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A COMPARISON OF REMEDIATION PRIORITIES DEVELOPED BY THE DEFENSE PRIORITY MODEL, THE RELATIVE RISK EVALUATION METHOD, AND A QUANTITATIVE RISK ASSESSMENT APPROACH

THESIS

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THESIS

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Presented to the Faculty of the School of Engineering of the

Air Force Institute of Technology

Air University

In Partial Fulfillment of the

Requirements for the Degree of

Master of Science in Engineering and Environmental Management

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Acknowledgments

First of all, I am grateful to my wife, Jama, and our children, for their patience and support during this thesis effort. We have been continually blessed during our stay in Dayton the last 18 months.

I especially thank my thesis committee members for their time and support this past year. The expertise and encouragement shared by my thesis advisor, Lt Col Mike Shelley, and my readers, Maj Ed Heyse and Professor Dan Reynolds was appreciated. Though this effort included some trying periods, this committee made it a valuable and rewarding learning experience.

Finally, I wish to thank Mr Bob Moore, Chief of Restoration, Headquarters, Air Combat Command, for sponsoring a data gathering trip to Langley Air Force Base. His staff provided valuable assistance during my visit that is deeply appreciated.

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Abstract

The Superfund, established by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980, seriously underestimated both the number of severely contaminated sites and the associated cleanup cost. The magnitude of projected cleanup costs, coupled with shrinking federal budgets, necessitated the development and use of risk-based prioritization models among some federal agencies.

Among these tools, the DOD prioritization models are meant to give priority to sites posing the greatest threat to human health. Their failure to properly rank sites could incorrectly shift the focus from those that pose substantial risk to sites of lesser risk.

The DOD site ranking models addressed in this study are the Relative Risk Evaluation Method (RREM), and its predecessor, the Defense Priority Model (DPM). RREM's site evaluation approach uses a quantitative assessment of contaminants and a qualitative assessment of both pathways and receptors to group sites into a low, medium, or high risk category. The RREM has been criticized as relying too much upon qualitative factors that could, perhaps, create inconsistencies between users of the model.

The DPM, not used after 1993, used a combination of quantitative and qualitative approximations to calculate pathway subscores combined to provide an overall site score from 0 to 100. One criticism is that some of DPM's models appear theoretically weak.

The problem is that these two DOD approaches for prioritizing sites for remediation have not been validated. The research objective was as an initial validation effort for the RREM and the DPM models by comparing their rankings of a sample of contaminated sites against those of a rigorous, quantitative risk assessment model.

The quantitative model used was the American Petroleum Institute Risk/Exposure Assessment Decision Support System (DSS), software designed to assess risk associated with exposure to petroleum contaminated sites. Site specific data obtained from study reports for 15 petroleum contaminated sites was entered into DSS. DSS produced cancer risk assessments for each site, allowing ranking and comparison with the DOD models.

For this sample of sites, the moderate to strong correlation between DPM and the validation model suggested that DPM captures some key elements of a quantitative approach. However, no to weak correlation between RREM and the validation model suggested RREM rankings are questionable for this sample.

RREM's said benefit over DPM is that it is easily understood by stakeholders in the restoration process. A drawback, however, is that the quality of site ranking assessments may suffer, causing improper resource allocation. A better approach might be to use DPM but make it more user friendly.

A Comparison of Remediation Priorities Developed by the Defense Priority Model, the Relative Risk Evaluation Method, and a Quantitative Risk Assessment Approach

1.0 Introduction

1.1 General Issue

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980 initially established a $1.6 billion Superfund program to assess hazardous-waste sites, determine those responsible for site contamination, and provide financing for cleanups when necessary [13:29]. Over the past decade or so, it has become apparent that the original assumption of just a few sites needing remediation was incorrect. During the late 1970's the Environmental Protection Agency (EPA) reported hazardous contamination at only around 400 sites; they now expect the National Priority List (NPL), a list of the most severely contaminated sites, to soon reach 2000 in number. In the early 1980s, the average cost of cleaning up an NPL site was estimated at $3.6 million. In 1990, EPA reported an average cost of $26 million per NPL site, translating to a total cost of around $27 billion [13:29]. Other cost estimates place the cost of the EPA program alone at closer to $100 billion [13:30]. The shear magnitude of the projected cleanup cost has necessitated the development of risk-based prioritization models among some of the federal agencies. The capabilities of these models as adequate tools for site ranking has been questioned [13:135-176;17:1-2].

Both the EPA and the Department of Defense (DOD) prioritization models are professed to be "worst-first" ranking methods, giving priority to sites posing the greatest potential for human health threat. In general, the models try to capture the key elements of a site-specific risk assessment using only the initial, somewhat limited site investigation data. Their function is to act as "predictors" of what might be verified by a complete risk assessment of the site. A complete, site-specific risk assessment, at an expense of hundreds of thousands of dollars or more, is not feasible at every site. Therefore, the prioritization models are meant to indicate sites in greatest need of further analysis/cleanup efforts. Their failure in this purpose could incorrectly shift the focus from sites that pose substantial health risks toward those of lesser risk.

1.2 DOD Remediation Prioritization Models

The *Defense Priority Model* (DPM) uses a combination of quantitative data and qualitative approximations and calculates subscores for adverse effects on human and ecological receptors from surface water, groundwater, air and soil pathways. It then combines them into an overall numeric site score, from 0 to 100, intended to represent the relative potential threat that a contaminated site poses [14:2]. Information called for by the DPM when it considers potential receptors, for example, includes the population size within a certain radius of the contaminated site (e.g., 0.5, 1, 2, and 4 miles). DPM also calls for various geologic parameter estimates from which the model makes qualitative predictions on the migration potential of contamination.

The stated objective of the DPM was "to assist decision makers in identifying priorities for remedial action and to aid in future year budget development [14:xiii]." Because the DPM does not evaluate all factors related to risk and unrealistically high or low DPM scores could be assigned to a site, the DPM score was to be considered along with additional management concerns, such as regulatory agreements, in establishing cleanup priorities [14:xiii].

However, the DPM was criticized in a recent National Research Council (NRC) report [13:135-176]. The NRC's Committee on Remedial Action Priorities for Hazardous Waste Sites was asked to examine the principle ranking methods being used by the federal agencies to rank contaminated sites for cleanup. This study effort began around 1991. The DOD, one of the project supporters, requested and received an interim report (in 1992) assessing the DPM. Some of the criticisms of DPM included the fact that 65% of the 284 sites evaluated in 1991 had DPM scores between 13 and 37, with such a narrow interval suggesting that DPM's ability to discriminate between sites may be limited [13:9]. Another criticism was that some of the fate and transport algorithms, toxicologic and exposure assumptions, and methods embedded in the DPM have weak theoretical foundations. Also, it was brought out that DPM had never been validated [13:174]. Prior to the release of final NRC comments, the DOD announced that the DPM would no longer be used as a site ranking tool.

The *Relative Risk Evaluation Method* (RREM) replaced the DPM as the site ranking tool for the DOD. The RREM uses site-specific information regarding contaminants and their toxicity, migration pathways, and potential receptors to group sites into High, Medium, and Low relative risk categories. Pathways considered include groundwater, surface water and sediment, and surface soils. The RREM does not consider the air pathway.

Evaluation of site contaminants under the RREM is somewhat quantitative in nature, whereas potential pathways and receptors are only addressed qualitatively [15:5]. Also, the RREM provides less information, in a sense, as it results in the placement of sites into groups (high, medium, and low risk) rather than providing for complete ranking of the sites as the DPM allows.

Under the DPM, a groundwater pathway score intended to indicate contaminant migration potential is developed by qualitative use of various concepts of contaminant transport and fate. For instance, hydraulic conductivity of the vadose zone, infiltration potential, and various other factors affecting contaminant migration potential into the groundwater are "scored" (as 0, 1, 2 or 3) depending on the range of established values they fall in. These scores are combined and manipulated to establish a groundwater pathway score, that in-turn becomes an input into the final calculation of a site score [14:85].

The RREM takes a different approach with the groundwater migration pathway factor characterized as "evident," as "potential," or as "confined," based on the judgment of the site evaluator(s). This subjective rating is factored into the final risk category (low, medium, or high) for groundwater exposure[15:26].

However, this type of qualitative approach is generally criticized by Garetz (1993), a member of the Center for Environmental Statistics Development Staff, EPA, as promoting "inflationary scoring" that can produce rankings that are inequitable with regard to allocation of restoration funds. He suggests that such an approach "has no limits to the manipulation possible [5:28-29]." Further support to this argument is provided in the NRC (1994) report stating that the disadvantage of non-scientifically based approaches is that "... places with the largest political influence ... are likely to receive a share [of funding for restoration efforts] disproportionate by nonpolitical criteria [13:47]."

In addition, a few among many specific criticisms aimed directly at the RREM include: 1) the framework generally relies too heavily on qualitative factors; 2) many of its definitions are unclear, such as how it defines a receptor; and 3) it is unclear how much rigor will be put into the analysis [18:1]. One overall recommendation included that "... the process define a more quantitative, rigorous approach to characterizing sites; and, that more variables (pathways and receptors) be included [18:1]."

1.3 Quantitative Risk Assessment

Fundamentally different than both the RREM or the DPM are the tools employed in strictly quantitative, rigorous risk assessment. Characteristics of such an approach are precise determination of pathways, rigorous contaminant fate and transport modeling, and quantitative assessment of risk for each chemical [16]. Specific guidance for the quantitative risk assessment approach, including suggested exposure scenarios, can be found in the EPA's Risk Assessment Guidance for Superfund (RAGS), Volume 1 (Dec 89), and the Superfund Exposure Assessment Manual (Apr 88).

1.4 Research Problem

As suggested by reviewers of current prioritization approaches, present models for ranking contaminated sites for remediation are disputable. The problem, therefore, is that the current DOD approach, the RREM, and its predecessor, the DPM, have not been validated.

1.5 Research Objective

This objective of this research is to serve as an initial validation effort for the RREM and the DPM models by comparing their rankings of a sample of contaminated sites against those established using a rigorous, quantitative risk assessment model to evaluate each site.

1.6 Scope of Study

The NRC (1994) report suggested the following approach as a DPM validation effort: "An appropriate validation study -- comparing model results with what they should be -- would involve perhaps 10 to 30 sites and the comparison of scores and rankings from the DPM with those from another approach, assumed *a priori* more likely to yield the right answer . . . [13:167]." This is the general method planned, with the quantitative risk assessment approach assumed beforehand most likely to be a better indicator of site risk.

Naturally, this study effort will consider only those contaminated sites that have been evaluated under both the RREM and the DPM. In addition, sites must have associated with them sufficient contaminant and hydrogeologic data to allow application of quantitative chemical fate and transport models, use of toxicity information, and postulation of exposure scenarios to arrive at quantitative measures of human-health risk at each site. As the quantitative risk assessment model to be used is designed for petroleum contamination, sites will be limited to storage tank and spill sites where chemical constituents of petroleum (e.g., benzene, ethylbenzene, toluene, etc.) are the chemicals of concern.

2.0 Literature Review

2.1 Site Risk Assessment

The National Research Council defines risk assessment as "... the characterization of the potential adverse health effects of human exposures to environmental hazards. . . . Risk assessment also includes characterization of the uncertainties inherent in the process of inferring risk[11:837]." Risk assessment serves as a tool for understanding the health and environmental hazards associated with hazardous waste management decisions [11:839].

The baseline risk assessment is an analysis of the possible adverse health effects caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases [20:1-6]. It contributes to the site characterization and the development of cleanup alternatives, if needed. The results of the baseline risk assessment can help determine what remediation efforts, if any, are necessary at the site and serve to document the magnitude of risk at a site, and the primary causes of that risk [20:4-1].

Baseline risk assessments are site-specific. They may vary in both detail and the extent to which qualitative and quantitative analyses are used, depending on the complexity and particular circumstances of the site[20:1-6].

