts shown in Table 4.7 as source term 2. These concentrations were used to generate 3 additional source terms just as for the RESRAD Offsite calculation, which are also provided in Table 4.7. The exposed population was assumed to be 200 people for the car park and 1000 people for the street only (people in the car park are also assumed to be exposed on the street). People were assumed to spend 20 minutes and 5 minutes in the car park and street, respectively.

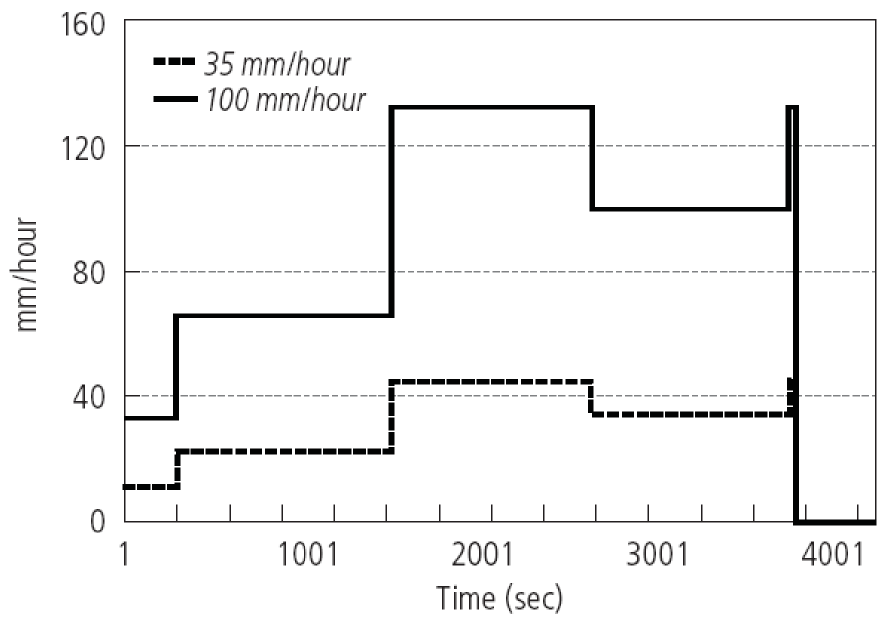


Figure 4.11 - Time dependent rainfall rates for scenarios in runoff analysis [12].

Table 4.10 - Surface activity in the municipal car park and street for 100 mm rain event.

Source Term	Probability	^{90}Sr surface activity in municipal car park (Bq/m^2)	^{137}Cs surface activity in municipal car park (Bq/m^2)	^{90}Sr surface activity in the street (Bq/m^2)	^{137}Cs surface activity in the street (Bq/m^2)
1	.2777	545	3712	101	3271
2	.3446	2010	10100	370	8900
3	.2777	3473	16488	639	14529
4	.1	5668	10100	1043	8900

The dose from groundshine (external gamma) from the ^{137}Cs surface concentration is calculated using the conversion factor $5.51*10^{-11}$ mrem/s per Bq/m^2 [12]. The average individual inhalation dose due to suspension of these radionuclides is calculated using the following equations recommended in [12]:

$$D_{inhalation} = \sum_j C_{air,j} \cdot \sum_i w_i \cdot IR_i \cdot DCF_{i,j}$$

$$C_{air}(Bq/m^3) = K_{susp,j}(t) \cdot C_{ground}(Bq/m^2)$$

$$K_{susp,j}(t) = 10^{-6} \cdot e^{-(\lambda_1+\lambda_j)\cdot t} + 2*10^{-8} \cdot e^{-(\lambda_j)\cdot t}$$

where

IR_i = The inhalation rate for age group i;

DCF_i = The dose conversion factor for age group i due to inhalation of nuclide j;

λ_1 = Initial decay constant for the decrease in resuspension = $1.26*10^{-2}$/day;

λ_j = Radioactive decay constant for nuclide j.

The age dependent inhalation rates and DCFs used in this calculation are provided in Table 4.8. The DCFs are for inhalation class D which means the radionuclides are expected to be cleared from the body relatively quickly after inhalation. Class D was selected based on data in [28] and also because the values for Class D are close to those used in [12].

Table 4.11 - Annual age dependent inhalation and DCFs for the inhalation of ^{90}Sr and ^{137}Cs.

Age Range (yrs)	Annual Inhalation (m^3/yr) [16]	^{90}Sr Inhalation DCF (Sv/Bq) [17]	^{137}Cs Inhalation DCF (Sv/Bq) [17]
0-1	3700	$1.3*10^{-7}$	$8.8*10^{-9}$
1-2	3700	$5.2*10^{-8}$	$5.4*10^{-9}$
2-7	3700	$3.1*10^{-8}$	$3.6*10^{-9}$
7-12	3700	$4.1*10^{-8}$	$3.7*10^{-9}$
12-17	8000	$5.3*10^{-8}$	$4.4*10^{-9}$
17+	8000	$2.4*10^{-8}$	$4.6*10^{-9}$

4.7 Cleanup Alternative Cost Calculation

The cost of cleaning up the KI RWDS with each of the cleanup alternatives will be estimated using generic unit cost data. Zero cost is assumed for the no remediation alternative. The cost for the clean cover alternative is given by the following equation:

$$C_{fill} = V_{fill} \cdot (UC_{fill} + UC_{spread})$$

where

V_{fill} = The volume of clean fill required = 1 m * 11360 m^2 = 11360 m^3;

UC_{fill} = The unit cost of quality fill ($\$/m^3$) = 16.75 with 10 mile haul [29];

UC_{spread} = The unit cost to properly spread the clean fill ($\$/m^3$) = 3.35 [29].

The cost of the grouting alternative was calculated using the following equation:

$$C_{grout} = V_{grout} \cdot UC_{grout} + C_{fill}$$

where

V_{grout} = The volume of soil to be grouted = 5 m * 11360 m^2 = 56800 m^3;

UC_{grout} = The unit cost of grouting soil = 238 $\$ / m^3$ for 1 m spacing [30].

The cost of the ISV alternative was calculated using the following equation:

$$C_{ISV} = V_{ISV} \cdot UC_{ISV} + C_{fill}$$

where

V_{ISV} = The volume of soil to be vitrified= 5 m * 11360 m^2 = 56800 m^3;

UC_{ISV} = The unit cost of vitrifying soil = 704 $\$ / m^3$ [24].

The cost of the excavation with offsite disposal alternative was calculated using the following equation:

$$C_{removal} = V_{removal} \cdot \left(UC_{removal} + UC_{fill} + UC_{spread}\right)$$

where

$V_{removal}$ = The volume of soil to be removed= 5 m * 11360 m^2 = 56800 m^3;

$UC_{removal}$ = The unit cost of removing, shipping, and offsite storage of waste = 918 \$ / m^3 [31].

The PW of each of the cost of each the cleanup alternatives was determined by estimating the length of time required to complete them, dividing the cost evenly over that time period, and using Equation 2.3 to determine the PW. The grout alternative is expected to take 1 year or less. The ISV alternative is estimated to take two take 2 yrs for the KI RWDS based a total waste mass of 113000 tons and three 3 units vitrifying 500 tons every 7.5 days with an 80% capacity factor [31]. The time required to complete the removal alternative was estimated from the length of time required to complete the actual remediation of the KI RWDS to be 4 years.

4.8 Dose to Workers during Cleanup

The dose to workers during completion of each of the cleanup alternatives considered was estimated from data on the dose workers received during the KI RWDS cleanup which are given by year in Table 4.9.

Table 4.12 - Average and cumulative dose to workers that participated in the actual cleanup of the KI RWDS.

Year [Reference]	Average Individual Dose (mrem)	Total Worker Dose (person*mrem)
2002 [32]	95	5000
2003 [13]	200	4100
2004 [13]	144	8000
2005 [13]	180	9000

The average of the total worker dose estimates from Table 4.9 will be used for the annual dose to workers from the removal alternative. The average annual individual dose from Table 4.9 is 157 mrem, which will be used for determination of the dose for the other alternatives. Grouting of the site is expected to require 10 workers for 1 year which would result in a cumulative dose of 1570 person*mrem. ISV would require 41 workers for 3 units [24] which, at the above dose rate, would yield an annual cumulative dose of 6437 person*mrem for 2 years. The worker dose for the clean cover alternative is expected to be low and will not be considered, because the workers would mainly be operating heavy machinery on the site which could be shielded to greatly reduce exposure.

4.9 Additional Cost Calculation Information

The dose, perceived risk, change in dose, and change in perceived risk were calculated and PWed using results from the above calculations and the methods outlined in the previous chapter. The VSL used in these calculations was 7 million $ which is the average value for studies based on US work risk-compensation data, which yields a willingness to pay of 5.11 $ per mrem. The VSL value stated above is probably greater would be measured in Russia because it typically varies with GDP per capita; however, because the risk is not voluntary and those exposed are not receiving any benefit this value should be a decent approximation. [33] Also, all values are expressed in USD at the time the decision is being made (2003).

The other costs associated with the contamination in general were not considered in this case study, because they are sunk or mandatory costs and will not affect the cleanup alternative selection. The CGL costs were not considered because there was no data available on the intended future use of the site or similar factors which could make some alternatives preferable to others.

Chapter 5 Results and Discussions

5.1 External Gamma, Inhalation, and Soil Ingestion

5.1.1 Dose

The results of the RESRAD Offsite dose calculation are summarized below in Figures 5.1 - 5.8, which give the average individual dose per year from the external gamma, inhalation, and soil ingestion pathways as a function of position with respect to the site at the following times: 0, 41,102, 202, 304, 508, 701, and 1008 years. The dose is also shown as function of distance from the center of the site on the E to W line (Figure 5.9). From these figures it can be seen that the total dose decreases with time and that the dose decreases with increasing distance from the site, both of which are expected behaviors. The decrease with time is driven by the radioactive decay of ^{137}Cs and ^{90}Sr, which have approximately 30 yr half lives, while the decrease with distance is driven by the increase of area and volume with distance. The maximum dose rate is roughly 12 mrem/yr and occurs at the southeast corner of the site at time 0 yrs. These doses are significant but much less than ICRP's annual public dose limit of 100 mrem (500 mrem max over a 5 year period) [23].

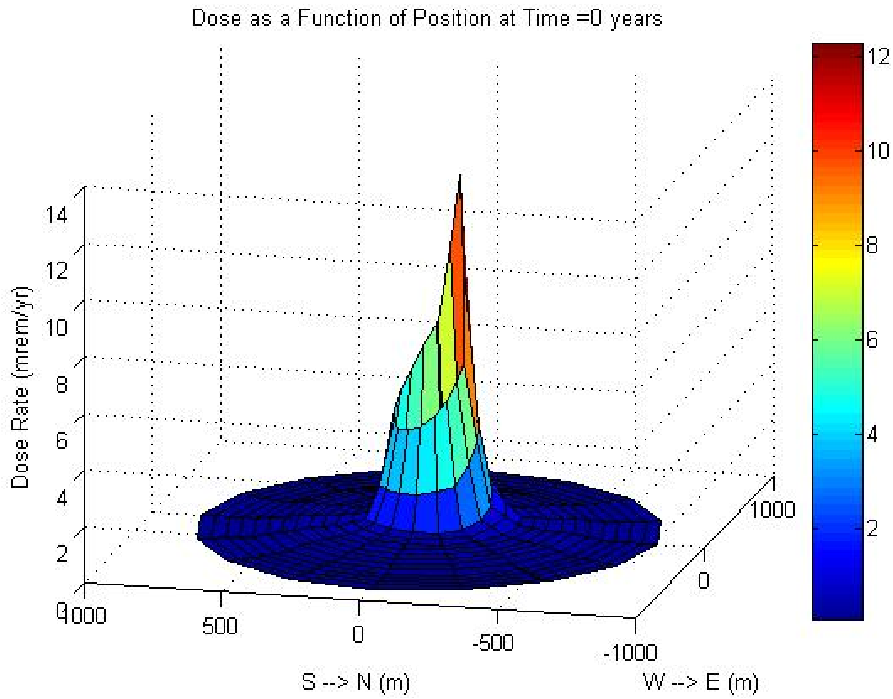


Figure 5.1 - Dose as a function of location relative to the site at time 0 yrs.

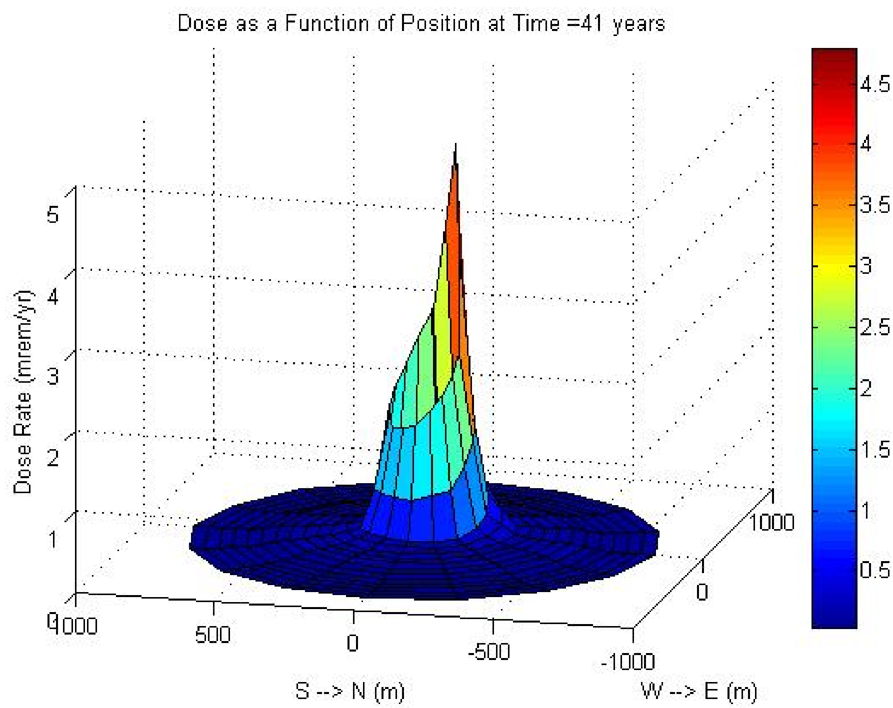


Figure 5.2 - Dose as a function of location relative to the site at time 41 yrs.

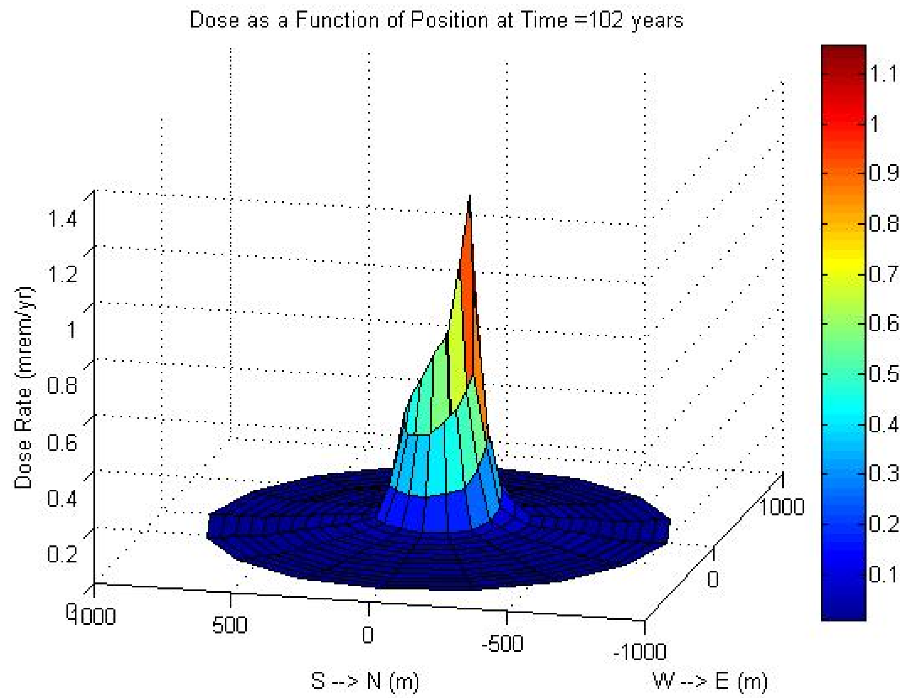


Figure 5.3 - Dose as a function of location relative to the site at time 102 yrs.

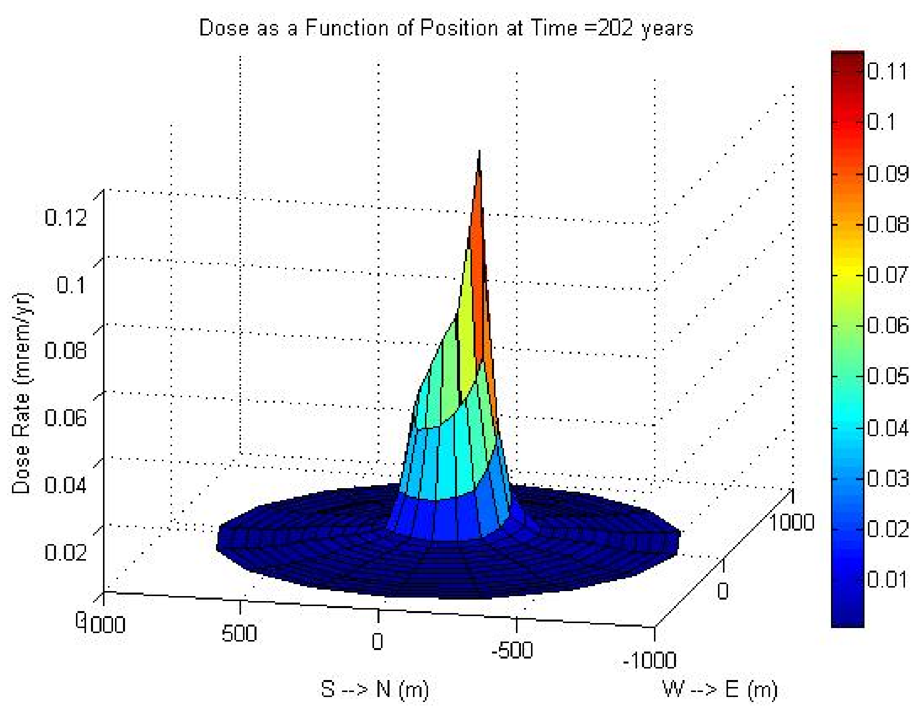


Figure 5.4 - Dose as a function of location relative to the site at time 202 yrs.

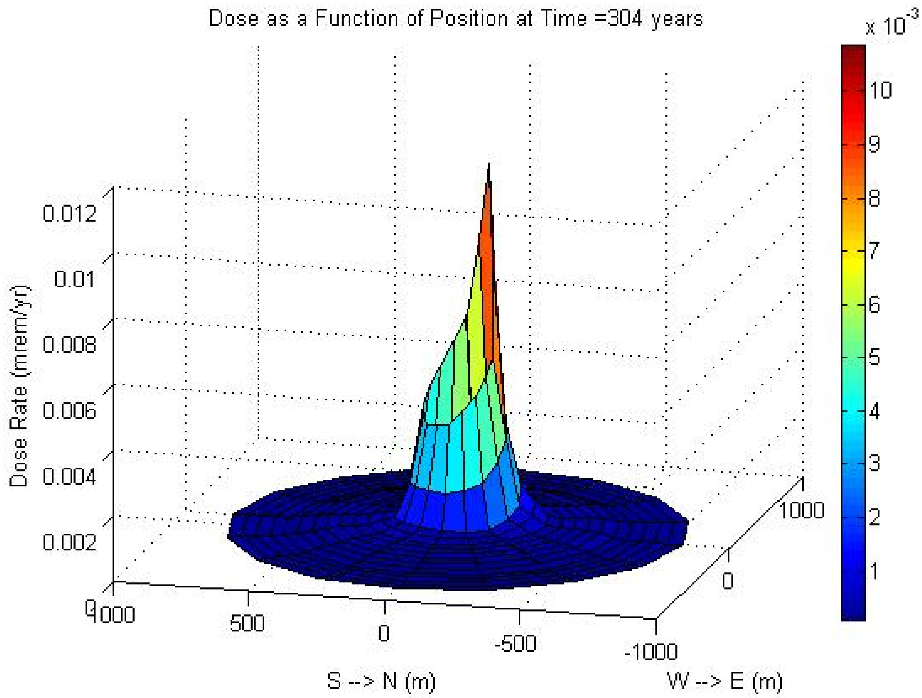


Figure 5.5 - Dose as a function of location relative to the site at time 304 yrs.

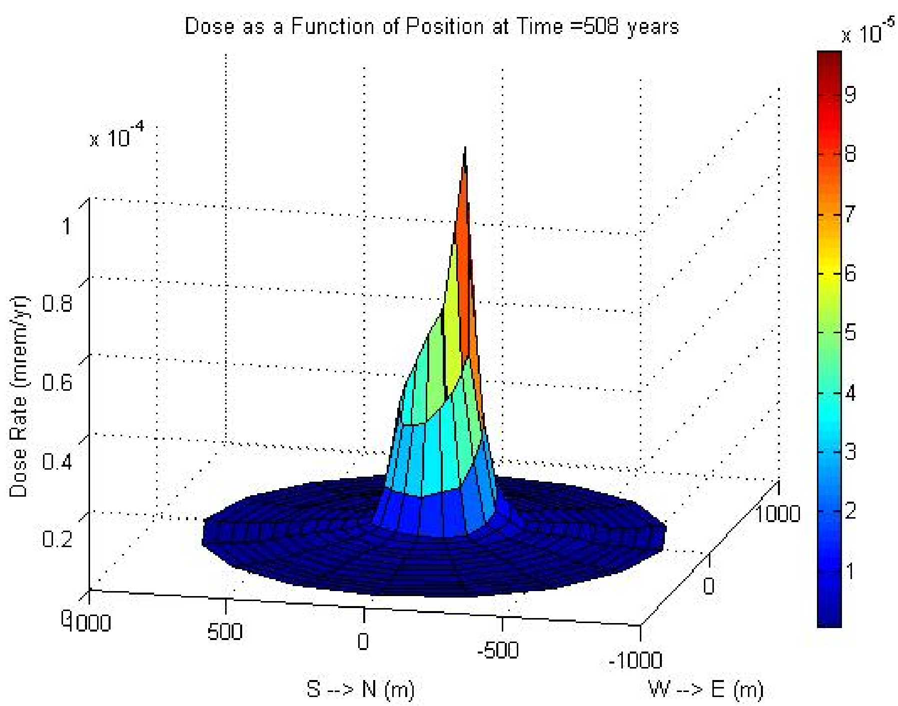


Figure 5.6 - Dose as a function of location relative to the site at time 508 yrs.

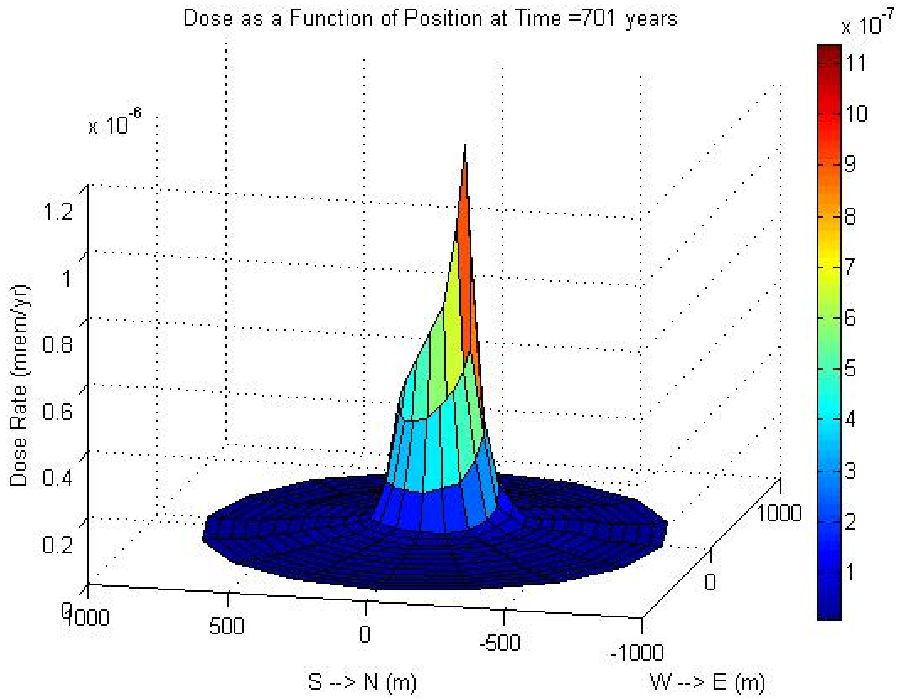


Figure 5.7 - Dose as a function of location relative to the site at time 701 yrs.

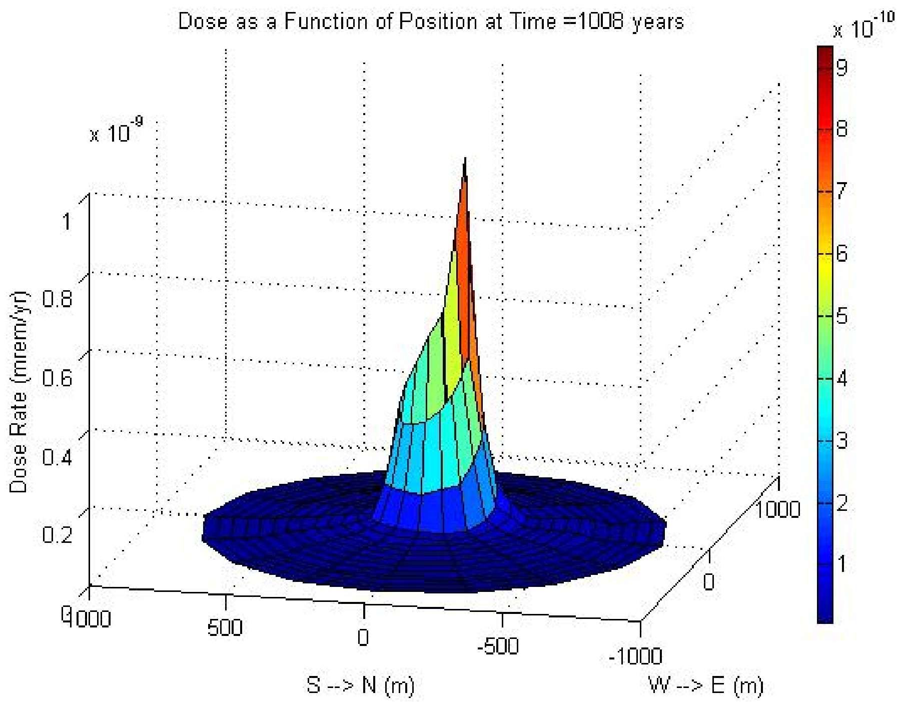


Figure 5.8 - Dose as a function of location relative to the site at time 1008 yrs.

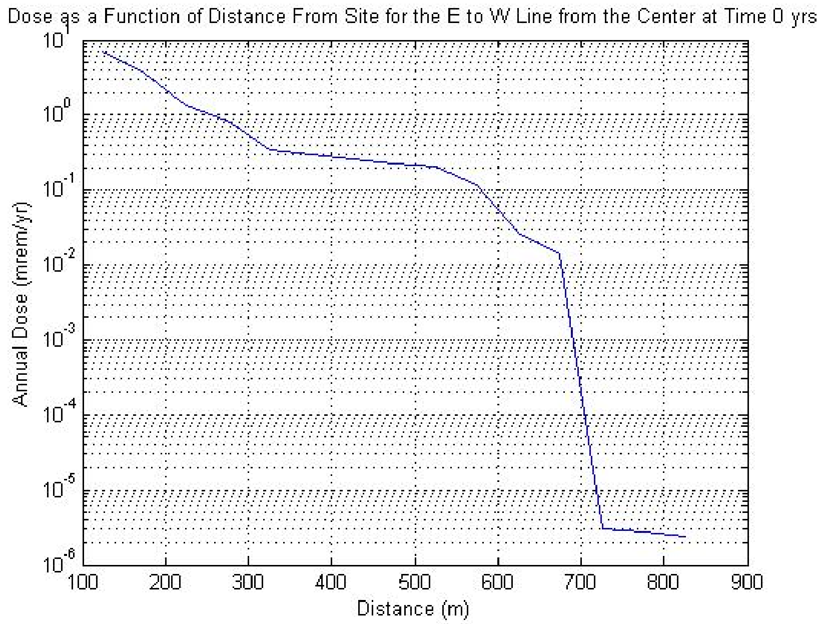


Figure 5.9 - Dose at time 0 yrs as a function of distance from the site from East to West.

5.1.2 Perceived Risk

The average individual perceived risk (dose) from the external gamma, inhalation, and soil ingestion pathways is plotted as a function of location at the following times: 0, 20, 61, 102, 141, and 163 years; in Figures 5.10 – 5.15. These figures show that the perceived dose also decreases with time and distance, but, because the perceived dose for a population group is zero if the dose for that group is less than 1 mrem/yr, the decrease is not as smooth. This also causes the perceived dose to be zero everywhere after approximately 160 years.

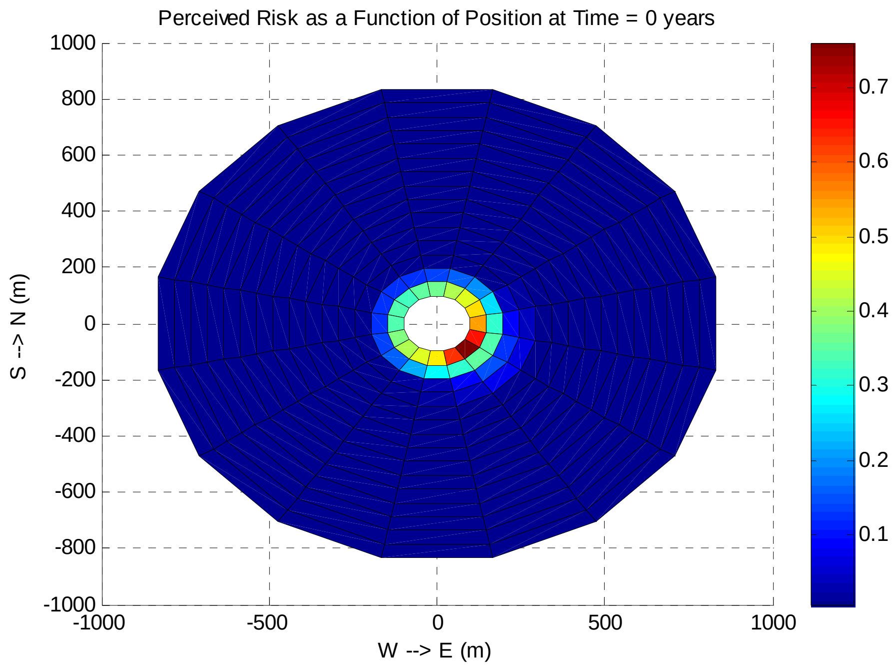


Figure 5.10 - Perceived risk (mrem/yr) as a function of location relative to the site at time 0 yrs.

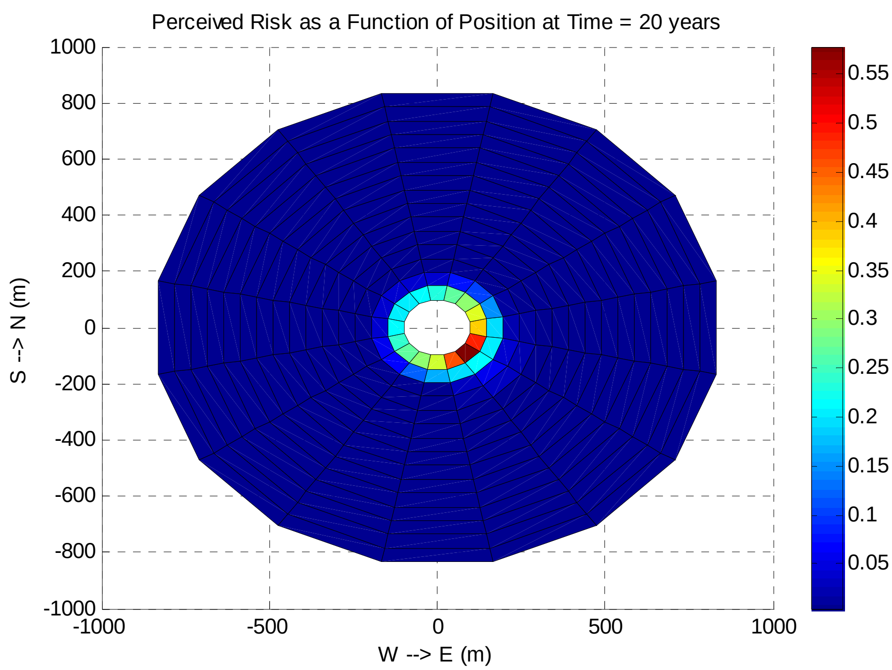


Figure 5.11 - Perceived risk (mrem/yr) as a function of location relative to the site at time 20 yrs.

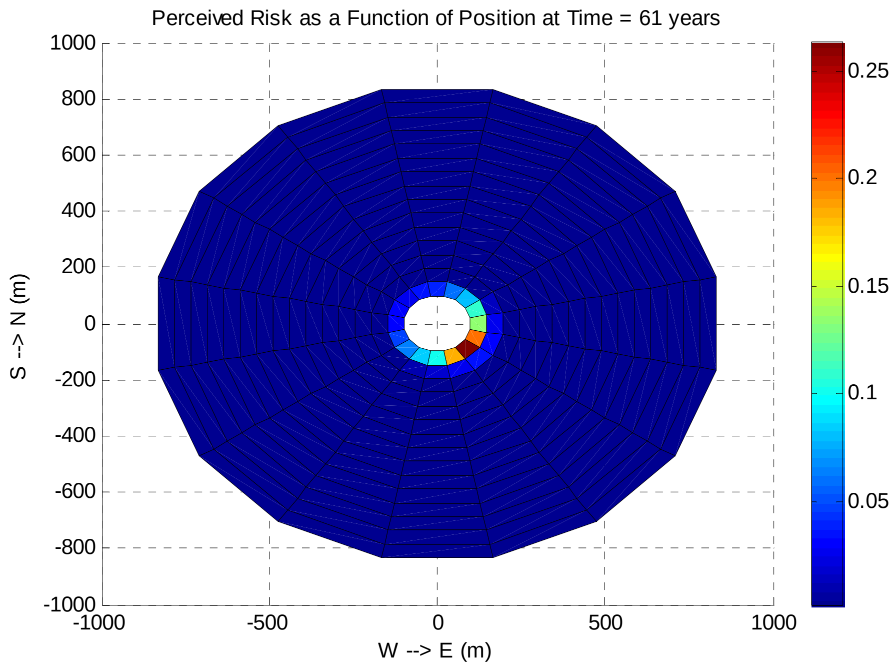


Figure 5.12 - Perceived risk (mrem/yr) as a function of location relative to the site at time 61 yrs.

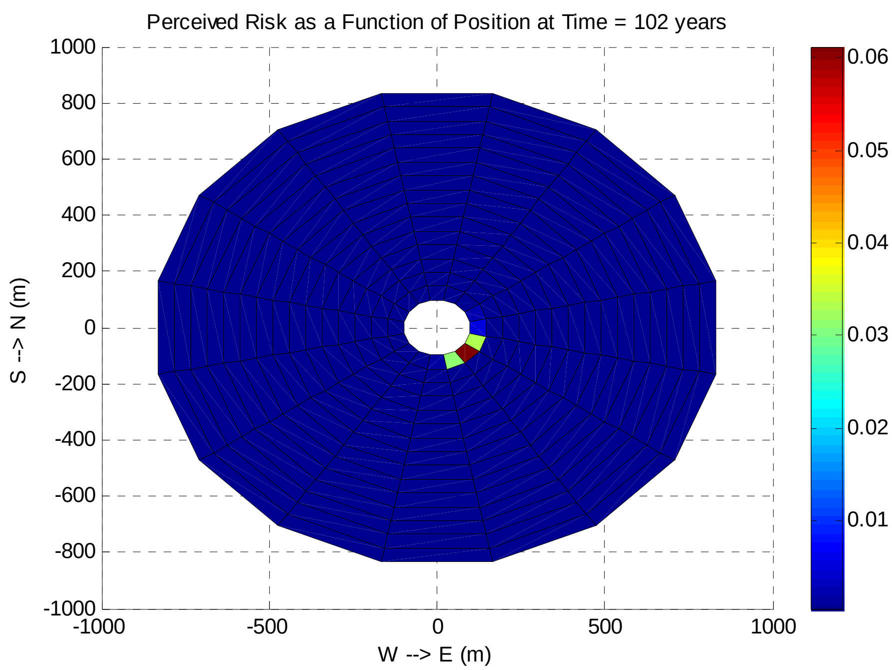


Figure 5.13 - Perceived risk (mrem/yr) as a function of location relative to the site at time 102 yrs.

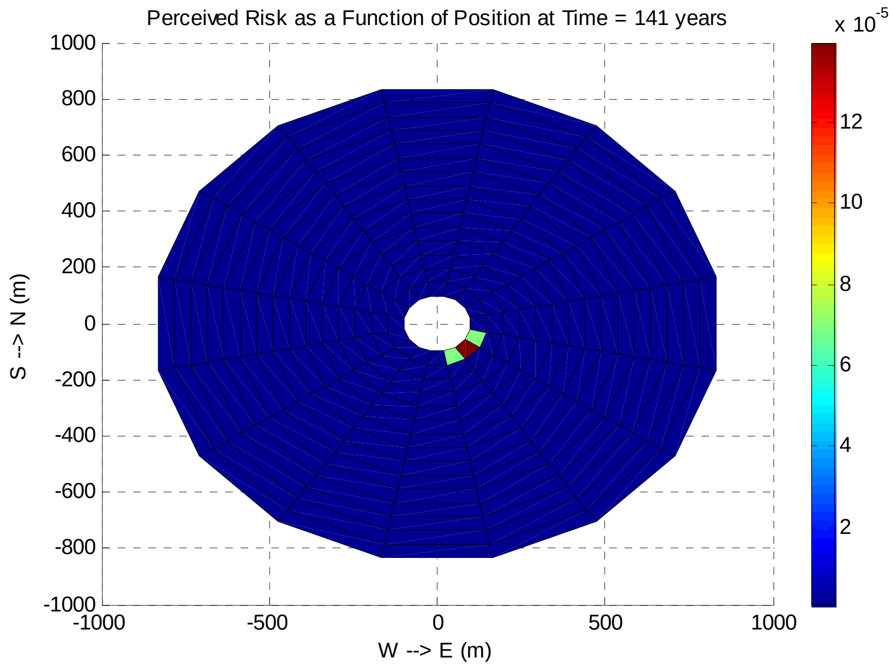


Figure 5.14 - Perceived risk (mrem/yr) as a function of location relative to the site at time 141 yrs.

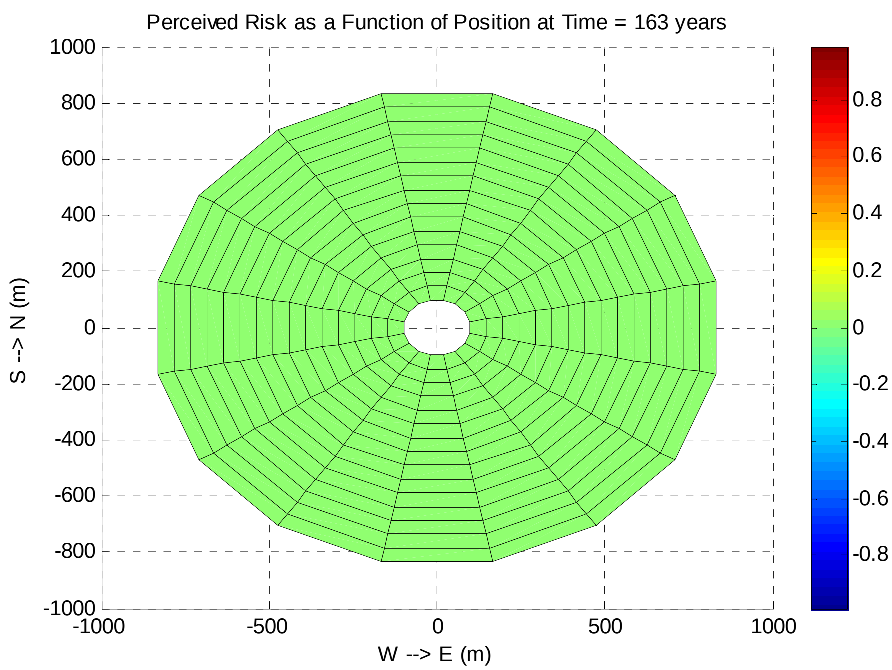


Figure 5.15 - Perceived risk (mrem/yr) as a function of location relative to the site at time 163 yrs.