As stated previously, the DOD prioritization model outputs are either numeric values or qualitative levels (e.g., high, medium or low risk) that indicate a risk level which can be compared with other sites. This allows ranking of a group of sites according to the estimated magnitude of risk. Though these prioritization models are not designed to replace complete baseline risk assessments, in general, the approach to site-specific risk assessment (which the models try to emulate) includes the following four key steps: data collection and analysis; exposure assessment; toxicity assessment; and risk characterization [20:1-6]. Each step is described briefly in the following.

2.1.1 Data Collection and Analysis

Data collection and analysis involves gathering and analyzing the site data relevant to the human health evaluation and identifying the substances and their concentration levels present at the site that are the focus of the risk assessment process. It may also include identification of a more narrow set of chemicals of potential concern than those initially identified by comparison with background levels or consideration of concentration and/or toxicity levels of identified substances.

2.1.2 Exposure Assessment

The detailed exposure assessment process begins after the chemical data have been collected and validated and the chemicals of potential concern have been selected [20:6-1,2]. An exposure assessment is conducted to estimate the magnitude of actual and/or potential human exposures, the frequency and duration of these exposures, and the pathways by which humans are potentially exposed. In the exposure assessment, reasonable maximum estimates of exposure are developed for both current and future land-use assumptions. Reasonable maximum estimates of exposure are subjective estimates of the maximum exposure the receptor in question could receive over each pathway under study, keeping in mind that this receptor likely would not receive maximum exposure through all pathways. Current exposure estimates are used to determine whether a threat exists based on existing exposure conditions at the site. Future exposure estimates are used to provide decision-makers with an understanding of potential future exposures and threats to include a quantitative estimate of the likelihood of such exposures occurring [20:1-6]. Generally, Superfund exposure assessments are concerned with both current and future exposures [20:6-1].

Conducting an exposure assessment involves studying contaminant releases (Table 1); identifying exposed populations and all potential pathways of exposure (Figure 1); estimating exposure point concentrations for specific pathways (based on both environmental monitoring data and predictive chemical modeling results); and estimating contaminant intakes for specific pathways. The results of this assessment are pathway-specific intakes for current and future exposures to individual substances [20:1-6].

Table 1
COMMON CHEMICAL RELEASE SOURCES AT SITES IN THE ABSENCE OF REMEDIAL ACTION [20:6-10]

Receiving Medium	Release Mechanism	Release Source
Air	Volatilization	Surface wastes - lagoons, ponds, pits, spills Contaminated surface water Contaminated surface soil Contaminated wetlands Leaking drums
	Fugitive dust generation	Contaminated surface soil Waste piles
Surface water	Surface runoff	Contaminated surface soil
	Episodic overland flow	Lagoon overflow Spills, leaking containers
	Ground-water seepage	
Ground water	Leaching	Surface or buried wastes Contaminated soil
Soil	Leaching	Surface or buried wastes
	Surface runoff	Contaminated surface soil
	Episodic overland flow	Lagoon overflow Spills, leaking containers
	Fugitive dust generation/deposition	Contaminated surface soil Wastes piles
	Tracking	Contaminated surface soil
Sediment	Surface runoff, Episodic overland flow	Surface wastes - Lagoons, ponds, pits spills Contaminated surface soil
	Ground-water seepage	Contaminated ground water
	Leaching	Surface or buried wastes Contaminated soil
Biota	Uptake (Direct contact, ingestion, inhalation)	Contaminated soil, surface water, sediment, ground water or air Other biota

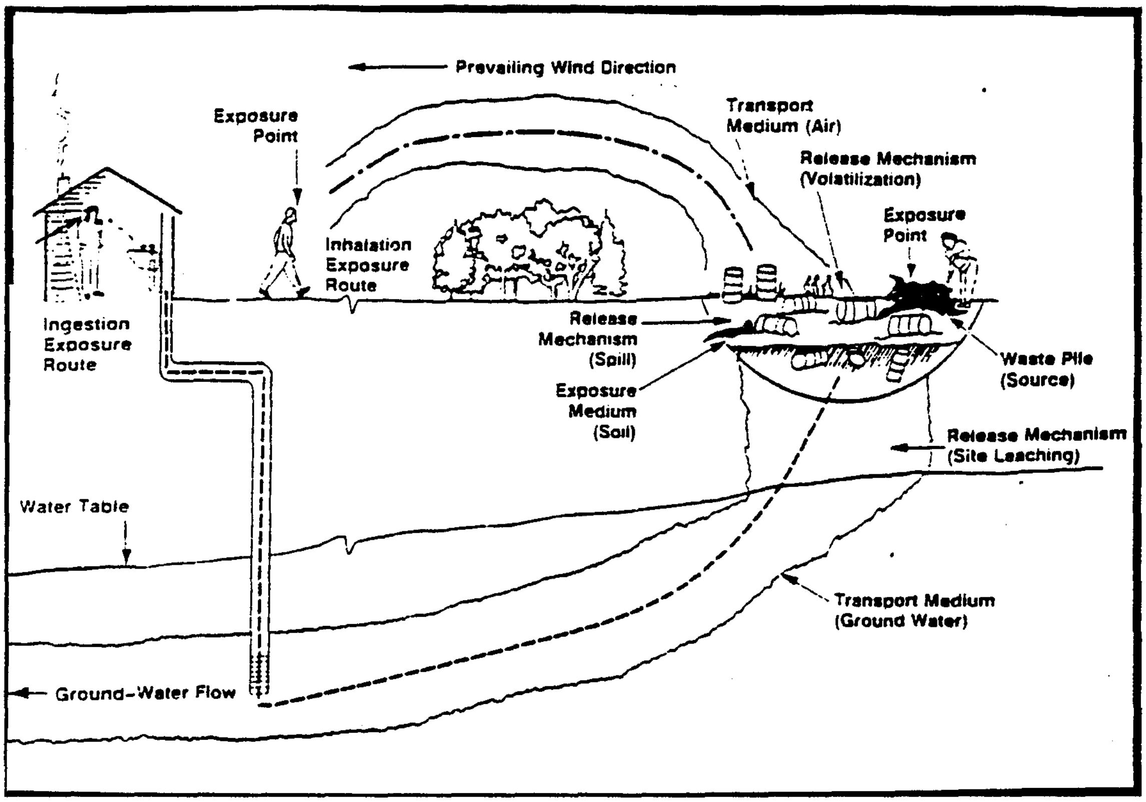


Figure 1 ILLUSTRATION OF EXPOSURE PATHWAYS [20:6-9]

The exposure assessment proceeds with the following three steps: (1) characterization of exposure setting; (2) identification of exposure pathways; and (3) quantification of exposure [20:6-2,4]. First, the exposure setting is characterized with respect to the general physical characteristics of the site such as: area climate and meteorology; geologic setting; site vegetation; soil type; and ground-water hydrology [20:6-5]. Also, the populations on or near the site are characterized with respect to location relative to the site, activity patterns, and the presence of sensitive subgroups. Sensitive subgroups are subpopulations such as children or the elderly, that may be at increased risk from chemical exposures due to increased sensitivity, behavior patterns that may result in high exposure, and/or current or past exposures from other sources.

This involves determining the distance and direction of potentially exposed populations from the site and identifying those populations that are closest to or are actually living on the site which, therefore, may have the greatest potential for exposure. It also requires consideration of populations that could be exposed in the future to chemicals that have migrated from the site [20:6-6].

The second step in the exposure assessment process is the identification of exposure points and routes. Any point of potential contact with a contaminated medium is an exposure point. Those exposure points where the concentration that would be contacted is the greatest are sought. If the site is currently used, has unrestricted access, or if contact is possible under an alternate future land use, inclusion of contaminated media or sources on site as a potential exposure point may be appropriate [20:6-11]. After exposure points are determined, probable exposure routes (i.e., ingestion, dermal contact) are identified based on the media contaminated and the anticipated activities at the exposure points [20:6-17].

The third and final step in the exposure assessment process, quantification of exposure, involves quantifying the magnitude, frequency and duration of exposure for the populations and exposure pathways selected for quantitative evaluation. This is done by first estimating exposure concentrations, and then quantifying pathway-specific intakes. The three types of variables that are used to estimate pathway-specific intake are: (1) chemical-related variables (e.g., exposure concentration); (2) variables that describe the exposed population (e.g., contact rate, exposure frequency and duration, and body weight; and (3) assessment-determined variable (e.g., averaging time) [20:6-19].

As described above, the objective of the exposure assessment is to estimate the type and magnitude of exposures to the chemicals of potential concern that are present or migrating from the site. The results of the exposure assessment are then combined with chemical-specific toxicity information to characterize potential risks [20:6-1].

2.1.3 Toxicity Assessment

The toxicity assessment portion of the baseline risk assessment considers: (1) the types of adverse health effects associated with chemical exposures; (2) the relationship between magnitude of exposure and adverse effects; and (3) related uncertainties such as the weight of evidence of a particular chemical's carcinogenicity in humans [20:1-6]. Typically, the Superfund site risk assessments rely heavily on existing toxicity information developed on specific chemicals [20:1-6].

Toxicity assessment for contaminants found at Superfund sites is usually completed in two steps: hazard identification and dose-response assessment [20:1-6]. First, the hazard identification is the process of determining whether exposure to an agent can cause an increase in the incidence of a particular adverse health effect (e.g., cancer, birth defect) and whether this adverse health effect is likely to occur in humans. Hazard identification involves characterizing the nature and strength of evidence of causation [20:7-1]. The second step, dose-response evaluation, is the process of quantitatively evaluating the toxicity information and characterizing the relationship between the dose of the contaminant administered or received and the incidence of adverse health effects in the exposed population. From this quantitative dose-response relationship, toxicity values (e.g., reference doses and slope factors) are derived that can be used to estimate the incidence or potential for adverse health effects as a function of human exposure to the agent. These toxicity values are then used in the risk characterization step to estimate the likelihood of adverse effects occurring in humans at different exposure levels [20:7-1]. The EPA's Integrated Risk Information System (IRIS) is the database containing the toxicity data used in site risk assessments [20:7-13].

2.1.4 Risk Characterization

The last step in the risk assessment process, risk characterization, summarizes and combines outputs of the exposure assessments (e.g., intakes for all exposure pathways and land uses and for all relevant substances) and toxicity assessment (e.g., toxicity values for all exposure routes and relevant substances) to characterize baseline risk, both in quantitative expressions and qualitative statements. During risk characterization, chemical-specific toxicity information is compared against both measured contaminant exposure levels and those levels predicted through fate and transport modeling to determine whether current or future levels at or near the site are of potential concern [20:1-6]."

2.2 Concept of Risk

An important distinction is whether risk is background, incremental, or total risk. Background risk is what people are exposed to in the absence of the particular source of risk being studied, incremental risk is that caused by the source, and total risk is just the sum of the two. For example, approximately 25% of the US population will experience cancer in their lifetime. This is an example of background risk [11:838]. The EPA target for Superfund sites of 1×10^{-6} excess lifetime cancer risk represents an incremental risk -- the probability of a person developing cancer from exposure to contaminants from a Superfund site is 1×10^{-6} (or "one in a million") in excess of that person's risk of cancer from all other sources combined [11:838].

At present the EPA has defined acceptable risks for carcinogens as within the range of 10^{-4} and 10^{-6} excess lifetime cancer risk and for non-carcinogens as a hazard index (discussed below) of less than 1.0. The EPA uses 10^{-6} as a point of departure, meaning that a higher risk may be considered acceptable only if there were special extenuating circumstances [11:865].

2.2.1 Determination of Carcinogenic Risk

For carcinogenic contaminants, a slope factor (plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime) is used to convert estimated daily intakes averaged over a lifetime to a value expressing incremental risk [19:240]. For Superfund risk assessments, cancer risks from various exposure pathways are assumed to be additive, as long as the risks are for the same individuals and exposure period [20:8-16]. When a baseline risk assessment indicates that an individual's cumulative risk, derived using reasonable maximum exposure assumptions for either current or future land use, exceeds 10^{-4} (i.e., 1 in 10,000 chance of developing cancer), action is generally warranted at the site. However, as mentioned previously, the EPA has expressed a preference for remediation to achieve the more protective end of the target risk range (10^{-6} or a 1 in 1,000,000 excess chance of developing cancer)[19:240].

2.2.2 Determination of Noncarcinogenic Risk

The potential for noncarcinogenic effects on health is evaluated by comparing an exposure level with a reference dose (toxicity value for evaluating noncarcinogenic effects resulting from exposures at Superfund sites). This ratio of exposure to toxicity is called a hazard quotient. The hazard quotient assumes that a level of exposure exists (i.e., reference dose (RFD)) below which it is unlikely for a receptor to experience adverse health effects. If the exposure level (E) exceeds this threshold (i.e., if E/RfD exceeds unity), the potential exists for noncarcinogenic effects to develop [19:241].

2.3 Need for Risk Assessment Modeling

Quantifying potential exposure concentrations can be attempted through actual monitoring of levels performed at points of anticipated exposure or through modeling, which simulates risk agent behavior to allow the user to predict exposure concentrations in a medium at a point.

Monitoring results can provide only a measure of the existing extent of contamination. Also, monitoring data alone may not allow the analyst to determine the contributions of specific sources (where multiple sources exist) to measured contamination. In all assessments, some degree of modeling contaminant movement in environmental media is necessary to predict the associated exposure over a 70-year lifetime. Thus a combination of monitoring and modeling techniques are necessary to conduct an analysis of contaminant fate for exposure assessment purposes [21:36].

2.4 Quantitative vs. Qualitative Modeling

The concept of risk has a long history of influencing environmental policy making. However, the initial efforts were based more on intuition than on the scientific principles of toxicology, chemistry, and fate and transport modeling intrinsic to modern risk assessments [11:839]. For example, qualitative assessments of groundwater flow (key in modeling contaminant transport) have been often based on the assumption that subsurface hydrologic gradients (which determine flow directions and rates) approximate surface topography, an approach that is unreliable and should be used only in the absence of hydrogeologic data. Groundwater flow is influenced by many factors including the hydraulic conductivity of soils, the hydraulic gradient in the area of concern, and the presence of discharge areas [21:40].

Only in the past few years has a science-based risk assessment played a major role in environmental decision making. The EPA has used the term quantitative risk assessment to describe the process of using scientific principles to calculate quantitative estimates of risk [11:839].

2.5 Site Characterization

Most of the information needed to prepare a risk assessment is generated during the site characterization process. The extent and degree of contamination, the potential receptors, the migration and exposure pathways, and the potential exposure levels are all determined through a detailed, site-specific characterization study. A lack of detail in the site characterization process results in a speculative risk assessment, which, in turn, necessitates the use of conservative or worst-case assumptions when establishing cleanup levels. As a result, the levels will be more stringent and will likely significantly increase the cost of remediation [8:286].

A detailed and site-specific characterization, therefore, is essential to the development of a realistic and appropriate assessment of risk [8:300]. The initial characterization of the physical setting that defines the risk assessment for a contaminated site, however, involves many professional judgments and assumptions [20:8-18]. One common example is the assumption of subsurface material with uniform properties throughout. Realistically, subsurface materials are almost always anisotropic (soil properties vary with direction) and heterogeneous (soil properties vary with location). To strictly account for this heterogeneity and anisotropy theoretically requires an infinite amount of data. Therefore, for the purposes of analysis of groundwater flow it is usually necessary to assume that a formation, or a portion of a formation, is homogeneous [11:149].

The extent to which the hydrogeologic properties (e.g., hydraulic conductivity, porosity, bulk density, fraction organic carbon, productivity) of the aquifer(s) are characterized may have a significant effect on the risk assessment. The ability to estimate future exposure concentrations depends on the extent to which hydrogeologic properties needed to evaluate contaminant migration are quantified [20:4-12].

The types of site data needed for a baseline risk assessment include the following: contaminant identities and concentrations in the key sources and media of interest; characteristics of sources, especially information related to release potential; and characteristics of the environmental setting that could effect the fate, transport, and persistence of the contaminants [20:4-2].

In addition to information developed during the site characterization, the physical and chemical characteristics of the contaminant must be established. The more information that is available on the contaminant, the easier the risk assessment process will be [8:286]. Among activities that should be conducted to reduce the number of chemicals of concern at a site are considering: the concentration and toxicity of the chemicals; their mobility, persistence, and bioaccumulation (e.g., a highly volatile and mobile chemical such as benzene); and considering special exposure routes (e.g., some highly volatile chemicals may pose a significant inhalation risk due to the home use of contaminated water, particularly for showering) [20:5-20].

Table 2 IMPORTANT PHYSICAL/CHEMICAL AND ENVIRONMENTAL FATE PARAMETERS [20:6-12]

K_{oc} provides a measure of the extent of chemical partitioning between organic carbon and water at equilibrium. The higher the K_{oc}, the more likely a chemical is to bind to soil or sediment than to remain in water.

K_d provides a soil or sediment - specific measure of the extent of chemical partitioning between soil or sediment and water, unadjusted for dependence upon organic carbon. To adjust for the fraction of organic carbon present in soil or sediment (f_{oc}), use $K_d = K_{oc} \times f_{oc}$. The higher the K_d, the more likely a chemical is to bind to soil or sediment than to remain in water.

K_{ow} provides a measure of the extent of chemical partitioning between water and octanol at equilibrium. The greater the K_{ow}, the more likely a chemical is to partition to octanol that to remain in water. Octanol is used as a surrogate for lipids (fat), and K_{ow} can be used to predict bioconcentration in aquatic organism.

Solubility is an upper limit on a chemical's dissolved concentration in water at a specified temperature. Aqueous concentrations in excess of solubility may indicate sorption onto sediments, the presence of solubilizing chemicals such as solvents, or the presence of a non-aqueous phase liquid.

Henry's Law Constant provides a measure of the extent of chemical partitioning between air and water at equilibrium. The higher the Henry's Law constant, the more likely a chemical is to volatilize than to remain in the water.

Vapor Pressure is the pressure exerted by a chemical vapor in equilibrium with its solid or liquid form at any given temperature. It is used to calculate the rate of volatilization of pure substance from a surface or in estimating a Henry's Law constant for chemicals with low water solubility. The higher the vapor pressure, the more likely a chemical is to exist in a gaseous state.

Diffusivity describes the movement of a molecule in a liquid or gas medium as a result of differences in concentration. It is used to calculate the dispersive component of chemical transport. The higher the diffusivity, the more likely a chemical is to move in response to concentration gradients.

Bioconcentration Factor (BCF) provides a measure of the extent of chemical partitioning at equilibrium between a biological medium such as fish tissue or plant tissue and an external medium such as water. The higher the BCF, the greater the accumulation in living tissue is likely to be.

Media-specific half-life provides a relative measure of the persistence of a chemical in a given medium, although actual values can vary greatly depending on site-specific conditions. The greater the half-life, the more persistent a chemical is likely to be.

The greatest amount of contaminant movement in soils is a function of liquid movement. Therefore, net groundwater recharge rates as well as contaminant solubility are important parameters to establish in the site characterization to estimate contaminant migration [21:40].

Site-specific meteorological conditions should be obtained (e.g., from the National Weather Service) during site characterization as they are necessary if air modeling is to be conducted [20:4-15]. The EPA states that Gaussian dispersion models can and have been successfully applied to the types of sources encountered at Superfund sites [11:211].

2.6 Contaminant Fate and Transport

The fate of contaminants is related to their persistence in the environment, which is a measure of how long a chemical will exist in a specific medium. The persistence of contaminants is reduce through biochemical degradation, oxidation, and volatilization. Volatile (VOC) and semi-volatile organic (SVOC) compounds are susceptible to biological degradation. Unlike organic compounds, metals are not degradable through biological or chemical actions and can be considered to be indefinitely persistent in the environment [9:5-3].

Solubility, advection, hydrodynamic dispersion, and adsorption of contaminants onto soil surfaces all effect the transport of dissolved contaminants in groundwater. Solubility is the maximum concentration that dissolves in pure water at a specific temperature and pH. Very soluble chemicals can be rapidly leached from wastes and contaminated soil, and are usually mobile in groundwater. Advection is transport of contaminants by movement with flowing groundwater. Hydrodynamic dispersion includes diffusion and dispersion. Diffusion is a process causing molecular spreading of contaminants from areas of greater to lesser concentration. Dispersion is mechanical mixing caused by uneven movement of groundwater through an aquifer [9:5-9].

Substances are attracted to the surface of soil particles through the process of adsorption. Sorption of an organic compound onto the organic fraction or mineral fraction of a soil retards contaminant migration causing a contaminant to migrate at a rate far less than the average groundwater velocity [9:5-3].

The fate and transport of contaminants in the environment is dependent on their chemical and physical characteristics. Some chemical and physical properties affecting the transport and fate of a chemical in the environment include aqueous solubility, volatility, Henry's Law Constant, specific gravity, organic carbon-water partitioning coefficient (K_{oc}), and half-life [9:5-1].

The volatilization rate of a chemical depends in part upon temperature, vapor pressure of the substance, and the difference in the concentration between the liquid and gas phases[11:136-37]. Volatilization can lead to venting from site soil into the atmosphere [9:5-2].

Henry's Law Constant is a good measure of a chemical's ability to move from water or moist soil to air, and provides an indication of the air/water partition at equilibrium [9:5-2]. Compounds with Henry's Law Constants greater than 10^{-3} atmospheres-cubic meter per mole can be expected to readily volatilize from water.

Those compounds with values ranging from 10^{-3} to 10^{-5} atm-m^3/mole exhibit moderate volatilization. Compounds with values less than 10^{-5} atm-m^3/mole show limited ability to volatilize from water or moist soil [9:5-2].

Specific gravity refers to the density of a given chemical in comparison to the density of pure water and, therefore, indicates its tendency to either sink or float in water. Those chemicals with specific gravities less than one will partition, if present in sufficient quantities, to the upper layers of the aquifer or body of water [9:5-2].

The K_{oc} (organic carbon-water partitioning coefficient) value indicates the tendency of a chemical to adsorb to organic matter. Almost all of the adsorption of organic chemicals by a soil is due to the organic carbon content of the soil. The organic carbon-water partition coefficient, Koc, is defined as:

$$K_{oc} = C_c / C \tag{1}$$

where

C_c = concentration adsorbed (μg adsorbed/ kg organic carbon)

C = concentration in water (μg/L).

The normal range of K_{oc} values is 1 to 10^7 milliliters per gram (mL/g), with larger values indicating greater sorption potential. Chemicals displaying strong sorption will show more limited mobility through the environment [9:5-2].

A half-life is the time required for the concentration of a substance to decrease from its starting level to one-half that level. Half of the original chemical amount or mass will have been altered (e.g., through natural biological degradation into by-products) in some manner over the half-life of the original chemical [9:5-2].

2.6.1 Volatile Organic Compounds Fate and Transport

Due to their physical characteristics, VOCs can be expected to be moderately to highly mobile in the environment. VOCs exhibit the ability to volatilize to the atmosphere and leach to groundwater. They have relatively low molecular weights and are often used as fuels or fuel components and as solvents. VOCs also have fairly high degrees of water solubility, vapor pressure, and air/water partition at equilibrium (Henry's Law Constant), and have a lower K_{oc} value. These properties that give VOCs their mobility also increase their degradability. Relative to chemicals in the other categories, VOCs show shortened half-lives in all environmental media [9:5-4].

The transport of VOCs in groundwater is primarily dependent upon both the organic content of the saturated soil and the solubility of the particular contaminant [9:5-5]. For example, some published data has shown that benzene (water solubility of 1.78E+3 mg/L) is highly mobile in groundwater/soil systems and that toluene (solubility of 5.15E+2 mg/L) is only moderately mobile [9:5-5].

2.6.2 Semi-volatile Organic Compounds Fate and Transport

SVOCs show low to moderate mobility in the environment. They usually have higher molecular weights than VOCs. Also, SVOCs have lower values for solubility, vapor pressure, and air/water partition, and show an increased tendency to sorb to organic matter (raised K_{oc}) [9:5-5]. Though the mobility of SVOCs is reduced in comparison to VOCs, their persistency as a class is slightly increased. These compounds have longer half-lives in soil, surface water, and groundwater than do VOCs with approximately equivalent atmospheric half-lives [9:5-5].