5.1.3 Value of Dose and Perceived Risk

The present worth of the dose and perceived dose from the external gamma, inhalation, and soil ingestion pathways was determined by fitting the time dependent dose data with a function of time at each population location. Examples of these fits are given in Figures 5.16 and 5.17. This was done because RESRAD Offsite outputs the time integrated dose so that the dose over a year is given by the value at the beginning of the year. Each of these functions was converted to value by multiplying them by the number of people that live at the location they represent and the value conversion factor (5.11 $ per person*mrem). The PW of each was then calculated using Equation 4 and the results were summed to yield the PW of dose and PW of perceived dose versus discount rate plots in Figures 5.18 and 5.19, respectively.

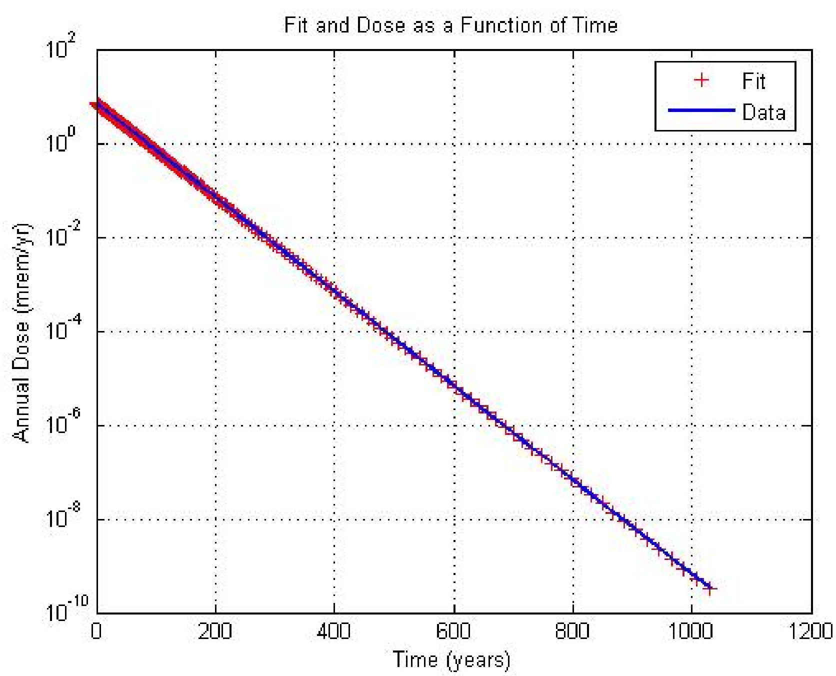


Figure 5.16 - Fit of dose(t) at the receptor due East of the site to an exponential.

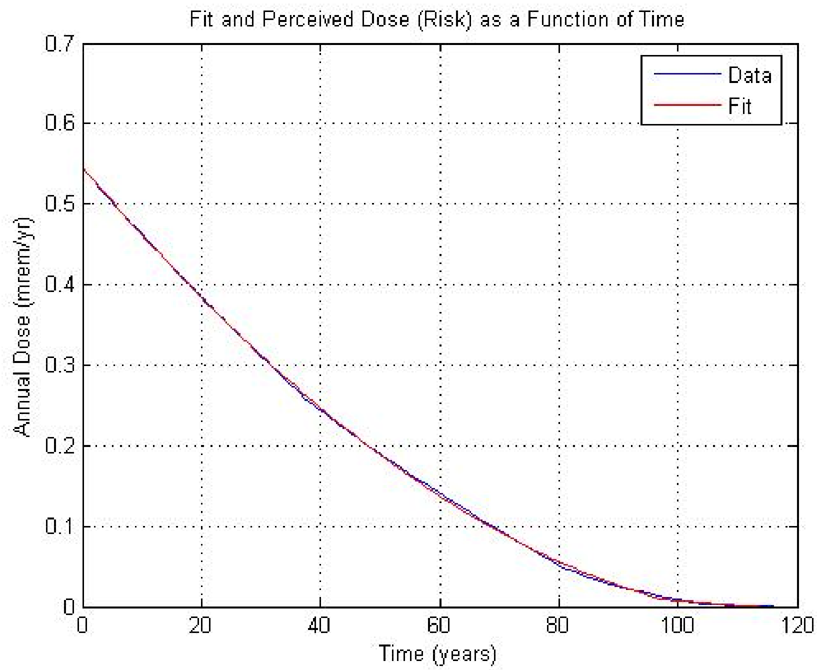


Figure 5.17 - Fit of perceived dose(t) at the receptor due East of the site to an to a piecewise function consisting of a cubic followed by two linear sections.

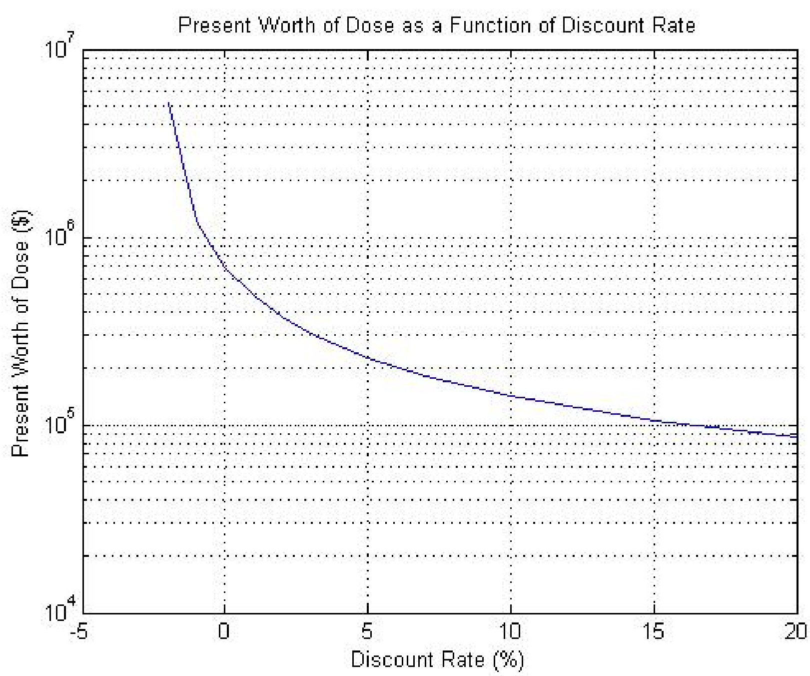


Figure 5.18 - PW of the dose from the external gamma, inhalation, and soil ingestion pathways as a function of discount rate.

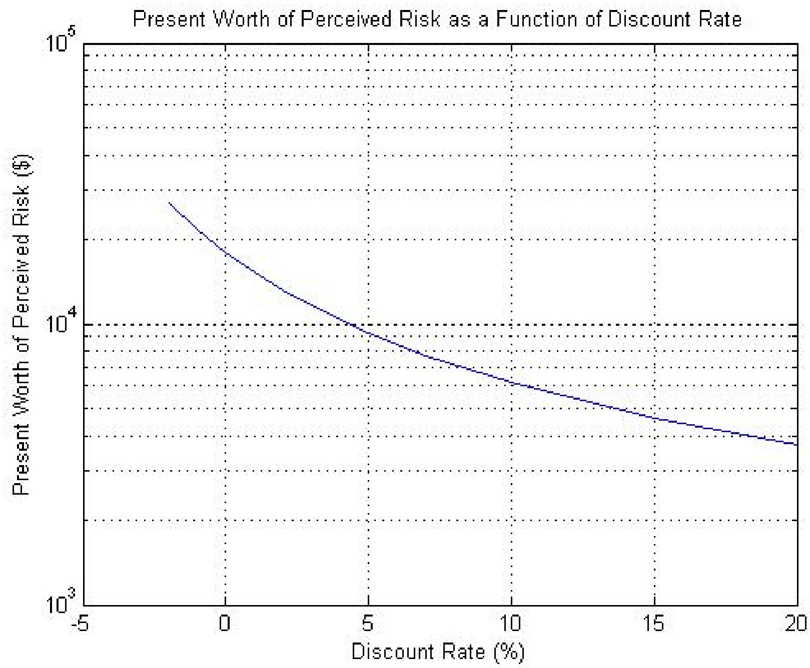


Figure 5.19 - PW of the perceived risk (dose) from the external gamma, inhalation, and soil ingestion pathways as a function of discount rate.

5.2 Value of Dose and Perceived Risk from Groundwater

5.2.1 Leach Rate of the Contaminants from the Site into the Aquifer

The first series of results for the groundwater transport calculations is the flux of contamination from the waste into the aquifer. The time-dependent activity flux of ^{90}Sr from the contamination into the aquifer is given in Figures 5.20 and 5.21 for scenarios 1a and 1b, respectively. The activity fluxes for the nuclides in scenario 2 are presented in Figures 5.22-5.24. These figures show that grouting and vitrifying the soil both lead to much lower release rates than the unaltered soil, which is expected because the purpose of these remediation actions is to limit the mobility of the contaminants. The difference in leach rates between the grouted and vitrified wastes should not be taken as a comparison between these methods, because some of the parameters were chosen as upper limits from ranges while others where only available as nominal values so the conservatism of the leach rates varies from one calculation to another; and, consequently, they are not necessarily comparable.

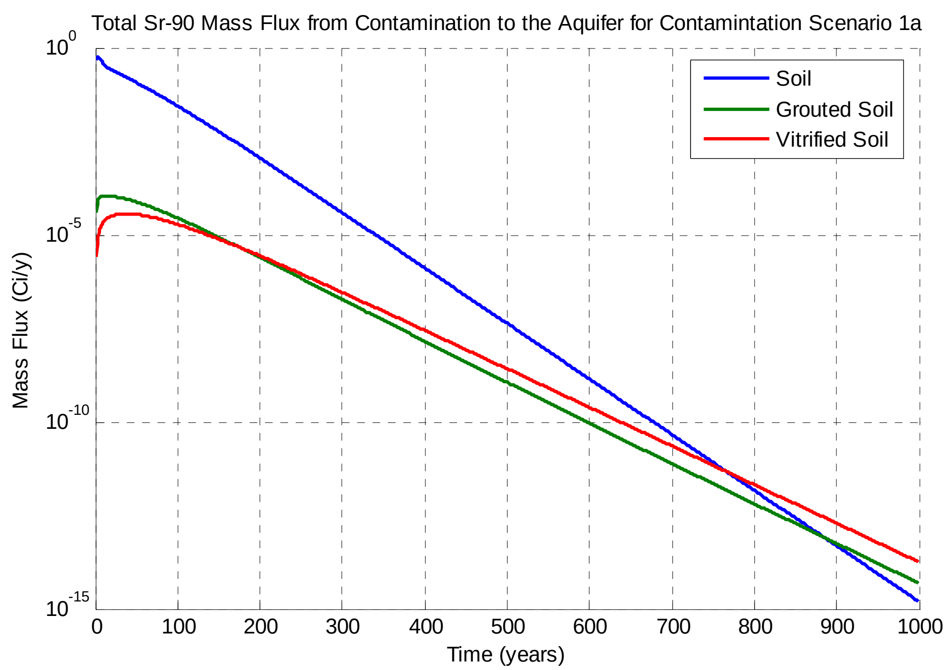


Figure 5.20 - Time-dependent release rate of ^{90}Sr from the contamination into the aquifer for scenario 1a.

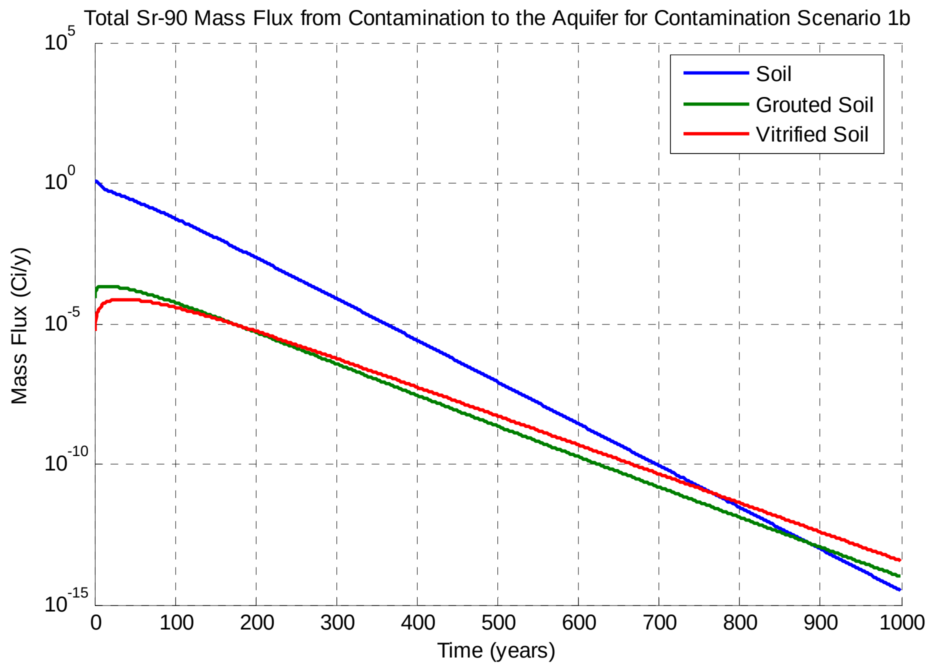


Figure 5.21 - Time-dependent release rate of ^{90}Sr from the contamination into the aquifer for scenario 1b.

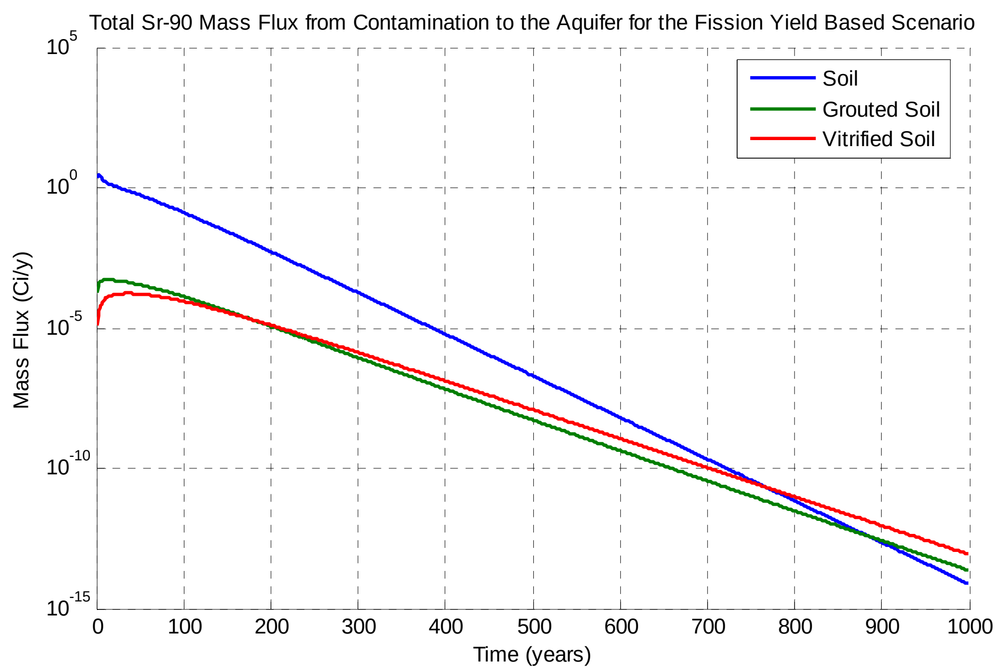


Figure 5.22 - Time-dependent release rate of ^{90}Sr from the contamination into the aquifer for the fission yield based scenario.

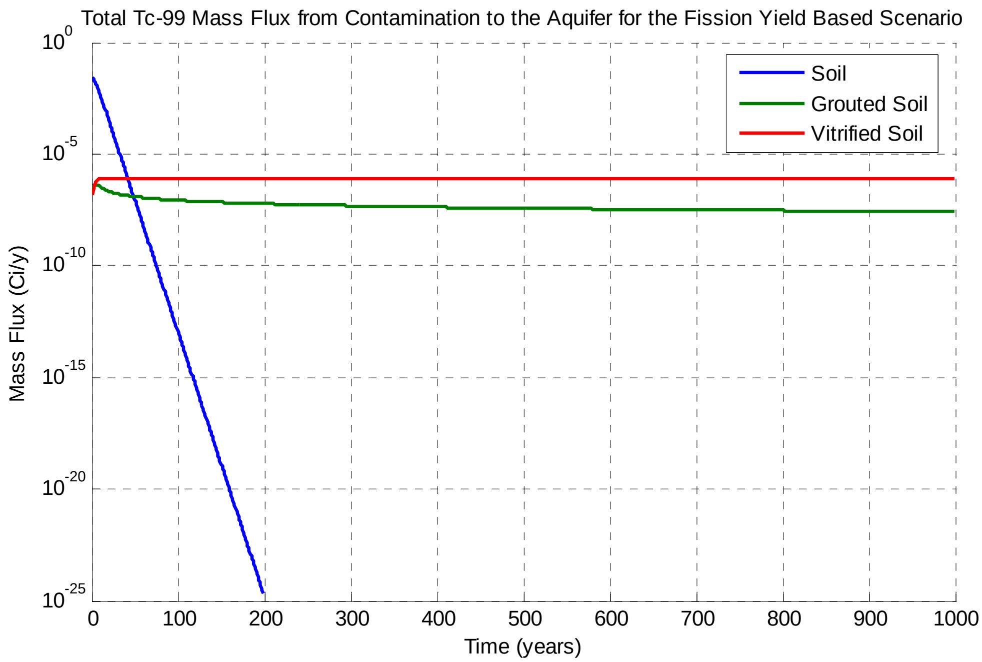


Figure 5.23 - Time-dependent release rate of ^{99}Tc from the contamination into the aquifer for the fission yield based scenario.

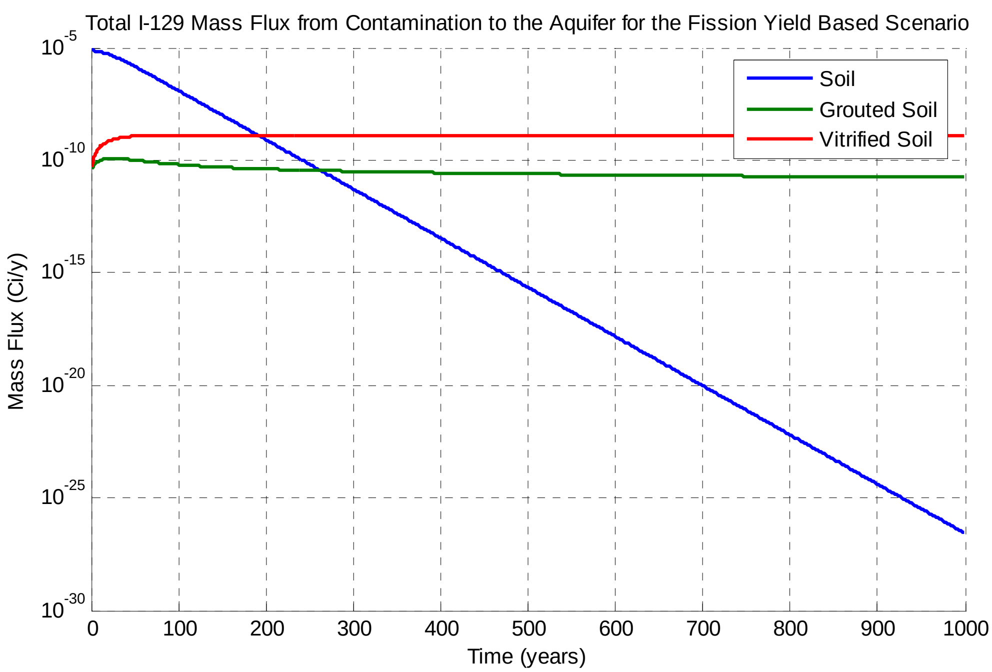


Figure 5.24 - Time-dependent release rate of ^{129}I from the contamination into the aquifer for the fission yield based scenario.