Polycyclic aromatic hydrocarbons (PAHs), one of the most commonly detected groups of SVOCs, generally have little mobility in the environment because of their high K_{oc} values, 10^3 to 10^6 mL/g, and low aqueous solubilities. This class of SVOCs, therefore, would not be expected to migrate in groundwater from a source area [9:5-5].

2.7 Scope (Current & Future Risk)

Site risk assessment is concerned both with any currently contaminated media to which individuals may be exposed or through which chemicals may be transported to potential receptors and with any currently uncontaminated media that may become contaminated in the near future due to contaminant transport [20:4-10].

As part of the exposure assessment portion of the site risk assessment, the assessor seeks to determine potentially exposed populations to include: present population in the vicinity of the site, future population in the vicinity of the site, and sub-populations of special concern (e.g., young children in the case of lead contamination) [11:849].

In evaluating current land use, land use patterns that will be applicable most often at Superfund sites are: residential; commercial/industrial; and recreational. Using basically a "common sense" evaluation, identify human activities and activity patterns associated with each land use based on a general understanding of what activities occur in residential, business, or recreational areas [20:6-6]. This should help to estimate such exposure parameters as exposure frequency, duration, and intake rate.

Determine if the current use activities are likely to be different under an alternate future land use or if land use of the site itself could change in the future. Because residential land use is most often associated with the greatest exposures, it is generally the most conservative choice to make when deciding what type of alternate land use may occur in the future. An assumption of future residential land use, however, may not be justifiable if the probability that the site will support residential use in the future is very small [20:6-7]. The development of scenarios involves making assumptions and includes subjective decisions. Ideally, this should produce credible scenarios; but in an effort to ensure conservative risk assessments, unrealistic scenarios have been used at some hazardous waste sites [11:851].

2.8 Uncertainties

Some doubt exists as to how well an exposure model or its mathematical expression (e.g., groundwater transport model) approximates the true relationships between site-specific environmental conditions. At a minimum, it is important to identify key model assumptions (e.g., linearity, homogeneity, steady-state conditions, equilibrium) and indicate the potential impact of each assumption (and any data gaps) on risk with respect to both direction and magnitude, as shown in Table 3 [19:8-18,19].

Table 3
EXAMPLE OF AN UNCERTAINTY TABLE FOR EXPOSURE ASSESSMENT [20:6-51]

	EFFECT ON EXPOSURE[a]		
ASSUMPTION	Potential Magnitude for Over-Estimation of Exposure	Potential Magnitude for Under-Estimation of Exposure	Potential Magnitude for Over-or Under Estimation of Exposure
Environmental Sampling and Analysis			
Sufficient samples may not have been taken to characterize the media being evaluated, especially with respect to currently available soil data.			Moderate
Systematic or random errors in the chemical analyses may yield erroneous data.			Low
Fate and Transport Modeling			
Chemicals in fish will be at equilibrium with chemical concentration in water.	Low		
Use of a Gaussian dispersion model to estimate air concentrations offsite.			Low
Use of a box model to estimate air concentrations onsite.	Low		
Use of Cowherd's model to estimate vehicle emission factors.		Moderate	
Exposure Parameter Estimation			
The standard assumptions regarding body weight, period exposed, life expectancy, population characteristics, and lifestyle may not be representative of any actual exposure situation.			Moderate
The amount of media intake in assumed to be constant and representative of the exposed population.	Moderate		
Assumption of daily lifetime exposure for residents.	Moderate to High		
Use of "hot spot" soil data for upper-bound lifetime exposure	Moderate to High		

[a] As a general guideline, assumptions marked as "low" may affect estimates of exposure by less than one order of magnitude; assumptions marked "moderate" may affect estimates of exposure by between one and two orders of magnitude; and assumptions marked "high" may affect estimates of exposure by more than two orders of magnitude.

2.9 The Defense Priority Model

2.9.1 Background

As mentioned previously, the DPM was used by the Air Force and DOD to prioritize IRP sites for remedial action based on the relative risk to human health and the environment. Site specific data collected during the PA/SI and RI/FS provide input into the DPM, which in turn provides a method to generate a score that can be viewed as a "common measuring stick," representing the relative environmental risk of a site [4:17].

Development of a methodology for ranking Air Force contaminated sites began in the mid-1980's. Ongoing efforts in model development/use lead to the development of the DPM, officially adopted in November 1987 as the Air Force site ranking model [4:18]. DPM has undergone several revisions since then -- the latest being automated *DPM Quick* (ADPM) in the beginning of fiscal year 1994. Soon after the latest DPM release, the DOD announced that the DPM would no longer be used as a site ranking tool. It was abandoned due to both NRC criticisms and because it was considered by DOD officials to be too complicated, making stakeholder understanding and participation in the process difficult [3].

2.9.2 Model Structure Overview

The DPM uses a combination of quantitative data and qualitative approximations and calculates subscores for adverse effects on human and ecological receptors from surface water, groundwater, air and soil volatiles, and air and dust pathways [4:19]. The numerical subscores are calculated for each of eight combinations of potential transport pathways and receptors. Each score is intended to represent relative risk and is based on the following three factors: (1) the characteristics, concentration and mobility of contaminants found at the site (hazards), (2) the potential for contaminant transport through the environmental media (pathways), and (3) the presence of potential receptors of site contamination. All three factors are required in order to score a site under the DPM [4:19].

The structure of DPM scoring is shown in Figure 2 [14:3]. The final score is obtained by weighing and combining the subscores using a weighted "root-mean-square" algorithm, an exponential algorithm intended to increase the importance of a single high pathway-receptor subscore on the final risk score [14:133]. As illustrated in Figure 2, subscores are first calculated for each *pathway,* intended to rate the potential for contaminants from a site to enter each pathway given the physical characteristics of the pathway [14:1]. These various physical characteristics are typically given a score to represent the numerical range the particular characteristic falls (or likely is) within. In other words, the actual parameter value (e.g., hydraulic conductivity) is not used in calculations, but rather a score (e.g., 0, 1, 2, or 3) representing the range the parameter falls within. This level of information on each characteristic is manipulated in overall scoring calculations. For the surface water pathway, some physical characteristics include: distance to surface water, net precipitation, erosion potential, hydraulic conductivity, average rainfall intensity, and flooding potential [14:11]. For the groundwater pathway, some characteristics considered include: distance from bottom of the waste to seasonal high groundwater, permeability of the unsaturated zone, infiltration potential, and ranges of various geochemical properties of the unsaturated zone (e.g. clay content, organic content, and pH) [14:35]. For the air/soil volatiles and air /soil dust pathways, physical characteristics include: average summer soil temperature, net precipitation, wind velocity, soil porosity, and level of site activity [14:49].

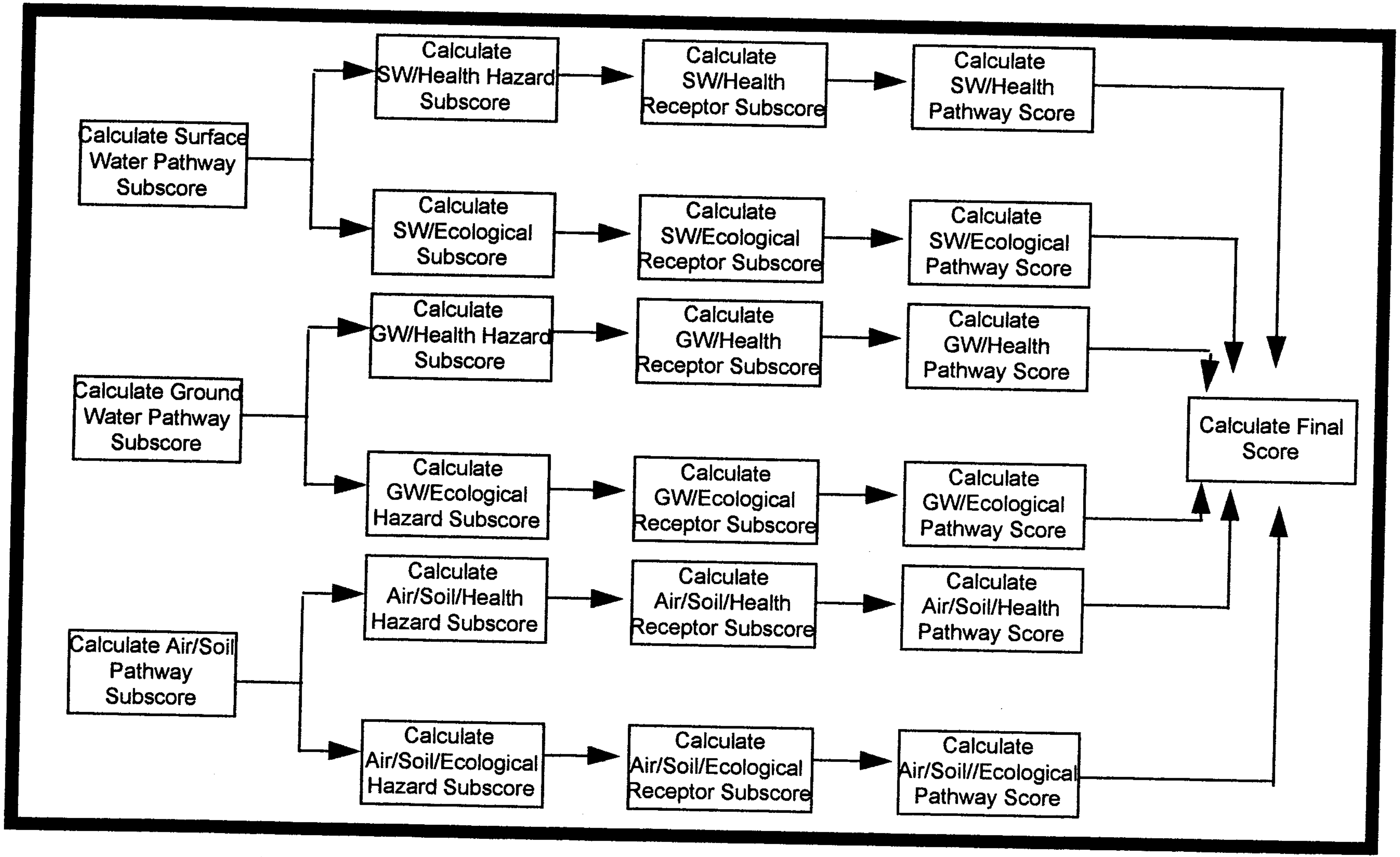


Figure 2: DPM STRUCTURE [14:3]

Besides consideration of the physical characteristics for each of the pathways mentioned above, another component of developing the pathway scores is the "waste containment effectiveness factor," which takes on a value between zero and one and is designed to adjust the pathway score to account for the effectiveness of engineered barriers or clean-up actions in reducing the potential for contaminant transport along a particular pathway [14:26].

The final component considered in the development of pathway scores is the "waste quantity factor," which is the amount of contaminant associated with the site. The DPM Manual suggests that "the goal is to simply differentiate between a small, moderate or large amount [of waste] [14:26]."

After pathway subscores are developed, the *contaminant hazard subscores* for each pathway are calculated. These subscores are intended to rate the human and ecological hazards of site contaminants. As partially illustrated in Figure 3, subscores are calculated for each of eight combinations of potential transport pathways and receptors.

Surface Water Human Health Score	= Surface Water Pathway Score	X Surface Water Human Health Hazard Score	X Surface Water Human Receptor Score	/10,000
Surface Water Ecological Score	= Surface Water Pathway Score	X Surface Water Ecological Hazard Score	X Surface Water Ecological Receptor Score	/10,000
Ground Water Human Health Score	= Ground Water Pathway Score	X Ground Water Human Health Hazard Score	X Ground Water Human Receptor Score	/10,000
Ground Water Ecological Score	= Ground Water Pathway Score	X Ground Water Ecological Hazard Score	X Ground Water Ecological Receptor Score	/10,000
Air/Soil Volatiles [1] Human Health Score	= Air/Soil Volatiles Pathway Score	X Air/Soil Volatiles Human Health Hazard Score	X Air/Soil Volatiles Human Receptor Score	/10,000
Air/Soil Volatiles [2] Ecological Score	= Air/Soil Volatiles Pathway Score	X Air/Soil Volatiles Ecological Hazard Score	X Air/Soil Volatiles Ecological Receptor Score	/10,000
Air/Soil Dust [1] Human Health Score	= Air/Soil Dust Pathway Score	X Air/Soil Dust Human Health Hazard Score	X Air/Soil Dust Human Receptor Score	/10,000
Air/Soil Dust [2] Ecological Score	= Air/Soil Dust Pathway Score	X Air/Soil Dust Ecological Hazard Score	X Air/Soil Dust Ecological Receptor Score	/10,000

[1] The higher of these two scores is used in the final computation.
[2] The higher of these two scores is used in the final computation.

Figure 3: ALGORITHM TO CALCULATE DPM SCORES [14:24]

For media in which contamination has been detected, the DPM employs the concept of Average Daily Intake (ADI) for scoring human health hazards. The detected concentration of a contaminant is first converted to a daily intake and then divided by the benchmark ADI (analogous to the reference dose) associated with the contaminant. A quotient (hazard quotient) is calculated for each contaminant and then summed to provide a score for the surface water and air/soil pathways. A hazard quotient greater than one is considered to indicate a threat. This procedure also applies to the groundwater pathway, except the quotients are divided by derived retardation factors calculated for the respective contaminants before being summed to provide the human health hazard score [14:71]. The method of calculating this retardation factor, described in the DPM User's Manual, requires the soil bulk density, effective porosity, and the fraction of organic content [14:73].

All contaminants known to be present at a site (detected above background levels) are considered in calculating the total human health and ecological hazard scores for both surface and groundwater pathways [14:85]. The contaminant hazard scores are set to zero for no detectable concentrations of contaminants for the air/soil volatile and air/soil dust pathways [14:61].

The final factor addressed under the DPM involves the development of *receptor scores*. The receptor portion of DPM rates the potential for human and ecological receptors to be exposed to contaminants released from the site under study. The eight types of receptors scenarios considered are both human and ecological receptors to the following: surface water contaminants, groundwater contaminants, and lastly, air/soil contaminants. For the air/soil pathways, only the more conservative of the two receptor scores (air/soil dust and air/soil volatile pathways) is used in final site scoring, reducing the number of pathway-receptor subscores used in the final scoring to six. Factors such as the size and proximity of nearby populations and the use of surface and groundwater are considered in developing receptor scores [14:111].

After the receptor scoring is complete, the pathway, hazard, and receptor scores for each pathway-hazard-receptor combination are multiplied to provide the subscores. These subscores are normalized on a 100-point scale and equally weighted [14:133]. The final site score is a combination of the six pathway-receptor subscores that are aggregated using a weighted (human health subscores weighted five times more than their respective ecological subscores) root-mean-square algorithm to obtain the final site score[14:133]. This root-mean-square methodology is an exponential algorithm. When a score for a single pathway-receptor is high, the algorithm will subsequently result in a high score. If additional subscores are high, the final score will increase, but not linearly. This methodology is intended to increase the importance of a single high pathway-receptor subscore on the final risk score [4:25]. Further information on this methodology may be found in the DPM User's Manual (May 1992).

2.9.3 Criticisms of the DPM

The DPM, however, was criticized in a recent NRC report [13:135-176]. The NRC's Committee on Remedial Action Priorities for Hazardous Waste Sites was asked to examine the principle ranking methods being used by the federal agencies to rank contaminated sites for cleanup. This study effort began in 1990. Among other issues, the committee was asked to consider ". . . the method's assumptions; the appropriateness of the assumptions for the method's intended purposes; the sources, magnitude, and treatment of significant uncertainties in each method [13:3]..."

The DOD, one of the NRC project supporters, requested and received an interim report (1992) assessing the DPM. Some of the criticisms of DPM included the fact that 65% of the 284 sites evaluated in 1991 had DPM scores between 13 and 37, with such a narrow interval suggesting that DPM's ability to discriminate between sites may be limited [13:9]. Another important observation of the NRC report was that "the extensive scientific peer review, public participation, and public comments that are needed for establishing credibility and acceptability of a model to be used in priority-setting have not been conducted with the DPM [13:174]."

Further review by the NRC revealed that the <u>DPM User's Manual</u> provides instructions on how to run the DPM but does not explain the science or rationale for the various model algorithms. Furthermore, the manual does not detail the objectives, assumptions, and intended use of the model, leaving it unclear why and how DPM's scores for potential threats are combined with the threat's magnitude, immediacy, and probability [13:146]. Such information ". . .is necessary for evaluating whether a particular type of risk is being quantified consistently and checking whether the model's default values are chosen consistently with some explicitly stated policy [13:146]."

The DPM makes use of pathway algorithms that are based on various concepts of contaminant transport and fate. These algorithms attempt to qualitatively capture the dependence of the contaminant concentration, at a given distance from the source, on various physicochemical and transport parameters. All contaminant transport and fate is modeled qualitatively in the DPM, except for the air and soil pathways. A major deficiency for the surface water and groundwater pathways, however, is that the qualitative algorithm does not allow consideration of the magnitude of the pollutant release rates or concentrations at the receptor site in arriving at the pathway scores [13:146]. Consideration of such contaminant information seems an essential element of any meaningful statement of potential site risk to the public.

Further NRC review of the DPM revealed that the algorithms used for the surface water, groundwater, and air and soil pathway factors are not entirely consistent with accepted theory. For example, the pathway algorithm uses a summation formula, whereas theory suggests that a multiplicative formula or summation on a logarithmic scale would be the preferable approach for scoring the pathway potential [13:147].

As the DPM has not been validated and has been viewed by some to be scientifically questionable, this thesis effort will attempt an initial validation effort against a quantitative baseline model, that is assumed *a priori* to provide the correct site rankings.

2.10 Relative Risk Evaluation Method

2.10.1 Background

In a Government Accounting Office (GAO) report entitled *Environmental Cleanup: Too Many High Priority Sites Impede DOD's Program* (3 May 1994), the GAO concluded that the current method by which regulators and the DOD prioritize sites for cleanup results in "(1) too many similar priorities where too little gets done, or (2) instances where DOD's worst sites are not getting priority attention [15:4]." The report further stated that the current approach based solely on regulation-driven rather than riskdriven requirements leads to significant cost growth that strains limited resources and forces difficult choices.

On 14 Apr 94, the Office of the Deputy Under Secretary of Defense (Environmental Security)(DUSD[ES]) directed each service within DOD to begin developing their environmental restoration program using a relative risk evaluation framework. Implementation procedures were developed and issued during summer 1994 [15:4].

Some said benefits of the RREM are: it is simple and easy to understand; it does not rely on "Black Boxes" for evaluation; it establishes a common approach for categorizing sites across DOD Components; and it serves as a basis for discussing relative site risks with stakeholders [15:C10].

According to the Relative Risk Site Evaluation Primer (summer 1994), the goal of the RREM is "to ensure that sites with higher risk (relative to other sites) are generally considered first in the priority setting process [15:2]." The function of the framework is to categorize sites into High, Medium, and Low categories based on relative risk [15:2].

2.10.2 Model Structure and Overview

Under RREM, the categorization of sites into relative risk groups is based on evaluation of contaminants, pathways, and human and ecological receptors in groundwater, surface water and sediment, and surface soils. Each of these environmental media are evaluated using the following three factors: (1) the Contaminant Hazard Factor (CHF), a combined measure of contaminant concentrations in a given environmental medium; (2) the Migration Pathway Factor (MPF), a measure of the movement or potential movement of contamination away from the original source; and (3) the Receptor Factor (RF), an indication of the potential for human or ecological contact with site contaminants [15:ES].

Community representatives and stakeholders at an installation provide input to the DOD for these site evaluations. The results will be used by the DOD, along with other risk management concerns (e.g., regulatory agreement status of sites, public health recommendations), to help sequence site cleanup [15:ES].

Relative risk site evaluations should be based on currently available information on contaminants, migration pathways, and receptors; additional data gathering activities are not required [15:5]. Information required to complete relative risk site evaluations should be obtained from existing restoration documents. Examples of such documents include completed site inspections, remedial investigations, feasibility studies, engineering evaluations/cost analysis studies, records of decision, decision documents, or equivalent types of information [15:17].

RREM utilizes available site information to evaluate groundwater, surface water and associated sediments, and surface soils, preferably from a depth of 0-6 inches (if such samples are not available results from depth up to, but not exceeding, 24 inches can be used) [15:6].

Each of the three media is evaluated using the three factors mentioned previously: the CHF, MPF, and RF. Each of these factors is given a rating (e.g., Significant, Moderate, or Minimal for CHF) based on available site information for a given media.

For each site, ratings are combined in an overall category of High, Medium, or Low [15:6]. This site-specific process is illustrated schematically in Figure 4. Figure 5 expands on Figure 4, and illustrates the decision points for media-specific relative risk site evaluations. Figures 6 through 8 provide definitions of the factor ratings for groundwater, surface water/sediment, and surface soils, respectively.