5.2.2 Dose and Perceived Risk from Drinking Contaminated Groundwater

The time-dependent average individual dose and perceived risk for a person drinking water from a hypothetical well located in the nearest feasible offsite location to the groundwater plume for each of the site condition and contamination scenarios considered was determined. The individual dose for scenario 1 (.9*1a+.1*1b) is shown in Figure 5.25. The peak dose of 1.19 mrem/yr occurs at the beginning of the simulation period. The time dependent perceived individual risk for scenario 1 from the groundwater contamination is given in Figure 5.26. Despite the much higher release for the unaltered soil scenario the dose is almost the same for all three alternatives. This is because the groundwater flow beneath the site is slow and ^{90}Sr has a moderate K_d of 5.1 so by the time the ^{90}Sr plume from the site reaches the hypothetical offsite well at approximately 300 yrs most of the contamination has decayed.

The ^{90}Sr dose for scenario 2 (Figure 5.27) behaves similar to the dose in scenario 1 except the soil dose is higher, because the ^{90}Sr concentration in the soil is higher, which leads to a second dose peak between 300 and 500 yrs. The dose from ^{99}Tc is presented in Figure 5.28. As can be seen in the figure, the dose from ^{99}Tc peaks, for the soil scenario, at .24 mrem/yr in year 28 but drops to 10^{-5} mrem/yr by around year 77 due to depletion of the source term and rapid movement through the aquifer. The doses from the grouted and vitrified soil never get much higher than around 10^{-5} mrem/yr but remain virtually constant because of the slow, steady leaching from the waste forms. The ^{129}I dose, shown in Figure 5.29, is about an order of magnitude less than the ^{99}Tc dose for all scenarios and the peak of .0145 mrem/yr, for the soil scenario, occurs at 116 yrs. The combined perceived risk for scenario 2 (Figure 5.30) is the same as that for scenario 1 except for a small increase between 20 and 40 yrs caused by the ^{99}Tc peak during this interval.

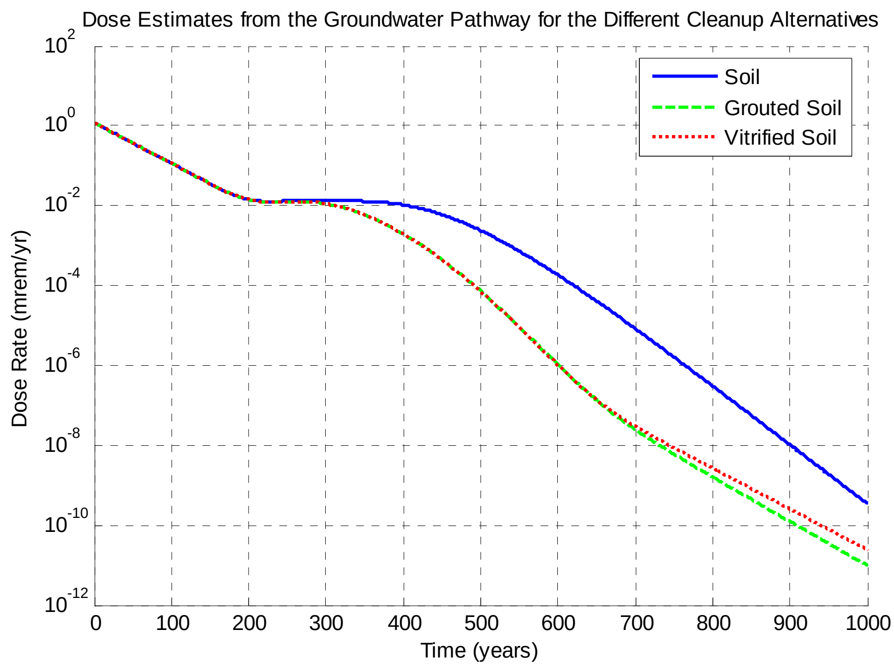


Figure 5.25 - Average individual dose rate to a person drinking water exclusively from the hypothetical contaminated well for scenario 1.

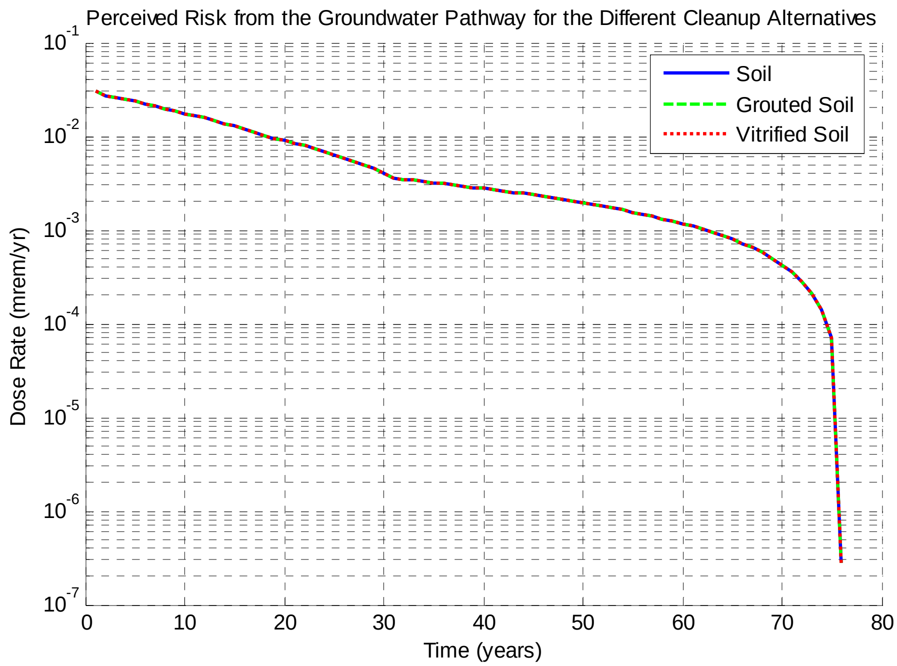


Figure 5.26 - Average individual perceived risk of a person drinking water from the hypothetical contaminated well for scenario 1 (0 after 76 years).

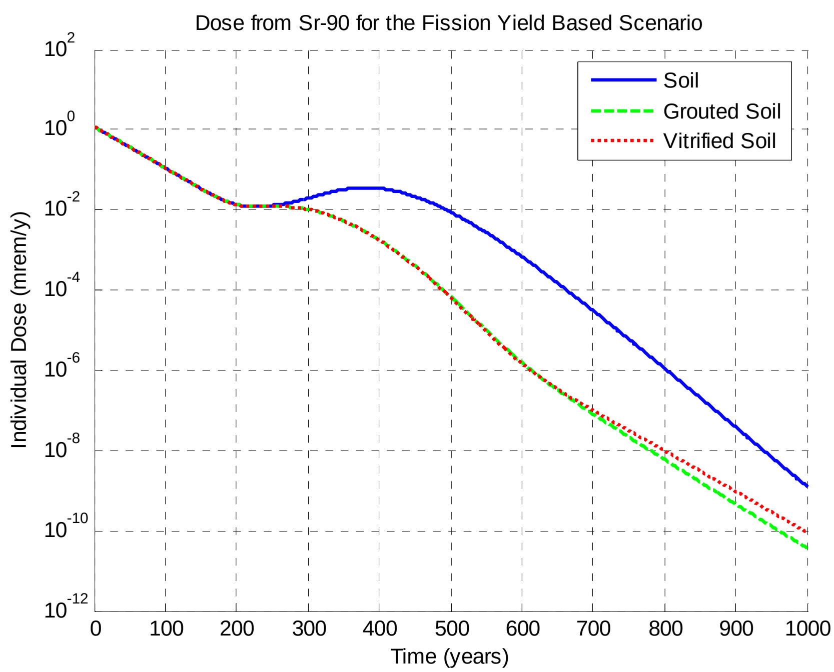


Figure 5.27 - Average individual dose from ^{90}Sr to a person drinking water from the hypothetical contaminated well for fission yield based scenario.

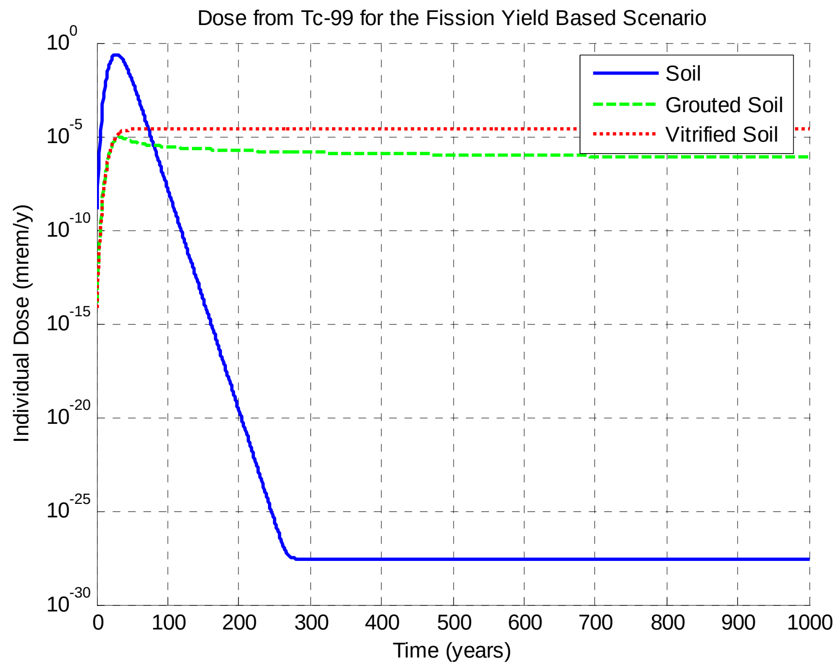


Figure 5.28 - Average individual dose from ^{99}Tc to a person drinking water from the hypothetical contaminated well for fission yield based scenario.

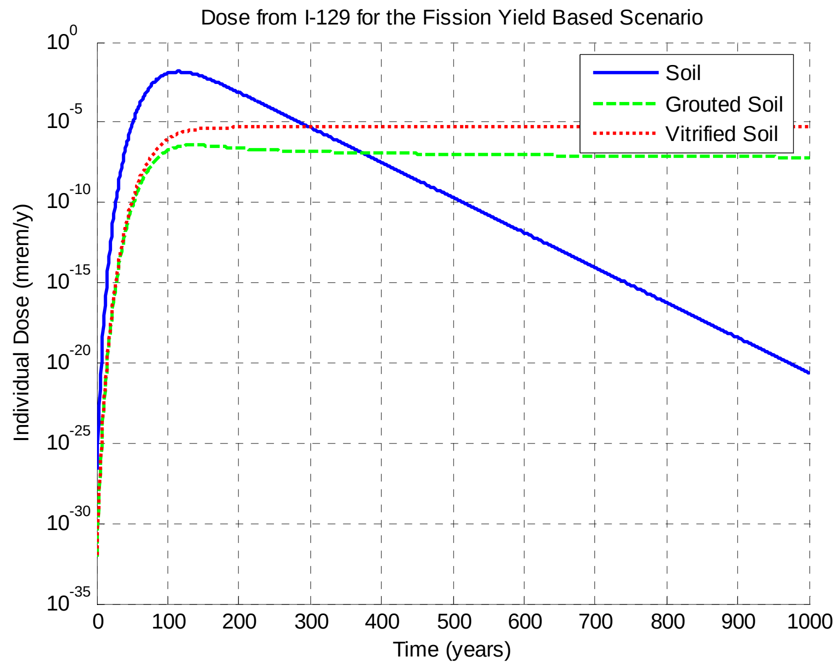


Figure 5.29 - Average individual dose from ^{129}I to a person drinking water from the hypothetical contaminated well for fission yield based scenario.

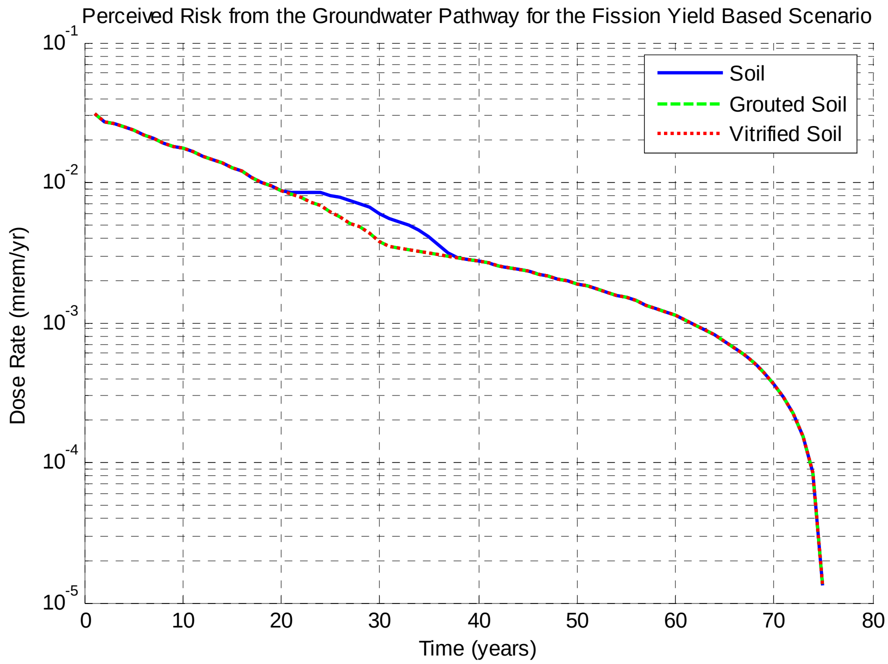


Figure 5.30 - Average individual perceived risk from all radionuclides to a person drinking water from the hypothetical contaminated well for fission yield based scenario.

5.2.3 Value of Dose and Perceived Risk

The PW of the dose and perceived risk from scenario 1 for each of the cleanup alternatives are given in Figures 5.31 and 5.32, respectively. The PW of the dose at a discount rate of 0 % is only about 1600 \$, and the perceived risk for this rate is less than 20 \$. The PW of the dose and perceived risk for scenario 2 are given in Figures 5.33 and 5.34, respectively. The only major difference between the scenario 2 results and the scenario 1 results is that the PW of the dose for the vitrified soil using a discount rate of 0 % is roughly an order of magnitude greater than the other cleanup alternatives because the dose rate for the vitrified soil is relatively constant with time. Consequently, unless a negative discount rate is selected there is no difference between the scenarios.

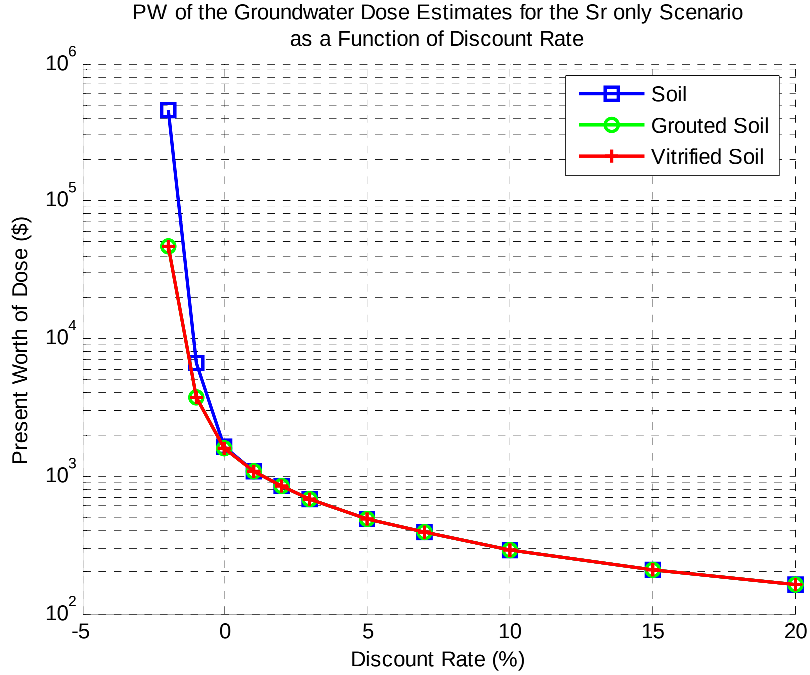


Figure 5.31 - PW of the expected dose from the groundwater pathway for various cleanup alternatives as a function of discount rate for scenario 1.

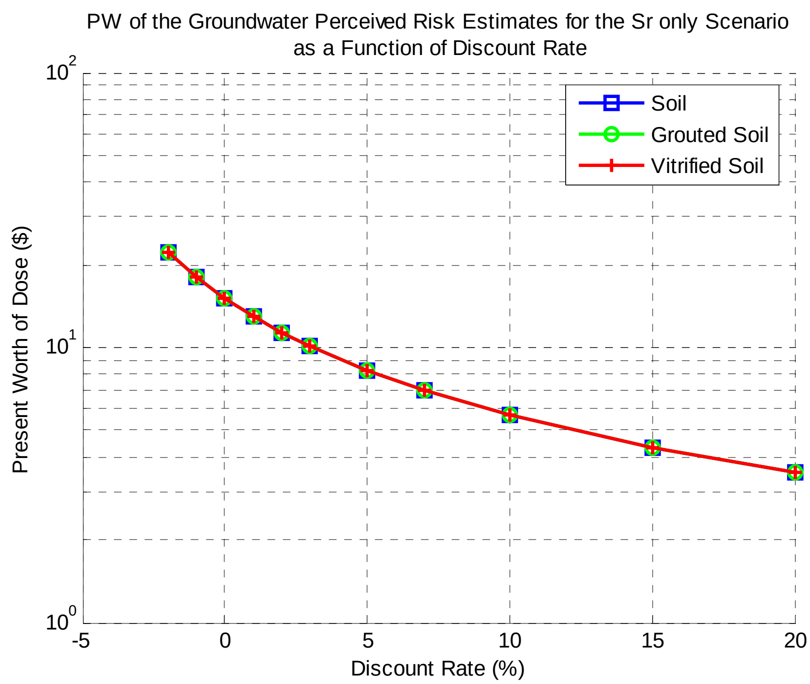


Figure 5.32 - PW of the expected perceived risk from the groundwater pathway for various cleanup alternatives as a function of discount rate for scenario 1.

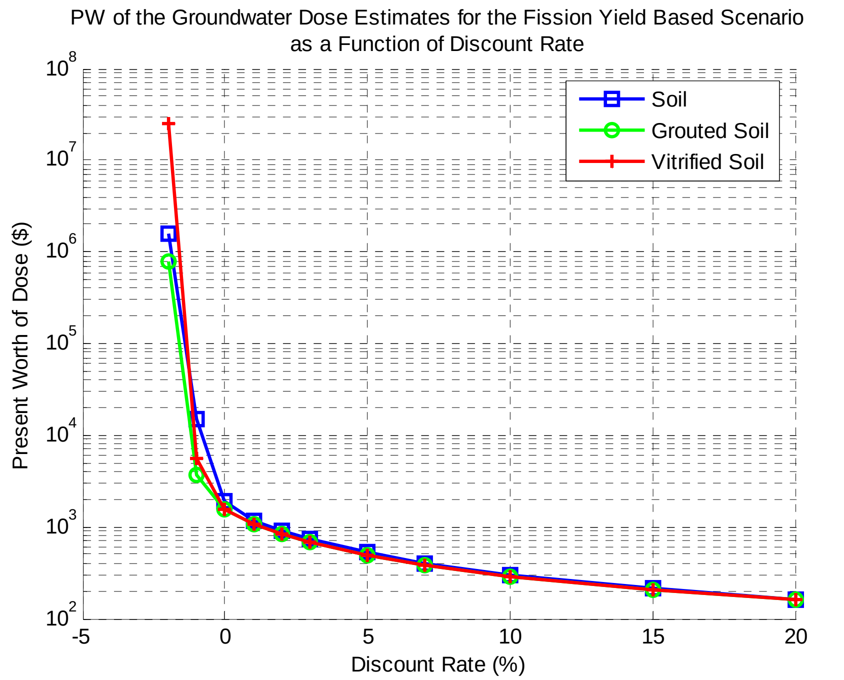


Figure 5.33 - PW of the expected dose from the groundwater pathway for various cleanup alternatives as a function of discount rate for scenario 2.

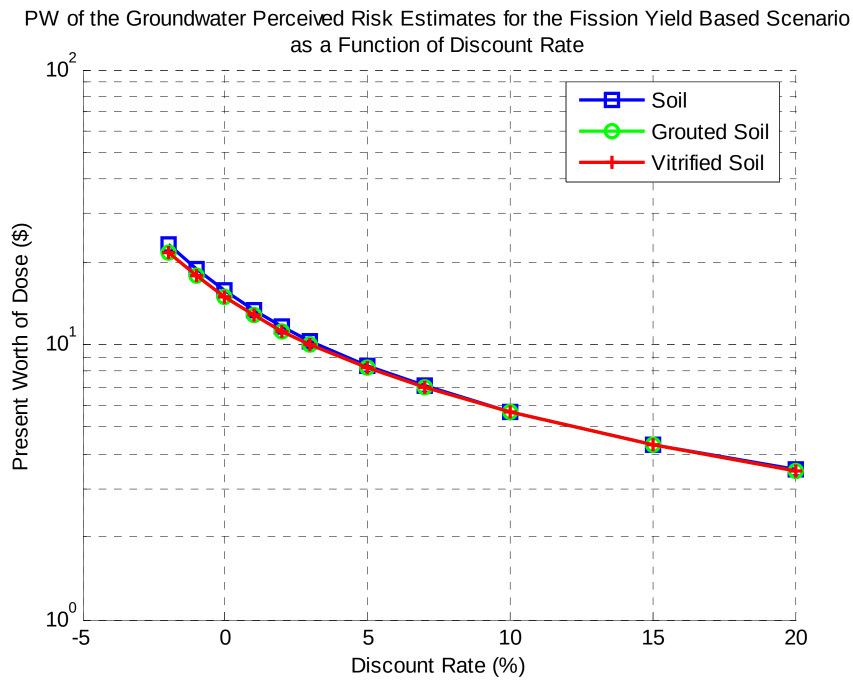


Figure 5.34 - PW of the expected perceived risk from the groundwater pathway for various cleanup alternatives as a function of discount rate for scenario 2.

5.3 Value of Dose and Perceived Risk from Surface Water

The results of the surface water pathway dose calculations for each of the source terms considered to people in the municipal car park and street adjacent to the car park are presented in Figures 5.35 and 5.36, respectively. Plots of the perceived risk (dose) for the car park and street are included in Figures 5.37 and 5.38, respectively. As should be expected, the values for the dose and perceived dose are highest for the plus sigma source term and lowest for the minus sigma source term. However, the difference between the base contamination and the half Sr case is very small, which was not expected. This result means that inhalation dose is not very significant for the surface water pathway, as was the case for the inhalation pathway in the RESRAD Offsite calculation. The PW of the dose and perceived dose from the surface water pathway as a function of discount rate are shown in Figures 5.39 and 5.40, respectively.

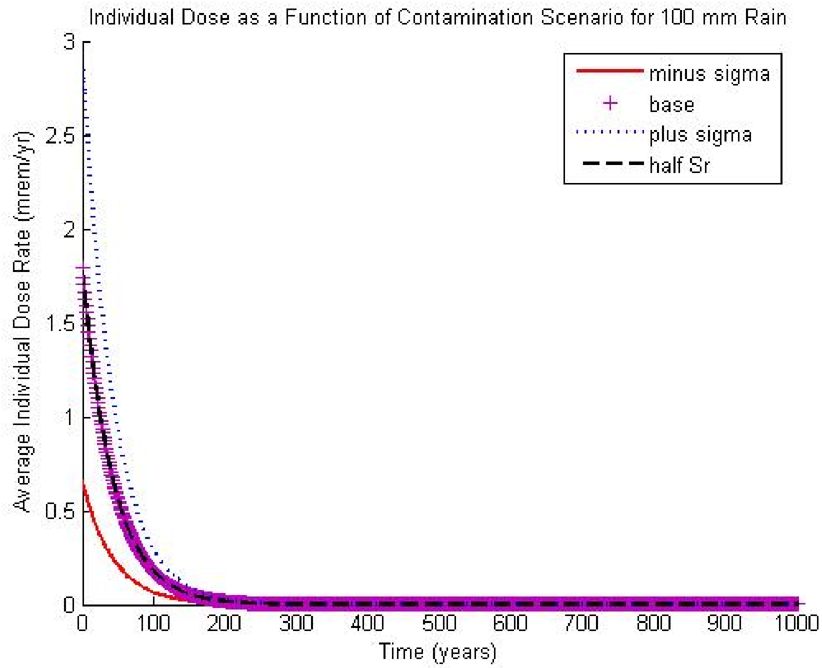


Figure 5.35 - Average time-dependent individual dose rate to people that use the car park from the surface water pathway that would result from a 100 mm rain event.

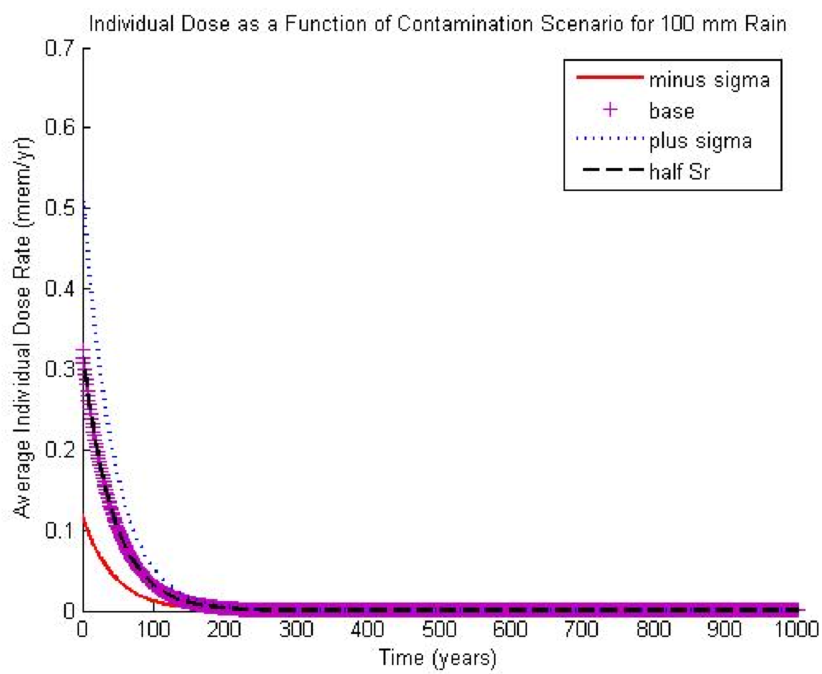


Figure 5.36 - Average time-dependent individual dose rate to people that use the street from the surface water pathway that would result from a 100 mm rain event.

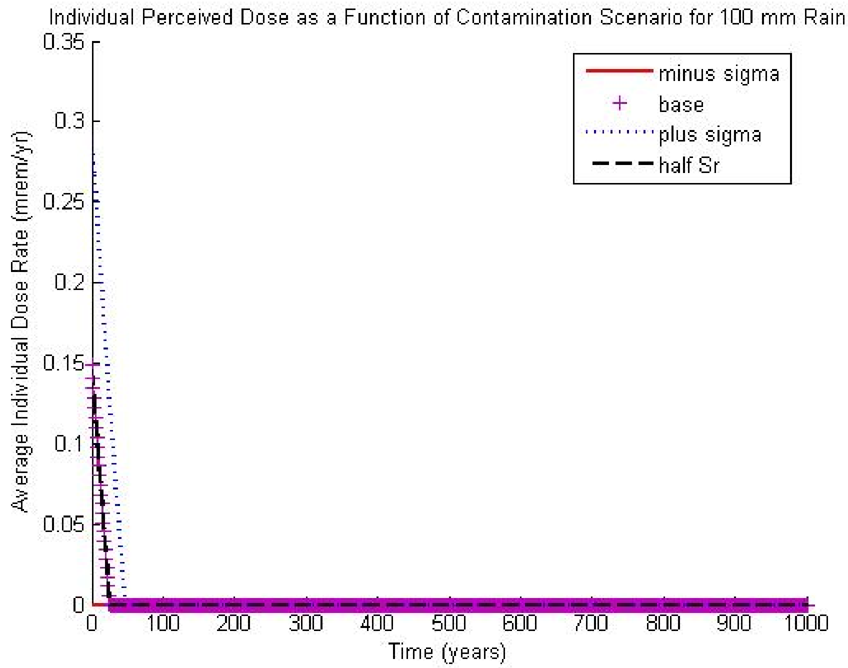


Figure 5.37 - Average time-dependent perceived dose (risk) of the people who use the car park from the surface water pathway that would result from a 100 mm rain event.

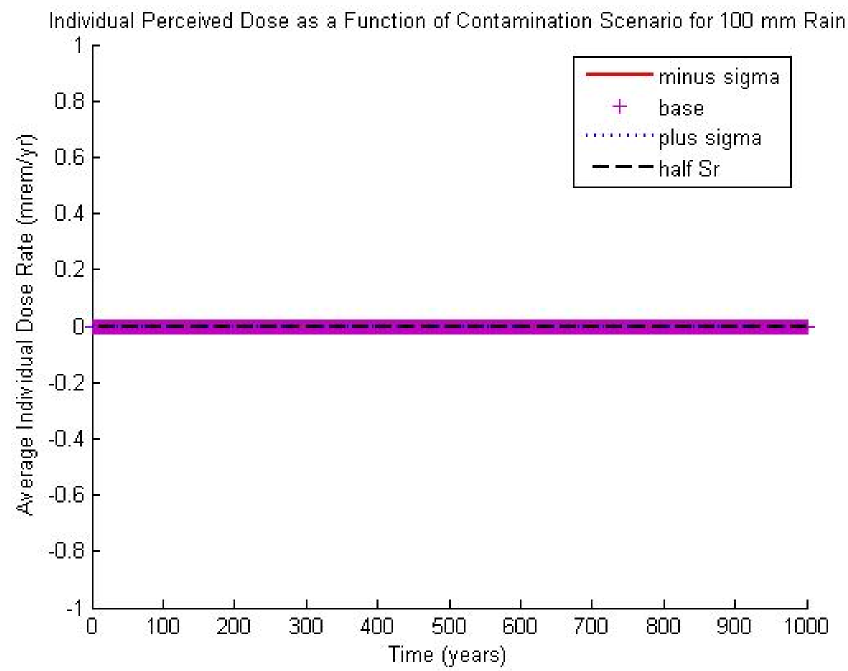


Figure 5.38 - Average time-dependent perceived dose (risk) of the people who use the street from the surface water pathway that would result from a 100 mm rain event.

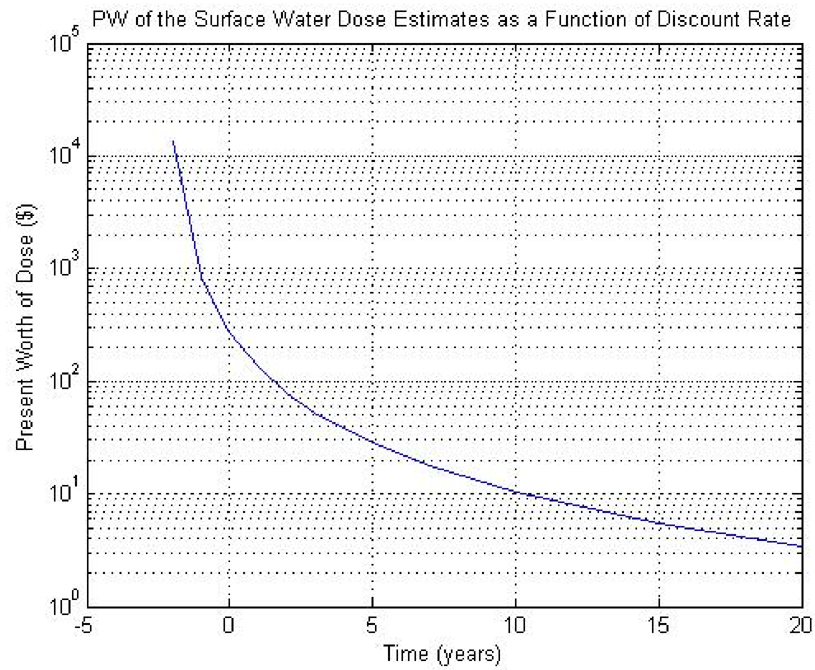


Figure 5.39 - PW of the dose from the surface water pathway as a function of discount rate.

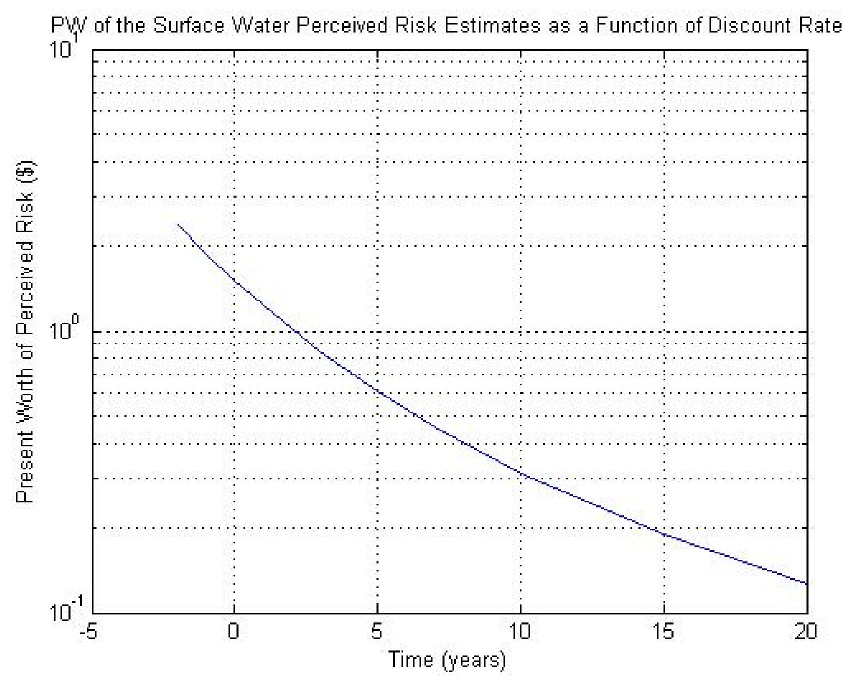


Figure 5.40 - PW of the perceived dose (risk) from the surface water pathway as a function of discount rate.

5.4 PW of the Cost of the Cleanup Alternatives and Worker Dose

The PW of the cost of the cleanup alternatives and estimated worker dose from the completion of each cleanup alternative is presented as a function of discount rate in Figures 5.41 and 5.42, respectively. The cost of the cleanup alternatives follows the expected trend based on the unit cost data, with clean cover being by far the cheapest alternative. The worker dose follows the same trend as the cost data, because the more expensive alternatives require more workers onsite for more time.

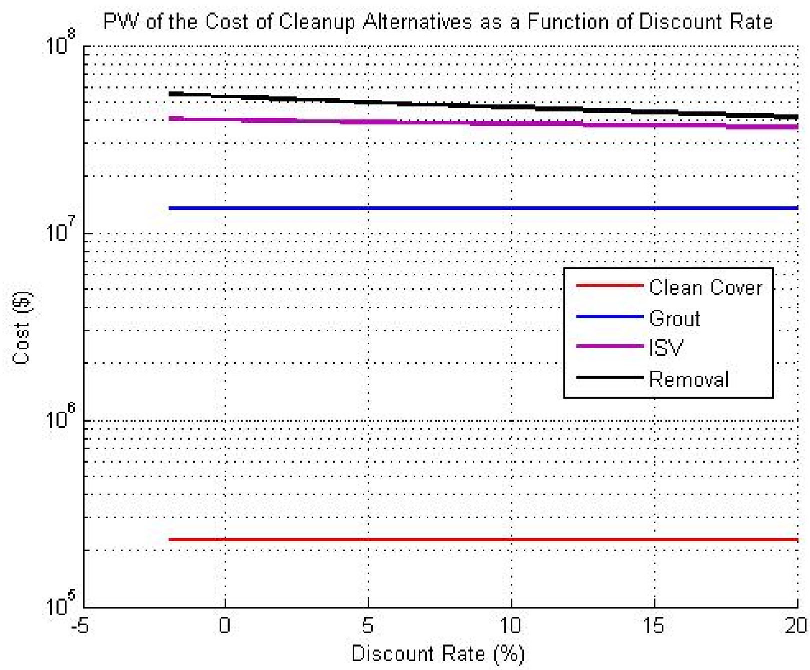


Figure 5.41 - The PW of the cleanup alternatives considered as a function of discount rate.

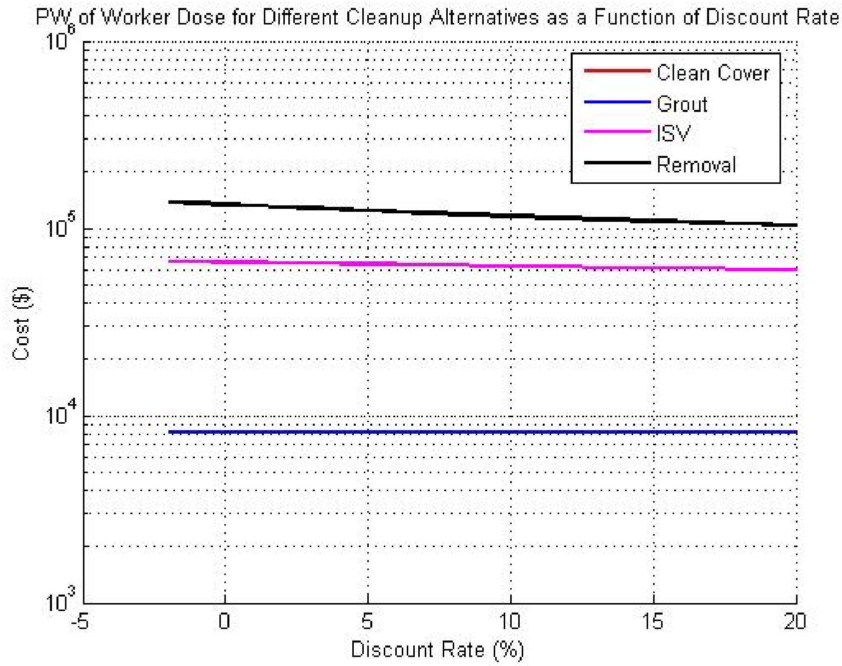


Figure 5.42 - The PW of the dose to workers performing the cleanup alternatives (clean cover is zero so it doesn't show up).

5.5 Dose Reduction from the Cleanup Alternatives

The results of the calculations for the dose to the public from the contamination due to the external gamma, inhalation, and soil ingestion pathways after the completion of the cleanup alternatives involving clean cover over the site are presented in Figures 5.43 – 5.45, which give the dose as a function of position for times of 0, 121, and 304, respectively. A plot of the dose as a function of time for the receptor 100m due East of the site is also included as Figure 5.46. These figures show that the dose to the public is very, very low at all receptor locations for all times considered and is considered negligible. The PW of the dose and perceived risk incurred by the population, assuming linear reduction, during the cleanup are given in Figures 5.47 and 5.48, respectively.