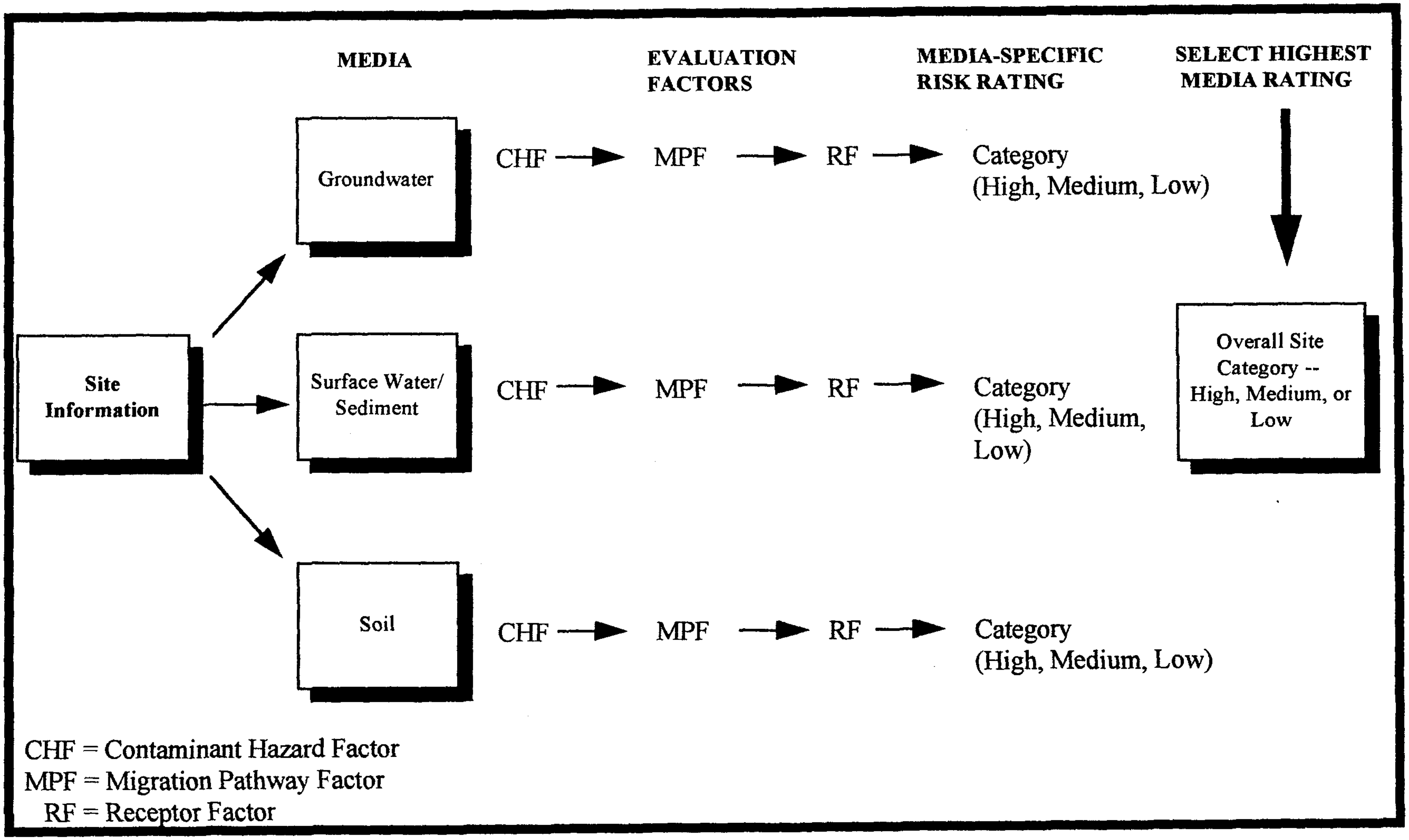
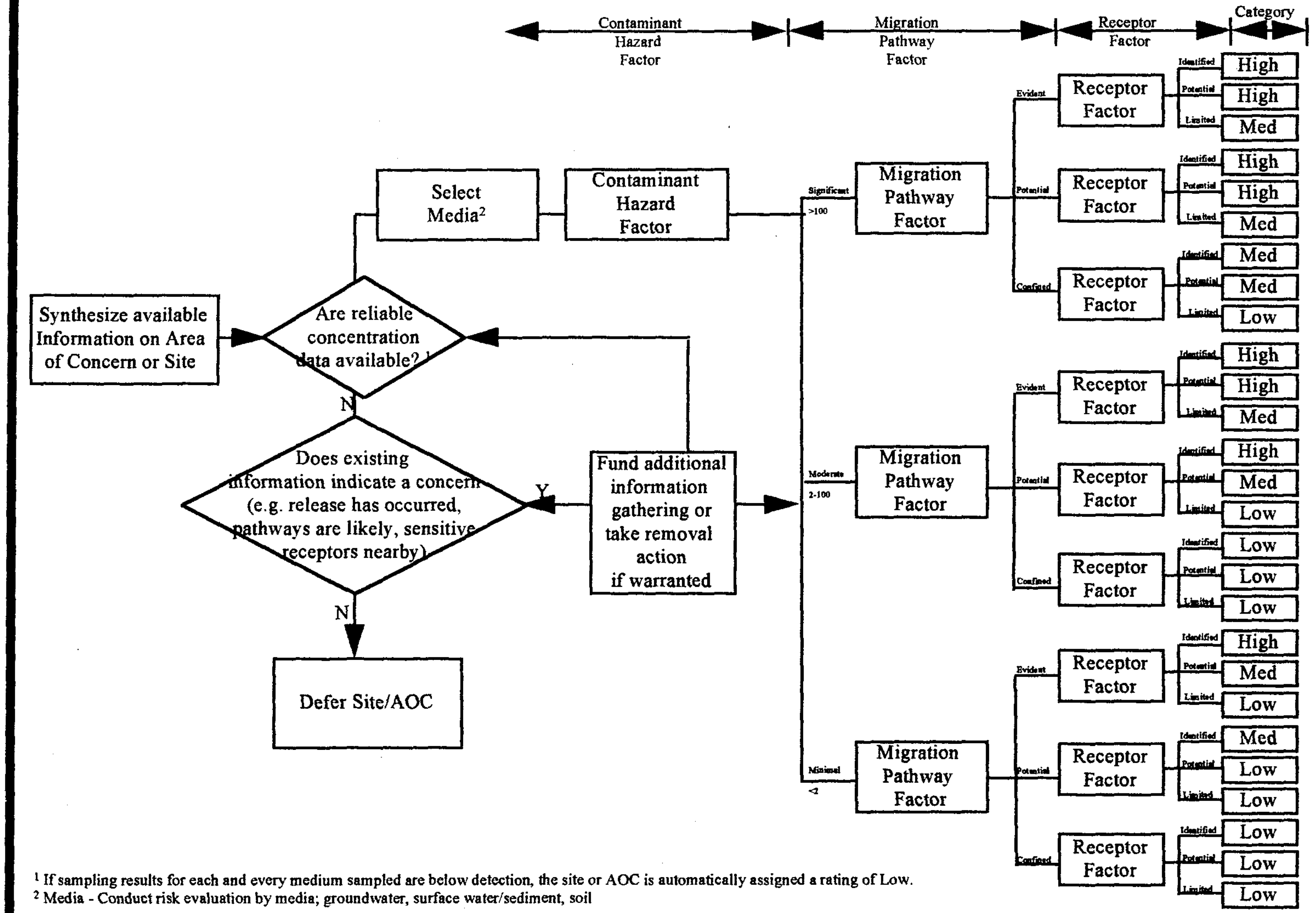


Figure 4: *FLOW DIAGRAM OF THE RELATIVE RISK SITE EVALUATION FRAMEWORK [15:8]*



[1] If sampling results for each and every medium sampled are below detection, the site or AOC is automatically assigned a rating of Low.

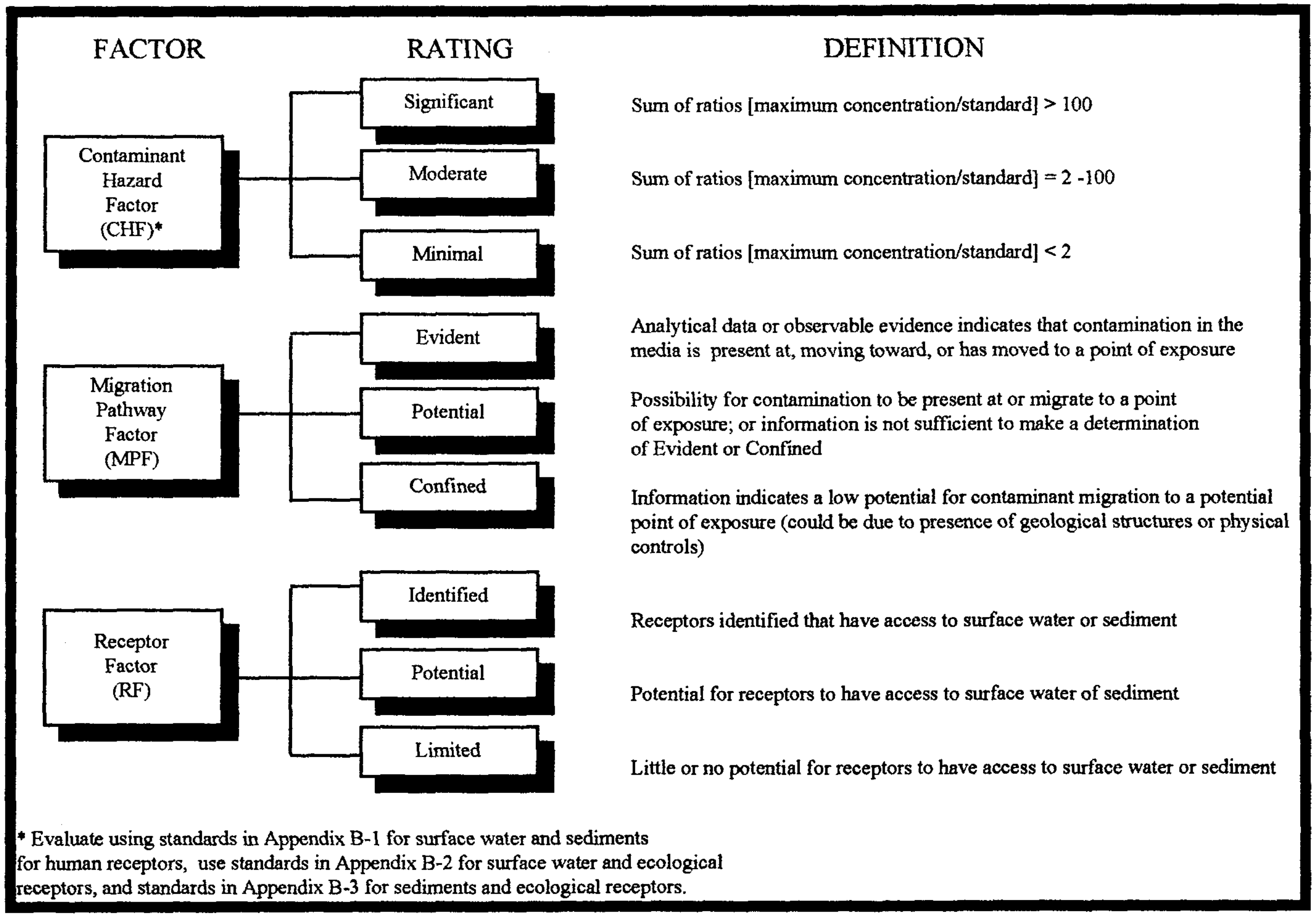
[2] Media - Conduct risk evaluation by media; groundwater, surface water/sediment, soil

Figure 5: RELATIVE RISK SITE EVALUATION FRAMEWORK: DECISION FLOWCHART [15:9]

FACTOR	RATING	DEFINITION
Contaminant Hazard Factor (CHF)*	Significant	Sum of ratios [maximum concentration/standard] > 100
	Moderate	Sum of ratios [maximum concentration/standard] = 2 -100
	Minimal	Sum of ratios [maximum concentration/standard] < 2
Migration Pathway Factor (MPF)	Evident	Analytical data or observable evidence indicates that contamination the media is moving away from the source
	Potential	Possibility for contamination to the be present at or migrate to point of exposure; or information is not sufficient to make a determination of Evident or Confined
	Confined	Information indicates that the potential for contaminant migration from the source is limited (due to geological structures or physical controls)
Receptor Factor (RF)	Identified	There is a threatened or potentially threatened water supply down gradient of the source. The groundwater (contaminated or not) is a current source of drinking water or source of water for other beneficial uses such as irrigation/ agriculture (equivalent to Class I or IIA aquifer)
	Potential	There is not potentially threatened water supply well downgradient of the source. The groundwater is potentially usable for drinking water, irrigation, or agriculture but is not presently used (equivalent to Class IIB aquifer)
	Limited	There is no potentially threatened water supply well downgradient of the source. The groundwater is not considered a potential source of drinking water and is of limited beneficial use (equivalent to Class IIA or IIIB aquifer, or where perched aquifer exists only)