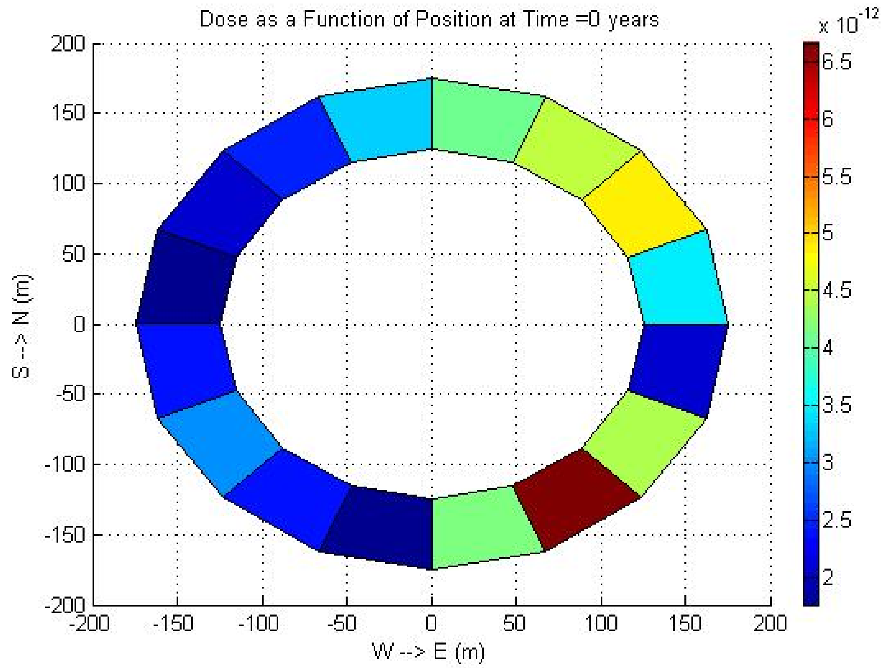


Figure 5.43 - Dose (mrem/yr) as a function of location at time 0 yrs after site has been covered by clean fill.

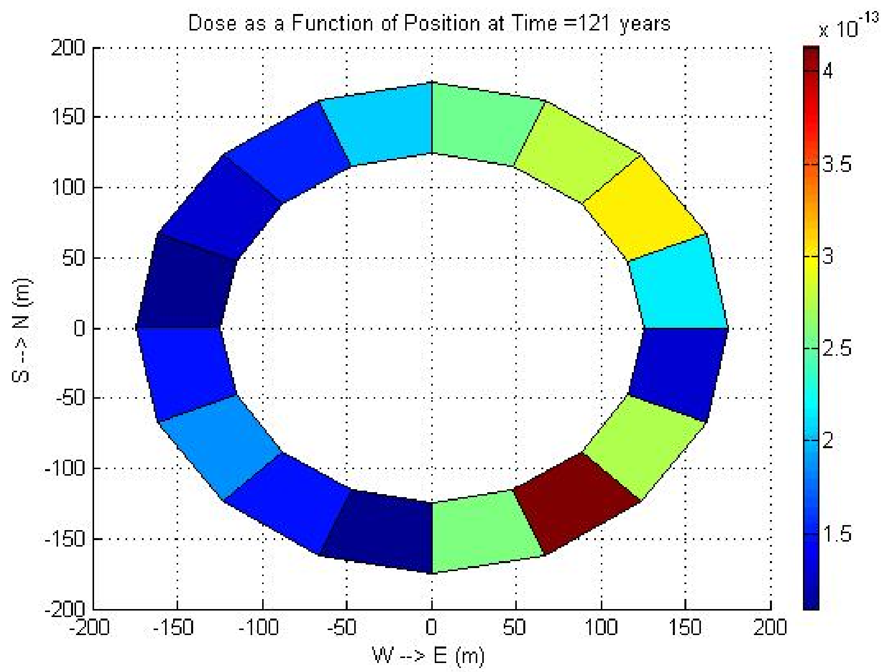


Figure 5.44 - Dose (mrem/yr) as a function of location at time 121 yrs after site has been covered by clean fill.

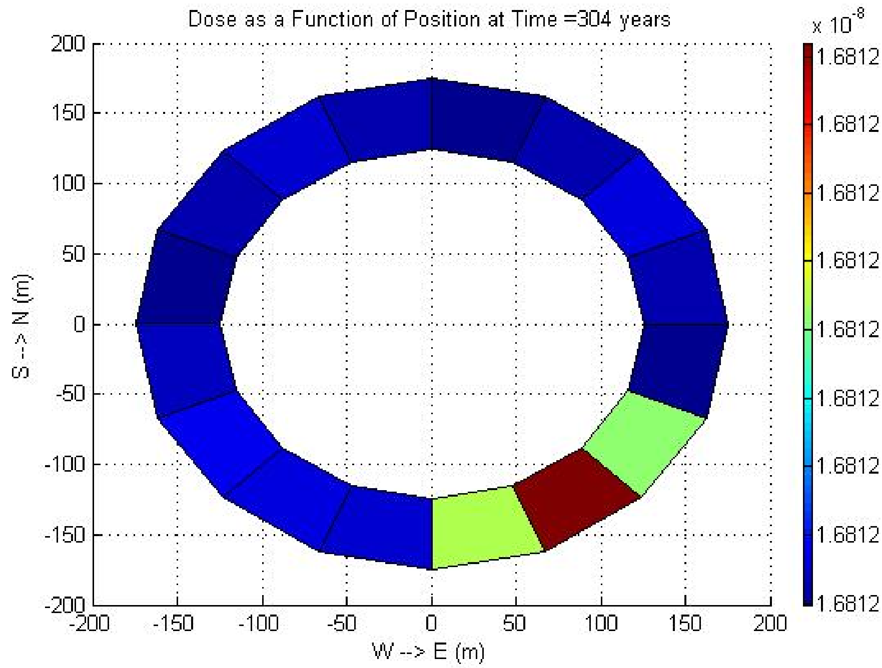


Figure 5.45 - Dose (mrem/yr) as a function of location at time 304 yrs after site has been covered by clean fill.

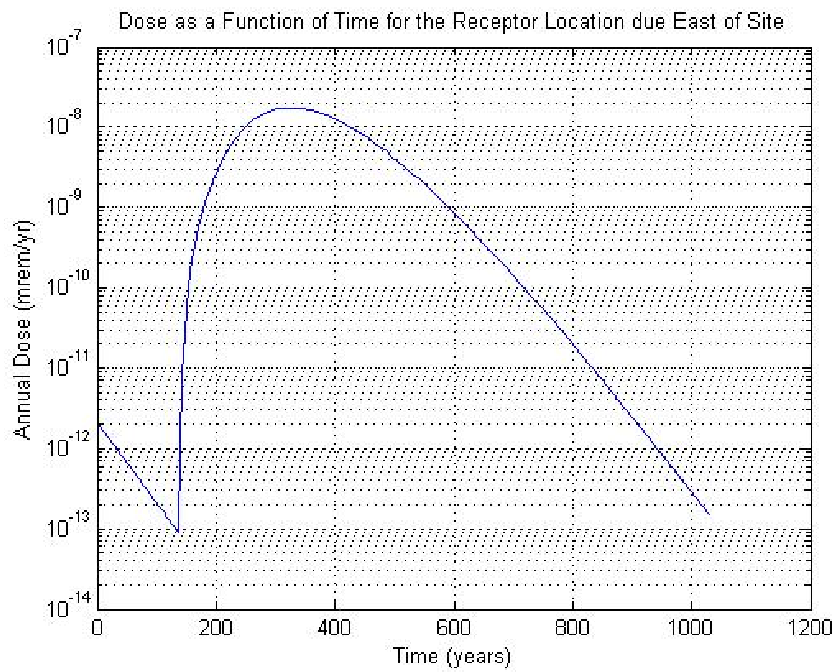


Figure 5.46 - Dose as a function of time after the site has been covered by clean fill.

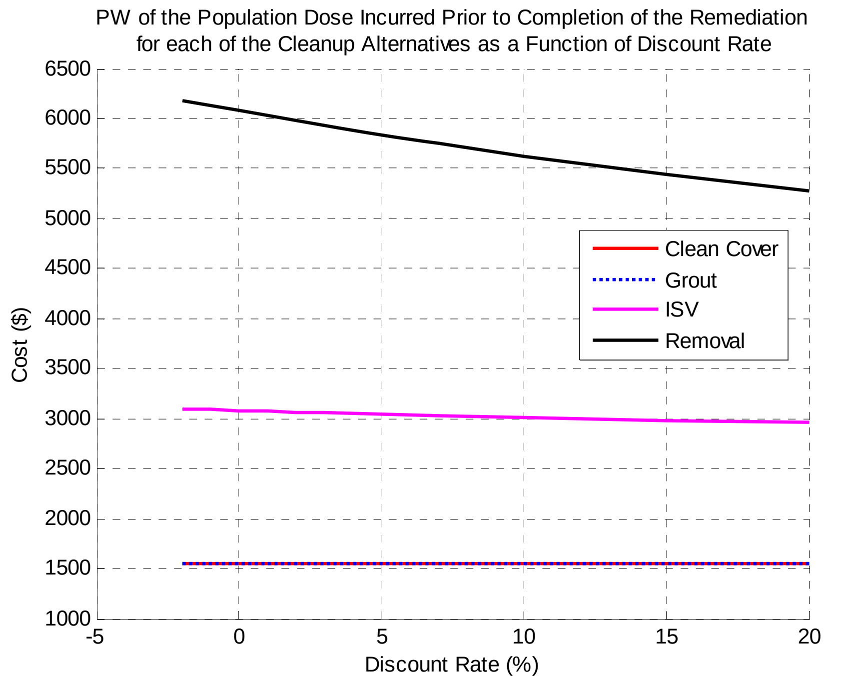


Figure 5.47 – PW of the population dose incurred prior to the completion of the remediation for each of the cleanup alternatives.

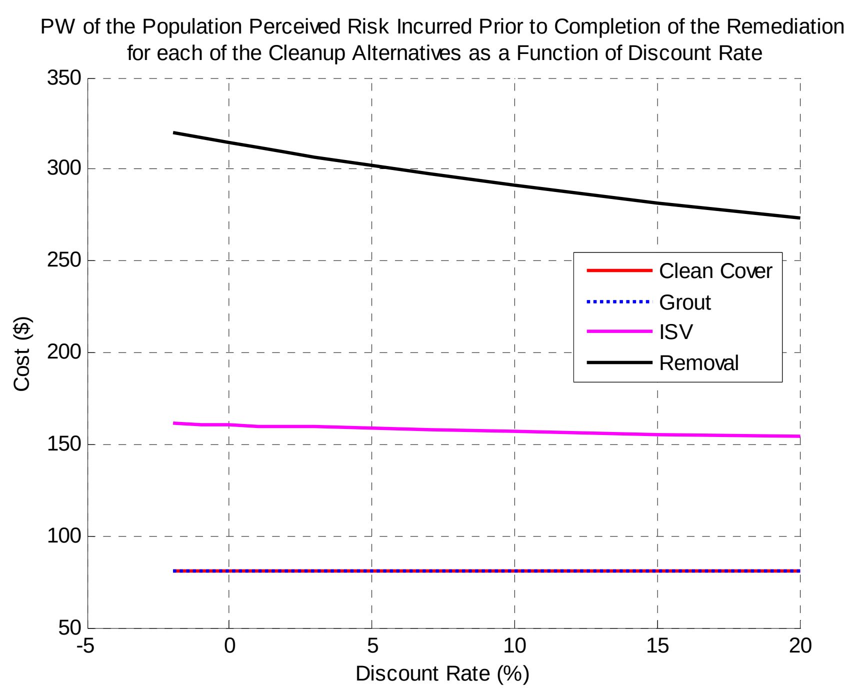


Figure 5.48 – PW of the perceived risk incurred prior to the completion of the remediation for each of the cleanup alternatives.

The dose from the surface water pathway was also assumed to be eliminated by all of the alternatives except no action, because there will be no contamination on the soil surface to be redistributed to the car park or street. Consequently the perceived risk from the surface water, external gamma, inhalation, and soil ingestion pathways was eliminated, because the fractional risk reduction was 1. According to Equations 6 and 7 the perceived risk from the groundwater pathway should double for the no action alternative and remain constant for the others, assuming no public involvement. This effect, however, is not significant because the perceived risk due to groundwater contamination is very low to begin with.

5.6 Summary of Costs for the Cleanup Alternatives

The results of this case study are compiled in Tables 5.1 and 5.2 below. Table 5.1 gives the PW of the individual costs/benefits and total cost for each of the alternatives for a discount rate of 0 %. Table 5.2 gives the PW of the total cost of each of the cleanup alternatives for all of the discount rates considered, which is plotted in Figure 5.38. From this figure and table it can be seen that the lowest cost option is clean cover if the discount rate is around 5 % or less, and, if it is more than this, the no action alternative has the lowest cost. This is because the cost of the actual cleanup alternatives is relatively constant with discount rate because the cleanups do not take very long to complete while the no action alternative is controlled entirely by the PW of the dose estimates and the higher discount rates value the future dose less decrease the overall PW. Because the discount rate used for environmental cleanups is typically low to avoid large discounting of future harms the clean cover alternative is the most likely to be selected as the best option.

Table 5.1 - PW of costs/benefits for cleanup alternatives using a 0% discount rate.

		Cleanup Alternative				
Cost or Benefit (Thousand $)	**Pathway**	No Remediation	Clean Cover	Grouting	ISV	Removal
Dose from Existing Contamination	External, Inhalation, Soil Ingestion	694				
	Groundwater	1.6				
	Surface Water	0.3				
	All	695.9				
Perceived Risk From Existing Contamination	External, Inhalation, Soil Ingestion	18				
	Groundwater	0				
	Surface Water	0				
	All	18				
Cost of Cleanup Alternative	N/A	0	227	13500	40000	53300
Worker Dose from Cleanup Alternative	N/A	0	0	8	66	133
Dose Reduction from Cleanup Alternative	External, Inhalation, Soil Ingestion	0	692	692	691	688
	Groundwater	0	0	0	0	0
	Surface Water	0	0.3	0.3	0.3	0.3
	All	0	692.3	692.3	691.3	688.3
Perceived Risk Reduction from Cleanup Alternative	External, Inhalation, Soil Ingestion	0	17.9	17.9	17.8	17.7
	Groundwater	0	0	0	0	0
	Surface Water	0	0	0	0	0
	All	0	17.9	17.9	17.8	17.7
Total Cost		713.9	230.7	13511.7	40070.8	53440.9

Table 5.2 - PW of cleanup alternatives for discount rates ranging from -2 to 20 %.

Discount Rate (%)	PW of Cost of Cleanup Alternative ($)				
	No Remediation	Clean Cover	Grouting	ISV	Removal
-2	$5.25*10^6$	$2.68*10^5$	$1.35*10^7$	$4.05*10^7$	$5.51*10^7$
-1	$1.25*10^6$	$2.33*10^5$	$1.35*10^7$	$4.03*10^7$	$5.42*10^7$
0	$7.14*10^5$	$2.31*10^5$	$1.35*10^7$	$4.01*10^7$	$5.34*10^7$
1	$5.04*10^5$	$2.30*10^5$	$1.35*10^7$	$3.99*10^7$	$5.26*10^7$
2	$3.98*10^5$	$2.30*10^5$	$1.35*10^7$	$3.97*10^7$	$5.19*10^7$
3	$3.22*10^5$	$2.30*10^5$	$1.35*10^7$	$3.95*10^7$	$5.11*10^7$
5	$2.39*10^5$	$2.29*10^5$	$1.35*10^7$	$3.91*10^7$	$4.97*10^7$
7	$1.91*10^5$	$2.29*10^5$	$1.35*10^7$	$3.88*10^7$	$4.84*10^7$
10	$1.49*10^5$	$2.29*10^5$	$1.35*10^7$	$3.82*10^7$	$4.66*10^7$
15	$1.11*10^5$	$2.29*10^5$	$1.35*10^7$	$3.75*10^7$	$4.39*10^7$
20	$8.97*10^4$	$2.29*10^5$	$1.35*10^7$	$3.67*10^7$	$4.15*10^7$

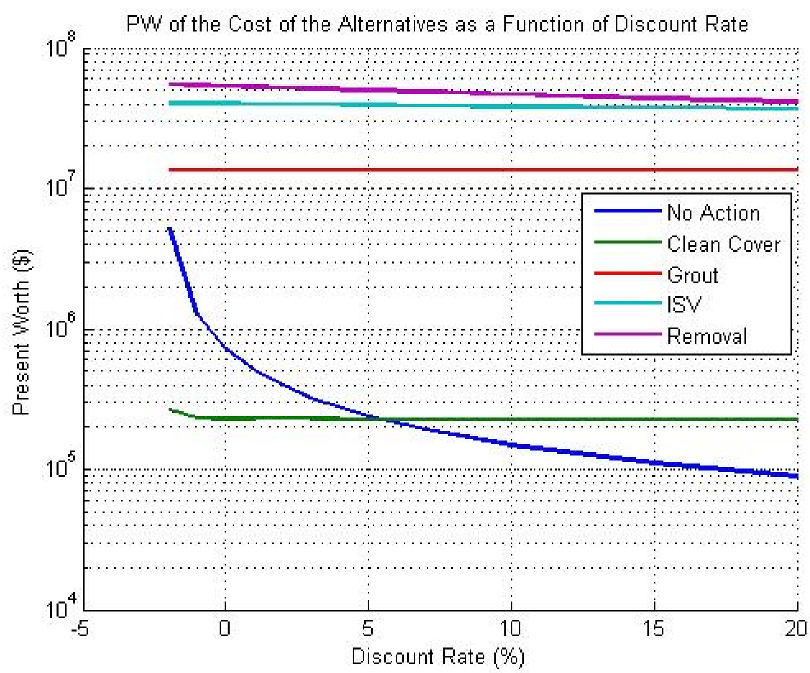


Figure 5.49 - PW of the cost of the cleanup alternatives considered as a function of discount rate.

5.7 Discussion

From Figure 5.38 it can be seen that the clean cover alternative has the lowest PW cost for discount rates around 5% or less and that for discount rates greater than this the no action alternative has the lowest PW cost. The probability of the discount rate being greater than 5 % depends on the decision maker's finances and ethics, but would be low for most decisions involving long time horizons because it would likely not be publicly acceptable to largely discount harms to future generations. The discount rate at which the decision switches from clean cover to no action would be shifted to around 10% for .5 m of clean cover; if this thickness of clean cover results in dose reduction close to the dose reduction from 1 m of clean cover. This would be significant because it is very unlikely that a discount rate greater than 10% would be selected.

There are other factors that might effect the selection of a cleanup alternative. The most important of these is whether or not the public perception for the various alternatives would be significantly different than was calculated. For instance, the clean cover and no action alternatives would probably be met with resistance by the US public for a similar contamination scenario. This would make the public perception costs for these alternatives higher, possibly making one of the other alternatives optimum. Also, if it was desired by the decision makers or the public to turn the site into a green field then the removal alternative would have a large advantage over the others because unlimited access could be granted to the site if contamination was removed and the cost of ongoing maintenance and security of the site could be avoided.

In addition to the low public perception values the low cost alternatives were found to be optimum because the published data on the contamination at the RWDS indicates it is primarily ^{137}Cs and ^{90}Sr, which have approximately 30 year half lives, and, consequently, the dose decreases rapidly with time, and the reduction in dose from the cleanup alternatives is less than it would be for longer lived nuclides. Therefore, the presence of significant quantities radionuclides at the KI RWDS other than ^{137}Cs and ^{90}Sr might also change the optimum alternative particularly if the additional contaminants have long half lives. Also these radionuclides have relatively high K_d, which limits the dose from the groundwater pathway because they decay significantly before they reach the potential well site.

The effect of longer lived radionuclides with lower K_d values was analyzed in the 2nd groundwater scenario. The effect of the additional nuclides (^{99}Tc and ^{129}I) was found to be small for the fission yield based concentrations used in the calculation. However, the dose behaved as expected and if the concentrations were higher there would have been significant dose from these nuclides for the unaltered soil site condition. Higher groundwater dose rates would reduce the attractiveness of the clean cover alternative because it does not reduce the inventory at the site or restrict access of the groundwater to the contaminants. So for radionuclides with low K_d values the groundwater pathway will be more significant and the more costly cleanup alternatives will be more attractive because they result in lower leach rates or no leach rate for the removal alternative.

Chapter 6 Conclusions and Future Work

6.1 Conclusions

A decision framework for making radioactive waste cleanup decisions capable of accounting for public perception without requiring public input has been developed. The framework utilizes multi attribute value theory to determine the best cleanup alternative. The following costs and benefits are the primary factors considered in the framework: dose, public perception, and cleanup cost. The framework utilizes a risk aversion factor published by NRPB which is dependent on the annual dose rate to the public to calculate the cost of the public perception. This risk aversion factor is also modified to calculate the public's perception of the cleanup alternatives. This framework was tested with a case study based on 2003 data for the contamination at the KI RWDS in Moscow, Russia.

Five remediation options were considered in this case study: no action, 1 m of clean cover, injecting grout into the soil at the site and covering it with 1 m of clean cover, ISV of the soil at the site with 1 m of clean cover placed over the vitrified soil, and removal of the contaminants to an offsite disposal facility. In this study the optimum cleanup method for the site was found to be covering it with clean fill 1 m thick if the appropriate discount rate is determined to be less than or equal to 5 percent and to leave the site alone if the discount rate is determined to be greater than 5 percent.

This result raises some questions about the ability of the framework to adequately characterize the public's perception of the cleanup alternatives, because the clean cover alternative is the cheapest and produces similar dose reductions as the other alternatives (excluding no action) yet it receives approximately the same perception costs as the more expensive alternatives. Refinement of the risk aversion factor or the use of another method might be necessary to overcome this problem.

6.2 Future Work

The results of the case study revealed several areas in which future work could be done to improve the decision for the case study or the framework in general. For the case study other thicknesses of clean cover could be analyzed to find an optimum thickness which would have a lower total cost than no action at discount rates greater than 5%. Also the public's perception of the cleanup alternatives needs to be reassessed because the results do not reflect what would be expected. Additional work could be done to better characterize the RWDS and determine what the effect on the decision would be if there were significant quantities of long lived radionuclides present. Research could also be done to determine the appropriate discount rate for the KI RWDS cleanup which would eliminate having two optimum cleanup alternatives.

For the framework in general, future work could include research about the VSL and risk aversion factor which were based on US and UK data, respectively. Analysis could be done to determine if these are appropriate for use in Russia and if they are to not be appropriate substitutes could be created. Also the method used to determine the public perception could be varied to include factors that account for more than just the magnitude of the risk. For instance, a factor could be added to the risk aversion factor to account for the different risk tolerances of different groups of people which would allow the framework to be more problem specific.

REFERENCES

1. Beierle, Thomas C. "The Quality of Stakeholder-Based Decisions." Risk Analysis 22.4 (2002): 739-49.

2. Flüeler, Thomas. "Options in Radioactive Waste Management Revisited: A Proposed Framework for Robust Decision Making." Risk Analysis 21.4 (2001): 787-99.

3. Piet, Steven J., et al. "Making Sustainable Decisions Using the KONVERGENCE Framework." Proceedings of Waste Management 2003, Tucson, AZ (2003).

4. Marcomini, Antonio, Glenn Walter Suter II, and Andrea Critto (Eds.). Decision Support Systems for Risk-Based Management of Contaminated Sites. New York: Springer, 2009

5. Bohnenblust, Hans, and Paul Slovic. "Integrating Technical Analysis and Public Values in Risk-Based Decision Making." Reliability Engineering and System Safety 59 (1998): 151-59.

6. Sandquist, Gary M. "Quantifying the Perceived Risks Associated with Nuclear Energy Issues." Int. J. Nuclear Energy Science and Technology 1.1 (2004): 61-8.

7. Sullivan, William G., Elin M. Wicks, and James T. Luxhoj. Engineering Economy. 12th ed. Upper Saddle River: Prentice Hall, 2003.

8. Lind, Neils. "Time Effects in Criteria for Acceptable Risk." Reliability Engineering and System Safety 78 (2002): 27-31.

9. Belzer, Richard B. "Discounting Across Generations: Necessary, not Suspect." Risk Analysis 20.6 (2000): 779-92.

10. Bohnenblust, Hans, and Paul Slovic. "Integrating Technical Analysis and Public Values in Risk-Based Decision Making." Reliability Engineering and System Safety 59 (1998): 151-59.

11. Ball, David, Megan Landon, and Tony Fletcher. Environmental Health Policy. Maidenhead, UK: Open University Press, 2006.

12. Novikov, Vladimir. (Ed.). The Nuclear Legacy in Urbanized Areas: Generic Problems and the Moscow Case Study. Laxenburg, Austria: IIASA, 2007.

13. Volkov, V. G. "Remediation of Contaminated Facilities at RRC "Kurchatov Institute"." Present at IAEA Meeting April 26-27, Stockholm, Sweden (2006).

14. NCRP. Uncertainty in NCRP Screening Models Relating to Atmospheric Transport, Deposition, and Uptake by Humans. Bethesda: NCRP, 1993.

15. Table of Nuclides. 2000. Korea Atomic Energy Research Institute. 15 April, 2009 <http://atom.kaeri.re.kr/>.

16. U.S. Nuclear Regulatory Agency. "Calculations of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I." US NRC Regulatory Guide 1.109 (1977).

17. ICRP. "Age-Dependent Doses to Members of the Public from Intake of Radionuclides Part 5, Compilation of Ingestion and Inhalation Coefficients, 72." Annals of the ICRP 26.1 (1996): 1:91.

18. Sullivan, T. M. "DUSTMS-D: Disposal Unit Source Term – Multiple Species – Distributed Failure Data Input Guide." BNL Report 75554-2006, Brookhaven National Laboratory (2006).

19. Sullivan, T. M., and C. J. Suen. "Low-Level Waste Shallow Land Disposal Source Term Model: Data Input Guides." US NRC Report NUREG/CR-5387, Brookhaven National Laboratory (1989).

20. Volkova, E., B. Iooss, and F. Van Dorpe. "Global Sensitivity Analysis for a Numerical Model of Radionuclide Migration from the RRC "Kurchatov Institute" Radwaste Disposal Site." Stoch Environ Res Risk Asses 22 (2008): 17-31.

21. Use of Probabilistic Methods in Nuclear Power Plant Decommissioning Dose Analysis. EPRI Report 1006949. Palo Alto, CA, 2002.

22. Mattigod, Shas V. "Diffusion of Iodine and Technetium-99 Through Waste Encasement Concrete and Unsaturated Soil Fill Material." Mat, Res. Soc. Symp. Proc. 824 (2004): CC7.6.1-8.

23. Gelhar, L. W., C. Welty, and K. R. Rehfeldt. "A Critical Review of Data on Field-Scale Dispersion in Aquifers." Water Resources Research. 28.7 (1992): 1955-74.

24. Thompson, L. E. "Mixed Waste Treatment Cost Analyses for a Range of Geomelt Vitrification Process Configurations." Proceedings of Waste Management 02 Conference, Tucson, AZ (2002).

25. Rastorguev, A.V. "Development of three-dimensional migration model and prognosis of Sr-90 contaminated ground water flow distribution from the RRC "Kurchatov Institute" Waste Disposal Site." Presented at the International Workshop on Cleaning up Sites Contaminated with Radioactive Materials, Moscow, Russia, June 4-6, 2007.

26. Yu, C., et al. "User's Manual for RESRAD-OFFSITE: Version 2." US DOE Report HS-0005, Argonne National Laboratory (2007).

27. Rohli, Robert V., and Anthony J. Vega. Climatology. Sudbury, MA: Jones and Bartlett Publishers, 2007.

28. Faw, Richard E., and J. Kenneth Shultis. Radiological Assessment: Sources and Doses. La Grange Park: ANS, 1999.

29. Waier, Phillip R., ed. Means Building Construction Data. 61st ed. Kingston, MA: RSMeans, 2003.

30. Andrus, Ronald D., and Riley M. Chung. "Liquefaction Remediation near Existing Lifeline Structures." Proceedings of the 6th Japan-U.S. Workshop on Earthquake Resistant Design of Lifeline Facilities and Countermeasures Against Soil Liquefaction, Tokyo, Japan, June 11-13, 1996.

31. Federal Remediation Technologies Roundtable. 14 July 2008. FRTR. 14 March 2009 <http://www.frtr.gov/>.

32. Volkov, V. G., et al. "The First Stage of Liquidation of Temporary Radwaste Repositories and Rehabilitation of the Radwaste Disposal Site at the Russian Research Center "Kurchatov Institute"." Proceedings of Waste Management 04 Conference, Tucson, AZ (2004).

33. Viscusi, W. Kip, and Joseph E. Aldy. "The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World." The Journal of Risk and Uncertainty 27.1 (2003): 5-76.

APPENDICES

Appendix A – Inputcreator Code

This appendix is a copy of the input generator code and two of its functions which were used to calculate the shape factors for the RESRAD Offsite dose calculation.

Appendix A.1 – Main Code

This is a copy of the inputcreator code used to generate the input files for the RESRAD Offsite dose calculation.

```
clear all;
clc;
% This program creates input files for use in RESRAD OFFSITE
%
% The program is capable of combining multiple contamination areas, source
% terms, and receptor locations (dwellings) into individual RESRAD input
% files.
%
% The input file names generated are named 'basename_ii_jj_kk_zz.ROF'
%
% basename = user defined base for the file series
% ii = contamination area identifier
% jj = receptor location identifier
% kk = source term identifier
% zz = age group identifier

basename = 'KI';
numtimepoints = 256;

% Read base input file to get all constant inputs this file can be edited
% to change the other inputs for all input files

ifile = fopen('resin.dat','r');
ii=1;
while feof(ifile) == 0
   P(ii) = cellstr(fgets(ifile));
   ii = ii + 1;
```

```
end
P=P';
fclose(ifile);

ifile = fopen('chn1.dat','r');
ii=1;
while feof(ifile) == 0
   Q(ii) = cellstr(fgets(ifile));
   ii = ii + 1;
end
Q=Q';
fclose(ifile);

ifile = fopen('chn2.dat','r');
ii=1;
while feof(ifile) == 0
   R(ii) = cellstr(fgets(ifile));
   ii = ii + 1;
end
R=R';
fclose(ifile);

% Define contamination and dwelling locations as well as contamination
% multiplier for non uniform contamination
%
% numC = number of contamination locations
% numD = number of dwelling locations
%
% C defines contamination location and dimensions
% C(1,i)= x coordinate of left side of ith contamination area
% C(2,i)= y coordinate of bottom of ith contamination area
% C(3,i)= x dimension of ith contamination area
% C(4,i)= y dimesion of ith contamination area
% C(5,i)= average Cs-137 in ith contamination area (survey data)
%
% D defines dwelling (receptor) location and dimensions
% D(1,i)= x coordinate of left side of ith dwelling area
% D(2,i)= y coordinate of bottom of ith dwelling area
% D(3,i)= x dimension of ith dwelling area
% D(4,i)= y dimesion of ith dwelling area

numC = 6;
numD = 72;

C = [0    21.24    68.38    68.38    102.57   0
27.46     1.56     -38.85   22.8     57.51    0
21.24     47.14    61.13    62.69    30.05    21.24
43.52     69.42    61.65    34.17    22.28    27.46
605.3503846    1291.384884    1726.845924    441.3594249    103.4213855    80.40441176] ;

CenD = [54 30] ;
```

```
D = [100	63.38834765	-25	-113.3883476	-150	-113.3883476	-25
	63.38834765	150	98.74368671	-25	-148.7436867	-200	-148.7436867
	-25	98.74368671	200	134.0990258	-25	-184.0990258	-250	-
184.0990258	-25	134.0990258	300	204.8097039	-25	-254.8097039	-350
	-254.8097039	-25	204.8097039	400	275.520382	-25	-325.520382
	-450	-325.520382	-25	275.520382	500	346.2310601	-25	-
396.2310601	-550	-396.2310601	-25	346.2310601	600	416.9417382	-25
	-466.9417382	-650	-466.9417382	-25	416.9417382	700	487.6524164
	-25	-537.6524164	-750	-537.6524164	-25	487.6524164	800
	558.3630945	-25	-608.3630945	-850	-608.3630945	-25	558.3630945
	900	629.0737726	-25	-679.0737726	-950	-679.0737726	-25
	629.0737726
-25	-113.3883476	-150	-113.3883476	-25	63.38834765	100	63.38834765
	-25	-148.7436867	-200	-148.7436867	-25	98.74368671	150
	98.74368671	-25	-184.0990258	-250	-184.0990258	-25	134.0990258
	200	134.0990258	-25	-254.8097039	-350	-254.8097039	-25
	204.8097039	300	204.8097039	-25	-325.520382	-450	-325.520382
	-25	275.520382	400	275.520382	-25	-396.2310601	-550	-
396.2310601	-25	346.2310601	500	346.2310601	-25	-466.9417382	-650
	-466.9417382	-25	416.9417382	600	416.9417382	-25	-537.6524164
	-750	-537.6524164	-25	487.6524164	700	487.6524164	-25	-
608.3630945	-850	-608.3630945	-25	558.3630945	800	558.3630945	-25
	-679.0737726	-950	-679.0737726	-25	629.0737726	900	629.0737726
50	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50
50	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50	50	50	50	50	50	50	50	50	50
	50	50];

% Define plausible source terms and the probability of each occuring
%
% numS = number of different source terms used
% numR = a vector defining the number of radionuclides in each source term,
%	n
%
% Concentration multipliers = the number which when multiplied by
%			the base Cs-137 concentration yields
%			the concentration for that source term
```

```
%
% S1(1,i)= text string defining the first radionuclide in ith source term
% S2(1,i)= concentration multiplier of the first radionuclide in ith source term
% S1(n,i)= text string defining the last radionuclide in ith source term
% S2(n,i)= concentration multiplier of the last radionuclide in ith source term

numS = 4;
numR = [2,2,2,2,2,2];

% 1
S1(1,1) = cellstr('Cs-137+D');
S1(2,1) = cellstr('Sr-90+D');
S2(1,1)=-.632455532;
S2(2,1)=-.7280109889;

% 2
S1(1,2) = cellstr('Cs-137+D');
S1(2,2) = cellstr('Sr-90+D');
S2(1,2)=0;
S2(2,2)=0;

% 3
S1(1,3) = cellstr('Cs-137+D');
S1(2,3) = cellstr('Sr-90+D');
S2(1,3)=.632455532;
S2(2,3)=.7280109889;

% 4
S1(1,4) = cellstr('Cs-137+D');
S1(2,4) = cellstr('Sr-90+D');
S2(1,4)=0;
S2(2,4)=1.5;

% Define age dependent factors inhalation rate, time spent outdoors

numAG = 3;

% inhalr = [8400];
time_id = [.5 .5 .5];
time_od = [3 1 .3]./24;

% Calculate the contamination shape factor and area fraction data required
% for input
%
% A1(1,ii)=x coordinate of the center of the ith contamination
% A1(2,ii)=y coordinate of the center of the ith contamination
% A1(3,ii)=distance to corner of contamination from its center rounded up
%         plus one (this is used in RESRAD to calculate shape factors)
%
% A2(1,jj)=x coordinate of the center of the jth dwelling
% A2(2,jj)=y coordinate of the center of the jth dwelling
```

```
%
% A3(jj,ii)=distance between the center of the jth dwelling and furthest
%          corner of the ith contaminatin rounded up plus one (used in RESRAD)

for ii=1:numC
   A1(1,ii)=C(1,ii)+C(3,ii)/2;
   A1(2,ii)=C(2,ii)+C(4,ii)/2;
   A1(3,ii)=ceil([(C(3,ii)/2)^2+(C(4,ii)/2)^2]^.5)+1;

   SF1(:,ii) = trap(C(3,ii)/2,C(4,ii)/2,A1(3,ii));
end

fprintf('The program is beginning to calculate the second set of shape factors.\n')

rr=1;
for jj=1:numD
   A2(1,jj)=D(1,jj)+D(3,jj)/2;
   A2(2,jj)=D(2,jj)+D(4,jj)/2;
   for ii=1:numC
      X=abs(A2(1,jj)+CenD(1)-A1(1,ii))+C(3,ii)/2;
      Y=abs(A2(2,jj)+CenD(2)-A1(2,ii))+C(4,ii)/2;
      A3(ii,jj)=ceil([(X)^2+(Y)^2]^.5)+1;

      SF2(:,ii,jj) = trap2(C(3,ii),C(4,ii),X,Y,A3(ii,jj));
   end

   if (floor(jj/4) == rr)
      fprintf('%.0f/%.0f of the way done\n',jj/4,numD/4)
      rr=rr+1;
   end
end

% Write RESRAD input files from program generated data
%
%

fprintf('\nStarting to write the input files\n')

f2=fopen('filelist.dat','w');
fprintf(f2,'%.0f\n',numS*numC*numD*numAG);
for zz=1:numAG
for ii=1:numS
   for jj=1:numC
      for kk=1:numD
         [ofile,ofile2]=cfile(basename,ii,jj,kk,zz);
         f1=fopen(ofile,'w');
         f3=fopen(ofile2,'w');

         if (ii < 9)
            for tt=1:40
               fprintf(f3,'%s\r\n',char(Q(tt)));
```

```
    end

else
    for tt=1:127
        fprintf(f3,'%s\r\n',char(R(tt)));
    end
end

fclose(f3);

for nn=1:14
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' NANUC = %.0f,\r\n',numR(ii));
fprintf(f1,'%s\r\n',char(P(16)));
fprintf(f1,' NPD = %.0f,\r\n',numR(ii));
fprintf(f1,' NPTS = %.0f,\r\n',numtimepoints);
fprintf(f1,' NS = 0,\r\n');
fprintf(f1,' NPDS = 2 ,\r\n',numR(ii));
for nn=21:26
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' NPTS = %.0f,\r\n',numtimepoints);

for nn=28:35
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' SOURCEXY(1) =%7.2f,\r\n',C(3,jj));
fprintf(f1,' SOURCEXY(2) =%7.2f,\r\n',C(4,jj));
for nn=38:53
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' DWELLXY(1) =%7.2f,\r\n',D(1,kk)+CenD(1)-C(1,jj));
fprintf(f1,' DWELLXY(2) =%7.2f,\r\n',D(1,kk)+D(3,kk)+CenD(1)-C(1,jj)) ;
fprintf(f1,' DWELLXY(3) =%7.2f,\r\n',D(2,kk)+CenD(2)-C(2,jj));
fprintf(f1,' DWELLXY(4) =%7.2f,\r\n',D(2,kk)+D(4,kk)+CenD(2)-C(2,jj));
for nn=58:83
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' AREA =%8.3f,\r\n',C(3,jj)*C(4,jj)) ;
for nn=85:112
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' AREAODWELL =%7.2f,\r\n',D(3,kk)*D(4,kk));
for nn=114:927
    fprintf(f1,'%s\r\n',char(P(nn)));
end
fprintf(f1,' INHALR = 8400,\r\n');
for nn=929:933
    fprintf(f1,'%s\r\n',char(P(nn)));
end
```

```
          for mm=1:12
             fprintf(f1,' RAD_SHAPE(%.0f) = %.4f,\r\n',mm,mm*A1(3,jj)/12);
          end
          for mm=13:24
             fprintf(f1,' RAD_SHAPE(%.0f) = %.4f,\r\n',(mm-12)*A3(jj,kk)/12);
          end
          for mm=1:12
             fprintf(f1,' FRACA(%.0f) =%6.3f,\r\n',mm,SF1(mm,jj));
          end
          for mm=13:24
             fprintf(f1,' FRACA(%.0f) =%6.3f,\r\n',mm,SF2(mm-12,jj,kk));
          end

          fprintf(f1,' FIND = 0,\r\n');
          fprintf(f1,' FOTD = 0,\r\n');
          fprintf(f1,' FINDDWELL = %.3f,\r\n',time_id(zz));
          fprintf(f1,' FOTDDWELL = %.3f,\r\n',time_od(zz));

          for nn=986:1023
             fprintf(f1,'%s\r\n',char(P(nn)));
          end

          fprintf(f1,' NUCNAM = ');
          for mm=1:numR(ii)
             fprintf(f1,'"%s", ',char(S1(mm,ii)));
          end
          fprintf(f1,'"LAST",\r\n');

          fprintf(f1,' S =');
          for mm=1:numR(ii)
             fprintf(f1,' %G,',(1+S2(mm,ii))*C(5,jj)/(5^(mm-1)));
          end
          fprintf(f1,'\r\n');

          for nn=1026:1074
             fprintf(f1,'%s\r\n',char(P(nn)));
          end

          fprintf(f2,'%s\n',ofile);
          fclose(f1);
       end
    end
end
   fprintf('%.0f/%.0f of the way done\n',zz,numAG)
end
fclose(f2);
```