* Evaluate using standards in Appendix B-1

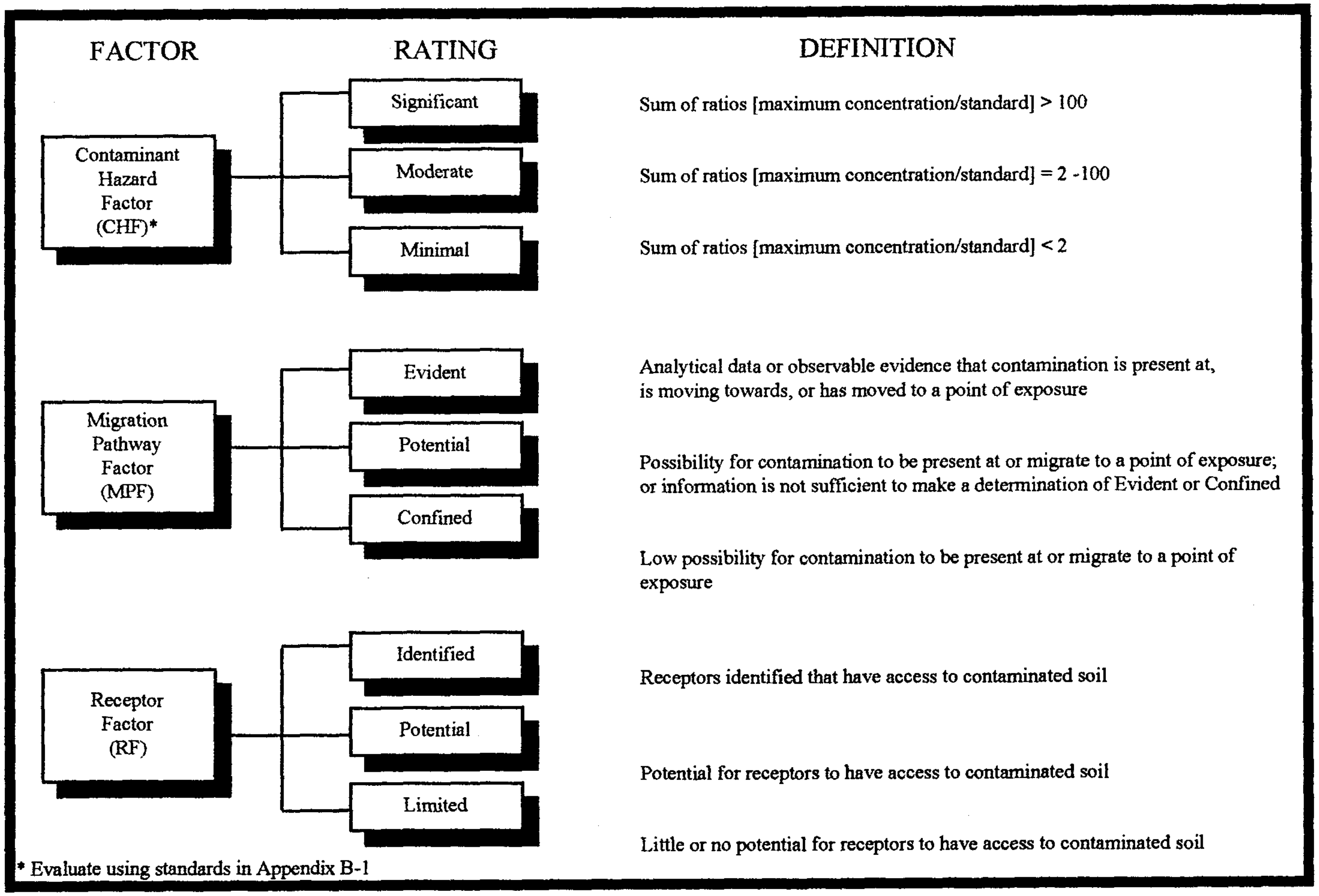
Figure 6: SITE EVALUATION FACTOR INFORMATION FOR GROUNDWATER [15:10]



FACTOR	RATING	DEFINITION
Contaminant Hazard Factor (CHF)*	Significant	Sum of ratios [maximum concentration/standard] > 100
	Moderate	Sum of ratios [maximum concentration/standard] = 2 -100
	Minimal	Sum of ratios [maximum concentration/standard] < 2
Migration Pathway Factor (MPF)	Evident	Analytical data or observable evidence indicates that contamination in the media is present at, moving toward, or has moved to a point of exposure
	Potential	Possibility for contamination to be present at or migrate to a point of exposure; or information is not sufficient to make a determination of Evident or Confined
	Confined	Information indicates a low potential for contaminant migration to a potential point of exposure (could be due to presence of geological structures or physical controls)
Receptor Factor (RF)	Identified	Receptors identified that have access to surface water or sediment
	Potential	Potential for receptors to have access to surface water of sediment
	Limited	Little or no potential for receptors to have access to surface water or sediment

* Evaluate using standards in Appendix B-1 for surface water and sediments for human receptors, use standards in Appendix B-2 for surface water and ecological receptors, and standards in Appendix B-3 for sediments and ecological receptors.

Figure 7: SITE EVALUATION FACTOR INFORMATION FOR SURFACE WATER/SEDIMENT [15:11]



FACTOR	RATING	DEFINITION
Contaminant Hazard Factor (CHF)*	Significant	Sum of ratios [maximum concentration/standard] > 100
	Moderate	Sum of ratios [maximum concentration/standard] = 2 -100
	Minimal	Sum of ratios [maximum concentration/standard] < 2
Migration Pathway Factor (MPF)	Evident	Analytical data or observable evidence that contamination is present at, is moving towards, or has moved to a point of exposure
	Potential	Possibility for contamination to be present at or migrate to a point of exposure; or information is not sufficient to make a determination of Evident or Confined
	Confined	Low possibility for contamination to be present at or migrate to a point of exposure
Receptor Factor (RF)	Identified	Receptors identified that have access to contaminated soil
	Potential	Potential for receptors to have access to contaminated soil
	Limited	Little or no potential for receptors to have access to contaminated soil

* Evaluate using standards in Appendix B-1

Figure 8: SITE EVALUATION FACTOR INFORMATION FOR SOILS [15:2]

The CHF rating is determined based on the ratio of the maximum concentration of a contaminant in the media to a risk-based concentration standard for that contaminant. For human health under the RREM, the concentration standard for carcinogens will be the concentration that presents a 1 in 10,000 risk of increased cancer incidence, which is the remedial action threshold for carcinogens defined by the EPA in QSWER Directive 9355.0-30. For non-carcinogens, the concentration standard will be the concentration that provides an exposed individual with the daily reference dose (RfD), which is the estimated daily exposure level of a contaminant to a human population below which adverse non-cancer health effects are not anticipated [15:7].

Concentration standards used by the RREM for various contaminants in soil and water are based on the most current information in the Integrated Risk Information System (IRIS) and Health Effects and Assessment Summary Tables (HEAST). The original concentrations for carcinogens (based on a one-in-one-million risk of increased cancer incidence) were multiplied by 100 for use in the CHF calculation [15:7].

For media containing more than one contaminant, the ratios from the individual contaminants are added. A CHF rating of significant (greater than 100), moderate (from 2 to 100), or minimal (less than 2) is assigned on the basis of the magnitude of the ratio or sum of ratios [15:13].

When conducting relative risk evaluations for sites contaminated solely with petroleum, oils, and lubricants (POL), only the concentrations for benzene, toluene, ethylbenzene, and xylene (BTEX) compounds in each media are used, together with corresponding BTEX standards, to calculate the CHF [14:17]. Calculation of the CHF is illustrated in Figures 9 and 10.

Contaminants

Carcinogen A: $[A]^1$ max
Carcinogen B: [B] max
Non-Carcinogen C: [C] max
Ecological D: [D] max

Calculation*

$$\frac{[A]^1\ \text{max}}{\text{Std}^2} + \frac{[B]\ \text{max}}{\text{Std}^2} + \frac{[C]\ \text{max}}{\text{Std}^3} = X_1$$

$$\frac{[D]\ \text{max}}{\text{Std}^4} = X_2$$

Rating

> 100 = Significant CHF
2-100 = Moderate CHF
< 2 = Minimal CHF

$[A]^1$ - Maximum concentration in media
Std^2 - Standard based on 10^{-4} human cancer incidence
Std^2 - Standard based on Hazard Quotient of 1 for humans
Std^2 - Standard for ecological receptors where available

* Use concentration standards in Appendix B

Note: Contaminants posing a threat to ecological receptors (i.e., ecological contaminants) should be evaluated separately from those posing a threat to human receptors

Figure 9: MECHANICS OF THE CHF CALCULATION [15:23]

Contaminants		Calculation*	Rating
Carcinogen A:	$[A]^1$ max	$\frac{[A]^1\ \text{max}}{\text{Std}^2} + \frac{[B]\ \text{max}}{\text{Std}^2} + \frac{[C]\ \text{max}}{\text{Std}^3} + \frac{[E]\ \text{max}}{\text{Std}^2} + \frac{[E]\ \text{max}}{\text{Std}^3} = X_1$	> 100 = Significant CHF 2-100 = Moderate CHF < 2 = Minimal CHF
Carcinogen B:	[B] max		
Non-Carcinogen C:	[C] max		
Carcinogen/ Non-carcinogen:	[E] max		
Ecological D:	[D] max	$\frac{[D]\ \text{max}}{\text{Std}^4} = X_2$	

$[A]^1$ - Maximum concentration in media
Std^2 - Standard based on 10^{-4} human cancer incidence
Std^2 - Standard based on Hazard Quotient of 1 for humans
Std^2 - Standard for ecological receptors where available

* Use concentration standards in Appendix B

Note: Contaminants posing a threat to ecological receptors (i.e., ecological contaminants) should be evaluated separately from those posing a threat to human receptors

Figure 10: MECHANICS OF THE CHF CALCULATION FOR SUBSTANCES WITH BOTH CARCINOGENIC AND NON-CARCINOGENIC EFFECTS [15:24]

Information about migration pathways of contamination for a site is summarized as the MPF. Ratings of evident, potential, or confined are made by matching available site information on the pathways with the three definitions about the likelihood of contaminant migration provided in Figures 4 through 6. Site evaluators should determine the MPF on the basis of professional judgment and consideration of available site information [15:13].

Information about the present or future likelihood of receptors for each site is summarized as the RF. Ratings of identified, potential, or limited are made by matching available information on receptors at sites with the definitions provided in Figures 4 through 6. Like those for the MPF, these statements should be considered based on available information and professional judgments about the site. For groundwater, human receptors include downgradient water supplies used for human consumption or in food production. For surface water/sediment, receptors include down gradient water supplies and recreational areas used for drinking water, irrigation of food crops, watering of livestock, aquaculture, and/or recreational activities as fishing. For surface soil, human receptors include workers, residents, schools, and daycare [15:13].

The CHF, MPF, and RF ratings for each medium at a site are combined using the relative risk flowchart as shown in Figure 3 to obtain a risk designation of High, Medium, or Low. The highest media designation of the three is chosen as the risk designation for the site [15:14].

2.10.3 Criticisms of the RREM

As discussed previously, specific criticisms of the RREM include: 1) the framework relies too heavily on qualitative factors; 2) many of its definitions are unclear, such as how it defines a receptor; and 3) it is unclear how much rigor will be put into the analysis [18:1]. One overall recommendation included that ". . . the process define a more quantitative, rigorous approach to characterizing sites; and, that more variables (pathways and receptors) be included [18:1]."

2.11 American Petroleum Institute's Risk/Exposure Assessment Decision Support System (DSS)

2.11.1 Introduction

The exposure/risk assessment process involves site characterization, contaminant fate and transport modeling, toxicology, environmental chemistry, and data analysis. Both federal and state guidance encourage the use of site-specific risk assessments in estimating risk (e.g. Risk Assessment Guidance for Superfund) [2:1-1]. It is important to develop a computerized framework to provide a consistent and cost-effective approach for determining risk and cleanup levels at contaminated sites. The DSS seeks to ensure a consistent approach to site risk assessment [2:1-1].

The DSS consists of the following four modules: 1) Development of Risk Scenario, 2) Fate and Transport, 3) Chemical Intake and Risk Calculation, and 4) Risk Presentation. Each module is discussed briefly below.

The *Development of Risk Scenario* module allows the user to develop a conceptual model of the site for the purposes of risk assessment [2:1-4,5]. Key elements of the conceptual model include the contaminated media, the chemicals of concern, the types and locations of receptors and relevant routes of exposure, and the manner in which the receptor point concentrations for each chemical are to be estimated (e.g., through directly entered concentrations or contaminant transport modeling) [2:4-1]. The data required to estimate the risk at a site depend on the conceptual model developed. Based on these elements input by the user, the DSS summarizes all the data that will be necessary to estimate the risk [2:4-7]. Specific values for these parameters can be estimated based on a combination of site-specific field investigations, a literature search, professional judgment, or default values included in the DSS databases [2:4-7].

The procedures used to compute intake in the DSS are consistent with those described in EPA's Risk Assessment Guidance for Superfund (EPA,1989a). This module can be used to estimate chemical intake for the following routes:

1) Ingestion of contaminated water
2) Inhalation while showering
3)Dermal contact with contaminated groundwater
4) Ingestion of contaminated soil
5) Dermal contact with contaminated soil
6) Inhalation of air containing chemical vapors or particulates.

These are typically the most important routes of exposure at petroleum contaminated sites [2:6-1].

Exposure routes should be selected based on the receptor type and expected location of the receptor. For example, if risks to an on-site resident are being estimated, then all the above routes may be of concern. However, if an on-site worker is being studied, the shower routes might be eliminated leaving just inhalation of soil emissions, dermal contact with soil, soil ingestion, and the ingestion of groundwater as potential routes of concern [2:4-4]. A general guideline for the DSS user is provided in Table 4.