Appendix A.2 – Shape Factor 1 Calculation Subroutine

This is the code used to calculate the first set of shape factors for the RESRAD Offsite dose calculation.

```
function SF = trap(x,y,L)
step=L/12;
syms t

var = floor(y/step);

sum=eval(int(((step)^2-t^2)^.5,0,min(step,x)));
SF(1) = 4*sum/(pi*step^2);

for jj=2:12
   sum=0;
   if (jj <= var)
      sum=eval(int(((step*jj)^2-t^2)^.5-((step*(jj-1))^2-t^2)^.5,0,min(step*(jj-1),x)));
      if (x > step*(jj-1))
         sum=sum + eval(int(((step*jj)^2-t^2)^.5,step*(jj-1),min(step*jj,x)));
      end
   else
      x2 = (-(y^2)+(step*(jj))^2)^.5;

      if (y < step*jj & y > step*(jj-1))
         sum=eval(int(y-((step*(jj-1))^2-t^2)^.5,0,min(x2,x)));
         if (x > x2)
            sum=sum + eval(int(((step*jj)^2-t^2)^.5-((step*(jj-1))^2-t^2)^.5,x2,min(step*(jj-1),x)));
            if (x > step*(jj-1))
            sum=sum + eval(int(((step*jj)^2-t^2)^.5,step*(jj-1),min(step*jj,x)));
            end
         end
      elseif (y < step*jj & y <= step*(jj-1))
         x1 = (-(y^2)+(step*(jj-1))^2)^.5;
         sum=eval(int(y-((step*(jj-1))^2-t^2)^.5,x1,min(x2,x)));
         if (x > x2)
            sum=sum + eval(int(((step*jj)^2-t^2)^.5-((step*(jj-1))^2-t^2)^.5,x2,min(step*(jj-1),x)));
            if (x > step*(jj-1))
            sum=sum + eval(int(((step*jj)^2-t^2)^.5,step*(jj-1),min(step*jj,x)));
            end
         end
      end
   end

   SF(jj) = 4*sum/(pi*step^2*(jj^2-(jj-1)^2));
end
```

Appendix A.3 – Shape Factor 2 Calculation Subroutine

This is the code used to calculate the second set of shape factors for the RESRAD Offsite dose calculation.

```
function SF = trap2(x4,y4,x3,y3,L)
step=L/12;
step2=L/10000;
h(2)=0;

if (y3-y4 < 0)
   y1(2)=x3;
   y2(2)=x3-x4;
   x2(2)=-y3+y4;
   r=y2(2);

   x1=[-y3:step2:y3+y4+step2];
else
   y1(2)=y3;
   y2(2)=y3-y4;
   x2(2)=x3;
   r=(y2(2)^2+(x3-x4)^2)^.5;

   x1=[x3-x4:step2:x3+step2];
end

for ii=1:12
   if (ii*step <= r)
      sum=0;
   else
      kk=1;
      sum=0;
      F1=@(t)((step*ii)^2-t^2)^.5;
      F2=@(t)((step*(ii-1))^2-t^2)^.5;

      x2(1)=ii*step;

      while (x1(kk)<min(x2))
         y1(1)=(F1(x1(kk))+F1(x1(kk+1)))/2;
         y2(1)=(F2(x1(kk))+F2(x1(kk+1)))/2;

         if (imag(y1(1)) ~= 0)
            y1(1)=0;
         end

         if (imag(y2(1)) ~= 0)
```

```
            y2(1)=0;
        end

        h(1)=(min(y1)-max(y2));

        sum=sum+step2*max(h);
        kk=kk+1;
    end
end

SF(ii) = sum /(pi*step^2*(ii^2-(ii-1)^2));

end
```

Appendix B – DUST-MS Input Files for Scenario 2

This appendix is a copy of the input files used to calculate the leach rate from the soil, grouted soil, and ISV soil; and to model the flow of these contaminants from the site to the point at which a well was assumed to have been dug for the fission yield based concentration data.

Appendix B.1 – Unaltered Soil Leach Calculation

This is a copy of the file used to model the contaminant flux from the soil scenario into the aquifer.

```
              Data Set 1: General Parameters
Kurchatov Institute Groundwater Input
NISO IACT     3    1
NNP ITRANS   21    1
RN Prop   SR-90      2.879e+01 1.000e+02 8.991e+01
RN Prop   TC-99      2.110e+05 1.000e+02 9.891e+01
RN Prop   I-129      1.570e+07 2.280e-07 1.289e+02
Decay Chn     0

              Data Set 2: Time Parameters
NTI DTCHG  1000    1
TIME STEP  1.000e+00 0.000e+00 1.000e+00 1.000e+03     2003.
TIME DTCHG 0.000e+00

              Data Set 3: Material Parameters
NMAT NCM      1    0
         K-d       DENSITY   DISP.     DIFFUSION
Sr-90     5.100E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1
Tc-99     1.000E-01 1.800E+00 7.000E+01 1.000E-05 Material #  1
I-129     1.000E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1

              Data Set 4: Output Control Parameters
Print Control Parameters for each time step (NTI)
```

```
000000000300000000030000000000000000000000000000030000000000000000000000
0000000000000000000000000000030000000000000000000000000000000000000000000
0000000003000000000000000000000000000000000000000000000000030000000000
0000000000000000000000000000000000000003000000000000000000000000000000
0000000000000000000300000000000000000000000000000000000000000000000003
0000000000000000000000000000000000000000000000000300000000000000000000
0000000000000000000000000000030000000000000000000000000000000000000000
0000000003000000000000000000000000000000000000000000000000030000000000
0000000000000000000000000000000000000003000000000000000000000000000000
0000000000000000000300000000000000000000000000000000000000000000000003
0000000000000000000000000000000000000000000000000300000000000000000000
0000000000000000000000000000030000000000000000000000000000000000000000
0000000003000000000000000000000000000000000000000000000000030000000000
0000000000000000000000000000000000000003000000000000000000000000000000
00000000000000000003
Trace var    1    1    1
Con Tr Loc   16
FX Tr Loc    16
```

Data Set 5: Facility Coordinate Data

```
Area      1.136E+08
Delta X      1    5    1      0.000E+00 2.000E+01 0.000E+00
Delta X      6   10    1      1.000E+02 5.000E+01 0.000E+00
Delta X     16    6    1      5.700E+02 2.000E+01 0.000E+00
```

Data Set 6: Initial and Boundary Conditions

```
SR 90 INV     1   21    1      0.000E+00 0.000E+00

Tc 99 INV     1   21    1      0.000E+00 0.000E+00

I 129 INV     1   21    1      0.000E+00 0.000E+00

BC FLAGS      2    1    2         SR 90
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    concentration

BC FLAGS      2    1    2         TC 99
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    concentration

BC FLAGS      2    1    2         I 129
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00    concentration
```

Data Set 7: Water Flow Parameters

Vel Pts 2
TIME 0.000e+00 1.000e+03
VALUE 7.605e-07 7.605e-07
MST CONT 1 21 1 1.400E-01 0.000E+00

Data Set 8: Container Parameters

NCON TYPE 10 1
TIME FAIL 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00
TIME FAIL 0.000E+00 0.000E+00 0.000E+00
FAIL FLAG 0
Con Loc 6 7 8 9 10 11 12 13 14 15

Data Set 9: Wasteform Leaching Parameters

WF Type 1
WASTE FORM: FLAG HLF WDTH HLF HGHT VOLUME
Shape Size 0 5.329E+03 2.500E+02 5.680E+10 WF type #1

RLSE DATA: SURF FRCT DIFF FRCT PART CO. DIFF CO. FR RLSRATE
Sr-90 1.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 WF # 1
Tc-99 1.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 WF # 1
I-129 1.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 WF # 1

Sr-90 INV 3.533E+01 3.533E+01 3.533E+01 3.533E+01 3.533E+01 3.533E+01 3.533E+01
Sr-90 INV 3.533E+01 3.533E+01 3.533E+01

Tc-99 INV 1.663E-02 1.663E-02 1.663E-02 1.663E-02 1.663E-02 1.663E-02 1.663E-02
Tc-99 INV 1.663E-02 1.663E-02 1.663E-02

I-129 INV 2.627E-05 2.627E-05 2.627E-05 2.627E-05 2.627E-05 2.627E-05 2.627E-05
I-129 INV 2.627E-05 2.627E-05 2.627E-05

Data Set 10: External Sources (F.D. model only)

SR 90 SRC 0 0 0

TC 99 SRC 0 0 0

I 129 SRC 0 0 0

Appendix B.2 – Grouted Soil Leach Calculation

This is a copy of the file used to model the contaminant flux from the grouted soil scenario into the aquifer. The results from this file were multiplied by ten because only one tenth of the contamination was modeled for this scenario.

```
              Data Set 1: General Parameters
Kurchatov Institute Groundwater Input
NISO IACT     3    1
NNP ITRANS   21    1
RN Prop   SR-90      2.879e+01 1.000e+02 8.991e+01
RN Prop   TC-99      2.110e+05 1.000e+02 9.891e+01
RN Prop   I-129      1.570e+07 2.280e-07 1.289e+02
Decay Chn     0

              Data Set 2: Time Parameters
NTI DTCHG  1000    1
TIME STEP  1.000e+00 0.000e+00 1.000e+00 1.000e+03     2003.
TIME DTCHG 0.000e+00

              Data Set 3: Material Parameters
NMAT NCM      1    0
         K-d       DENSITY   DISP.     DIFFUSION
Sr-90     5.100E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1
Tc-99     1.000E-01 1.800E+00 7.000E+01 1.000E-05 Material #  1
I-129     1.000E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1

              Data Set 4: Output Control Parameters
Print Control Parameters for each time step (NTI)
         0000000003000000000300000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         00000000000000000003
Trace var     1    1    1
```

```
Con Tr Loc   16
FX Tr Loc    16

            Data Set 5: Facility Coordinate Data
Area        1.136E+08
Delta X       1    5    1      0.000E+00 2.000E+01 0.000E+00
Delta X       6   10    1      1.000E+02 5.000E+01 0.000E+00
Delta X      16    6    1      5.700E+02 2.000E+01 0.000E+00


            Data Set 6: Initial and Boundary Conditions
SR 90 INV     1   21    1      0.000E+00 0.000E+00

Tc 99 INV     1   21    1      0.000E+00 0.000E+00

I 129 INV     1   21    1      0.000E+00 0.000E+00

BC FLAGS      2    1    2         SR 90
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     concentration

BC FLAGS      2    1    2         TC 99
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     concentration

BC FLAGS      2    1    2         I 129
Time Top   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000E+03
VALUE      0.000E+00 0.000E+00     concentration


            Data Set 7: Water Flow Parameters
Vel Pts       2
TIME       0.000e+00 1.000e+03
VALUE      7.605e-07 7.605e-07
MST CONT      1   21    1      1.400E-01 0.000E+00


            Data Set 8: Container Parameters
NCON TYPE    10    1
TIME FAIL  0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00
TIME FAIL  0.000E+00 0.000E+00 0.000E+00
FAIL FLAG     0
Con Loc       6    7    8    9   10   11   12   13   14   15

            Data Set 9: Wasteform Leaching Parameters
```

WF Type 1
WASTE FORM: FLAG HLF WDTH HLF HGHT VOLUME
Shape Size 3 1.685E+03 2.500E+02 5.680E+09 WF type #1

RLSE DATA: SURF FRCT DIFF FRCT PART CO. DIFF CO. FR RLSRATE
Sr-90 0.000E+00 1.000E+00 0.000E+00 5.200E-10 0.000E+00 WF # 1
Tc-99 0.000E+00 1.000E+00 0.000E+00 8.000E-12 0.000E+00 WF # 1
I-129 0.000E+00 1.000E+00 0.000E+00 1.300E-12 0.000E+00 WF # 1

Sr-90 INV 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00
Sr-90 INV 3.533E+00 3.533E+00 3.533E+00

Tc-99 INV 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03
Tc-99 INV 1.663E-03 1.663E-03 1.663E-03

I-129 INV 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06
I-129 INV 2.627E-06 2.627E-06 2.627E-06

Data Set 10: External Sources (F.D. model only)
SR 90 SRC 0 0 0

TC 99 SRC 0 0 0

I 129 SRC 0 0 0

Appendix B.3 – ISV Soil Leach Calculation

This is a copy of the file used to model the contaminant flux from the ISV soil scenario into the aquifer. The results from this file were multiplied by ten because only one tenth of the contamination was modeled for this scenario.

```
              Data Set 1: General Parameters
Kurchatov Institute Groundwater Input
NISO IACT     3    1
NNP ITRANS   21    1
RN Prop   SR-90      2.879e+01 1.000e+02 8.991e+01
RN Prop   TC-99      2.110e+05 1.000e+02 9.891e+01
RN Prop   I-129      1.570e+07 2.280e-07 1.289e+02
Decay Chn     0

              Data Set 2: Time Parameters
NTI DTCHG  1000    1
TIME STEP  1.000e+00 0.000e+00 1.000e+00 1.000e+03     2003.
TIME DTCHG 0.000e+00

              Data Set 3: Material Parameters
NMAT NCM      1    0
         K-d       DENSITY   DISP.     DIFFUSION
Sr-90     5.100E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1
Tc-99     1.000E-01 1.800E+00 7.000E+01 1.000E-05 Material #  1
I-129     1.000E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1

              Data Set 4: Output Control Parameters
Print Control Parameters for each time step (NTI)
         0000000003000000000300000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         00000000000000000003
Trace var     1    1    1
```

Con Tr Loc 16
FX Tr Loc 16

Data Set 5: Facility Coordinate Data
Area 1.136E+08
Delta X 1 5 1 0.000E+00 2.000E+01 0.000E+00
Delta X 6 10 1 1.000E+02 5.000E+01 0.000E+00
Delta X 16 6 1 5.700E+02 2.000E+01 0.000E+00

Data Set 6: Initial and Boundary Conditions
SR 90 INV 1 21 1 0.000E+00 0.000E+00

Tc 99 INV 1 21 1 0.000E+00 0.000E+00

I 129 INV 1 21 1 0.000E+00 0.000E+00

BC FLAGS 2 1 2 SR 90
Time Top 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 flux
Time Bot 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 concentration

BC FLAGS 2 1 2 TC 99
Time Top 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 flux
Time Bot 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 concentration

BC FLAGS 2 1 2 I 129
Time Top 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 flux
Time Bot 0.000E+00 1.000E+03
VALUE 0.000E+00 0.000E+00 concentration

Data Set 7: Water Flow Parameters
Vel Pts 2
TIME 0.000e+00 1.000e+03
VALUE 7.605e-07 7.605e-07
MST CONT 1 21 1 1.400E-01 0.000E+00

Data Set 8: Container Parameters
NCON TYPE 10 1
TIME FAIL 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00 0.000E+00
TIME FAIL 0.000E+00 0.000E+00 0.000E+00
FAIL FLAG 0
Con Loc 6 7 8 9 10 11 12 13 14 15

Data Set 9: Wasteform Leaching Parameters

WF Type 1
WASTE FORM: FLAG HLF WDTH HLF HGHT VOLUME
Shape Size 3 1.685E+03 2.500E+02 5.680E+09 WF type #1

RLSE DATA: SURF FRCT DIFF FRCT PART CO. DIFF CO. FR RLSRATE
Sr-90 0.000E+00 0.000E+00 0.000E+00 0.000E+00 5.312E-06 WF # 1
Tc-99 0.000E+00 0.000E+00 0.000E+00 0.000E+00 5.312E-06 WF # 1
I-129 0.000E+00 0.000E+00 0.000E+00 0.000E+00 5.312E-06 WF # 1

Sr-90 INV 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00 3.533E+00
Sr-90 INV 3.533E+00 3.533E+00 3.533E+00

Tc-99 INV 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03 1.663E-03
Tc-99 INV 1.663E-03 1.663E-03 1.663E-03

I-129 INV 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06 2.627E-06
I-129 INV 2.627E-06 2.627E-06 2.627E-06

Data Set 10: External Sources (F.D. model only)
SR 90 SRC 0 0 0

TC 99 SRC 0 0 0

I 129 SRC 0 0 0

Appendix B.4 – Transport of Radionuclides in the Aquifer

This is a copy of the file used to model the contaminants movement in the aquifer from the site to the receptor location.

```
              Data Set 1: General Parameters
Kurchatov Institute Groundwater Input
NISO IACT     3    1
NNP ITRANS  121    1
RN Prop   SR-90      2.879e+01 1.000e+02 8.991e+01
RN Prop   TC-99      2.110e+05 1.000e+02 9.891e+01
RN Prop   I-129      1.570e+07 2.280e-07 1.289e+02
Decay Chn     0

              Data Set 2: Time Parameters
NTI DTCHG  1000    1
TIME STEP  1.000e+00 0.000e+00 1.000e+00 1.000e+03     2003.
TIME DTCHG 0.000e+00

              Data Set 3: Material Parameters
NMAT NCM      1    0
         K-d       DENSITY   DISP.     DIFFUSION
Sr-90     5.100E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1
Tc-99     1.000E-01 1.800E+00 7.000E+01 1.000E-05 Material #  1
I-129     1.000E+00 1.800E+00 7.000E+01 1.000E-05 Material #  1

              Data Set 4: Output Control Parameters
Print Control Parameters for each time step (NTI)
         0000000003000000000300000000000000000000000000000030000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         0000000000000000000300000000000000000000000000000000000000000000000003
         0000000000000000000000000000000000000000000000000300000000000000000000
         0000000000000000000000000000030000000000000000000000000000000000000000
         0000000003000000000000000000000000000000000000000000000000030000000000
         0000000000000000000000000000000000000003000000000000000000000000000000
         00000000000000000003
Trace var     4    4    1
Con Tr Loc   30   70   80   90
FX Tr Loc    30   70   80   90
```

Data Set 5: Facility Coordinate Data

```
Area       1.200E+08
Delta X      1   20    1      0.000E+00 2.100E+02 0.000E+00
Delta X     21    5    1      4.400E+03 2.000E+02 0.000E+00
Delta X     26    5    1      5.380E+03 1.800E+02 0.000E+00
Delta X     31   86    1      6.300E+03 2.000E+02 0.000E+00
Delta X    117    5    1      2.340E+04 1.000E+02 0.000E+00
```

Data Set 6: Initial and Boundary Conditions

```
SR 90 INV    1   20    1      8.108E-13 0.000E+00
SR 90 INV   21    5    1      2.703E-13 0.000E+00
SR 90 INV   26    5    1      1.351E-13 0.000E+00
SR 90 INV   31   86    1      2.703E-14 0.000E+00
SR 90 INV  117    5    1      0.000E+00 0.000E+00

TC 99 INV    1  121    1      0.000E+00 0.000E+00

I 129 INV    1  121    1      0.000E+00 0.000E+00

BC FLAGS     2    1    2    1      SR 90
Time Top   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     concentration

BC FLAGS     2    1    2    1      TC 99
Time Top   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     concentration

BC FLAGS     2    1    2    1      I 129
Time Top   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     flux
Time Bot   0.000E+00 1.000e+03
VALUE      0.000E+00 0.000E+00     concentration
```

Data Set 7: Water Flow Parameters

```
Vel Pts      2
TIME       0.000e+00 1.000e+03
VALUE      1.096E-05 1.096E-05
MST CONT     1  121    1      3.500E-01 0.000E+00
```

Data Set 8: Container Parameters

```
NCON TYPE    1    1
TIME FAIL  0.000E+00
FAIL FLAG    0
```

Con Loc 1

Data Set 9: Wasteform Leaching Parameters

WF Type 1

WASTE FORM: FLAG HLF WDTH HLF HGHT VOLUME

Shape Size 0 5.329E+03 2.500E+02 5.680E+10 WF type #1

RLSE DATA: SURF FRCT DIFF FRCT PART CO. DIFF CO. FR RLSRATE

Sr-90 0.000E+00 0.000E+00 0.000E+00 0.000E+00 1.000E-09 WF # 1

Tc-99 0.000E+00 0.000E+00 0.000E+00 0.000E+00 1.000E-09 WF # 1

I-129 0.000E+00 0.000E+00 0.000E+00 0.000E+00 1.000E-09 WF # 1

Sr-90 INV 1.000E-20

Tc-99 INV 1.000E-20

I-129 INV 1.000E-20

Data Set 10: External Sources (F.D. model only)

SR 90 SRC 0 0 0

TC 99 SRC 0 0 0

I 129 SRC 0 0 0