Table 4
SUGGESTED ROUTES OF EXPOSURE FOR SELECTED RECEPTOR TYPE AND CONTAMINATED MEDIUM [2:4-5]

Receptor Type and Location	Contaminated Media	Possible Routes of Exposure					
		1	2	3	4	5	6
On-site Resident	Soil	•	•	•	•	•	•
	Groundwater	•	•	•	---	---	---
	Air	---	---	---	•	---	---
Off-Site Resident	Soil	•	•	•	•	---	---
	Groundwater	•	•	•	---	---	---
	Air	---	---	---	•	---	---
On-Site Worker	Soil	---	---	---	•	•	•
	Groundwater	•	---	---	---	---	---
	Air	---	---	---	•	---	---
Off-Site Worker	Soil	---	---	---	•	---	---
	Groundwater	•	---	---	---	---	---
	Air	---	---	---	•	---	---
Visitor	Soil	---	---	---	•	•	•
	Groundwater	---	---	---	---	---	---
	Air	---	---	---	•	---	---

1. Ingestion of water
2. Dermal absorption while showing
3. Inhalation while showering
4. Inhalation of emissions from soil
5. Dermal contact with soil
6. Ingestion of soil

Models included are atmospheric emission and dispersion models, and unsaturated and saturated zone contaminant transport models [2:1-4,5]. In some cases the fate and transport models are linked together. For instance, the Jury model (a vadose zone model) may be used to calculate contaminant mass loading to groundwater and volatile emission rates, with both of these outputs used as inputs to other models. The loading to groundwater is used as an input to the AT123D saturated zone model, and the volatile emission rates are used as an input to an air dispersion model (if applicable to the scenario under study) [2:4-6].

The *Chemical Intake and Risk Calculation* module uses the computed receptor point concentrations (calculated in the fate and transport module) or user-entered concentrations to estimate chemical intake by the human receptor for the exposure routes mentioned previously. This module also estimates the carcinogenic risk and non-carcinogenic effect (hazard index) due the these exposures. As shown in Figure 11, receptor point concentrations from fate and transport modules (or user specified concentrations) are used to calculate the intake of chemicals by human receptors for each exposure route. Chemical intake is computed using equations that include variables for exposure concentrations, frequency, duration and averaging time of exposure, body weight, and the receptor contact rates with the contaminated media [2:6-1]. This intake rate, expressed in milligrams per day of chemical taken into the body per unit body weight (mg/kg-d), is then averaged over time to quantify the risk for each exposure route [2:6-1].

The *Risk Presentation* module presents as output the estimated chemical intake, carcinogenic risk, and hazard index for each pathway [2:1-4,5].

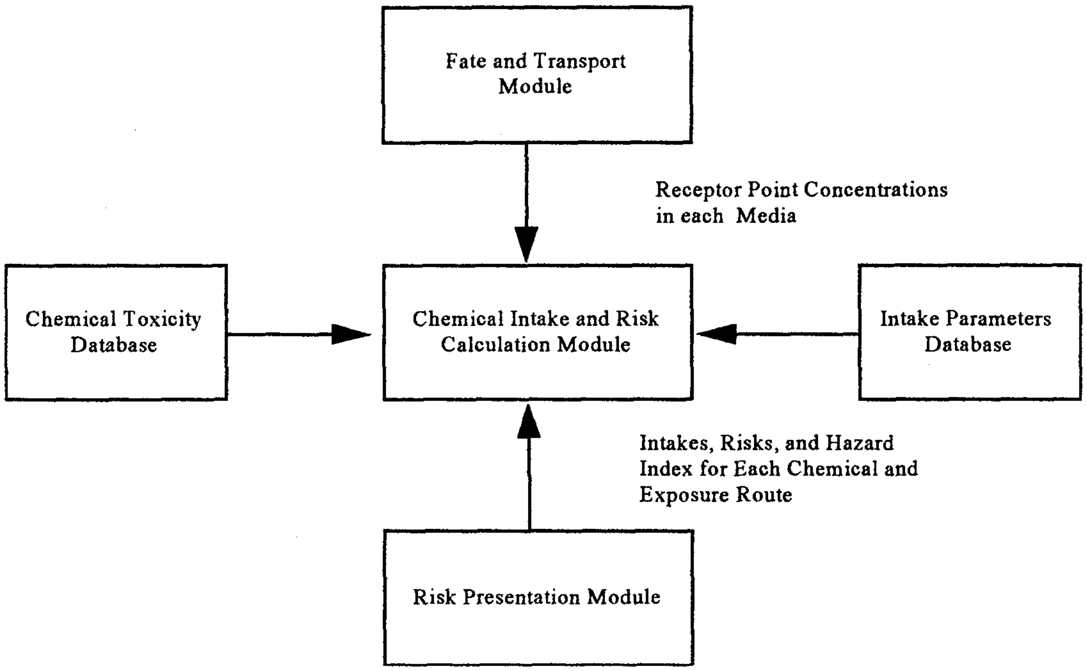


Figure 11:
RELATIONSHIP OF THE CHEMICAL INTAKE MODULE TO OTHER MODULES AND DATABASES [2:6-2]

2.11.2 General Limitations of DSS

Some distinct limitations of DSS are listed below:

For any one analysis, the maximum number of chemicals is limited to five.

The DSS can only consider one receptor per exposure route for each run.

Exposure routes such as the ingestion of produce grown in contaminated soil and the inhalation of indoor air in confined spaces are not addressed.

The EPA Biokinetic model for lead is not included in the software; DSS treats lead the same way as other chemicals requiring slope factor or reference dose.

Risks and chronic effects are additive.

Certain fate and transport models assume steady-state and/or compositional equilibrium between all phases at all times.

Risk is to the maximum exposed individual -- population risks are not addressed.

Risks are carcinogenic only -- teratogenicity, mutagenicity, etc., are not considered.

It is assumed that potency factors, RFDs apply to children as well as adults.

The fate and transport models can only simulate a single source, e.g. the only contaminated medium at time = 0 is the soil [2:3-11,12].

2.11.3 Model Descriptions

The DSS contains modules that allow the user to develop a conceptual model of the site under study through the selection of chemicals of concern, uptake routes, and specific fate and transport models to estimate receptor point concentrations at the site.

2.11.3.1 Jury Unsaturated Zone Model

Concentrations in the surface soil over time can be estimated using the SESOIL or the Jury model [2:5-5]. The SESOIL model, however, is quite complex dividing an assumed heterogeneous unsaturated zone into layers for analysis. It also requires input parameters not normally measured at sites nor easily estimated [7].

The model description that follows is found (in greater detail) in the American Petroleum Institute Risk/Exposure Assessment Decision Support System (DSS) Version 1.0 (user's guide), Appendix B.

The Jury model, Jury et. al. (1990), is a screening level model that estimates the chemical flux volatilizing from soil and the time-varying concentration profile within the unsaturated zone. The latter can be used to estimate the contaminant mass loading to the water table. This model is based on the analytical solution to the differential mass balance equation and the boundary and initial conditions presented below:

$$(\partial C_T/\partial t) + \mu C_T = D_E(\partial^2 C_T/\partial z^2) - V_E(\partial C_T/\partial z) \qquad (2)$$

where

C_T = total soil concentration (mg of contaminant/cm^3 of wet soil)

t = time (day)

μ = first order decay rate constant (1/day)

D_E = effective diffusion coefficient (cm^2/day)

z = depth measured positive downwards from the soil surface (cm)

V_E = effective contaminant velocity (cm/d).

The initial condition is:

$$C_T(0 < z < L, t=0) = C_o$$

$$C_T(z >= L, t = 0) = 0. \qquad (3)$$

These initial conditions imply that initially the contaminant is uniformly incorporated to a depth L. For the case of a contaminant source buried under a clean fill, the solution is obtained by superposition.

The upper boundary condition is:

$$-D_E\, \partial C_T/\partial z + V_E C_T = -H_E C_T \qquad \text{at } z = 0. \qquad (4)$$

This condition simulates the volatilization of chemical vapor to the atmosphere through a stagnant air boundary layer above which the chemical concentration in air is assumed zero.

The lower boundary condition is:

$$C_T(z = \alpha\,, \text{all } t) \quad = \quad 0 \tag{5}$$

where

C_o = initial total contaminant concentration in soil (mg/cm^3)

L = initial depth of contaminant concentration (cm)

H_E = a mass transfer coefficient variable (cm/day).

The total soil concentration is assumed to be distributed between the solid, aqueous and the vapor phase and is estimated using:

$$C_T = \rho_b C_s + \theta_w C_l + \theta_a C_g \tag{6}$$

where

ρ_b = the bulk density of soil (g of dry soil/cm^3 of wet soil)

C_s = the adsorbed phase concentration (g of contaminant/g of dry soil)

θ_w = the volumetric water content (cm^3 of water/cm^3 of wet soil)

C_l = the dissolved phase concentration (g contaminant/cm^3 solution)

θ_a = the air porosity (cm^3 gas in soil/cm^3 wet soil)

C_g = the dissolved phase concentration (g contaminant/cm^3 gas in soil).

This precludes the use of this model in situations where immiscible liquids or precipitates are present in the source zone [2:B-2].

The three individual phase concentrations C_s, C_l, and C_g are related by partition coefficients as follows:

$$C_s = K_d C_l$$

$$C_g = H C_l \tag{7}$$

where

K_d = the chemical-specific soil-water partition coefficient $= K_{oc} f_{oc}$

H = the dimensionless form of the Henry's constant.

Assumptions underlying the Jury model are as follows: (1) the soil column is assumed to be homogeneous and isotropic without any variations with depth, (2) the infiltration rate is assumed to be uniform and steady, (3) the contaminant is initially incorporated uniformly from the top of the soil column to a depth 'L' cm below the surface or may be incorporated in a thickness 'D' below a clean soil layer, (4) contaminant decay is assumed to follow first order decay rate, (5) the partitioning of contaminant concentrations between the three phases, i.e., solid phase, dissolved aqueous phase and the vapor phase is assumed to be linear and compositional equilibrium among phases is assumed at all locations at all times. Depending on the degree of departure between the field situation and these assumptions, the Jury model may overestimate or, in some cases, underestimate the emission rate [2:B-10].

2.11.3.2 AT123D (Saturated Zone) Model

AT123D is a semi-analytic model that uses site-specific information related to the contaminant source, saturated zone, and the distance to the receptor well to estimate concentrations in the well. A key input parameter for AT123D is the contaminant mass or the mass loading that defines the source for the AT123D model. If the AT123D model is run in conjunction with an unsaturated zone model ((SESOIL or JURY), the mass loading is internally estimated by the DSS as the sum of the advective and dispersive loadings.

When the AT123D model is run independently (e.g., when the water table is very shallow or an underground storage tank leaks directly into the water table) the mass loading is a user specified variable. The model allows three different options: (1) constant mass release rate for the entire simulation; (2) constant mass release rate for a specific period of time (pulse) with the mass release going to zero after release rate period; or (3) instantaneous release [2:5-5].

The output of the AT123D model is a depth-averaged concentration at a specified point in the aquifer that is used to estimate intake due to (i) ingestion of groundwater and (ii) dermal contact with water. This concentration is also used as an input to the shower model to estimate the amount of volatile emissions [2:5-5].

The DSS employs an approximate method to estimate average well concentrations. Concentrations are estimated by the AT123D at a user-specified number of equally-spaced vertical intervals across the well screen; these are arithmetically averaged to compute the well concentration. This method assigns equal weight to all vertical intervals between the top and bottom of the screen, and does not account for well water drawn from below or above the well screen or for lower concentrations in well water drawn radially from the downgradient side of the well [2:C-9]. Also of note is that the model estimates concentrations in a monitoring well, not a pumping well. The concentration in a pumping water well is significantly lower than concentrations in a monitoring well due to spatial dilution effects [2:C-7].

The AT123D, developed by Yeh (1981), is a screening-level analytical model that has been used for several applications related to the evaluation of risk at hazardous waste sites. The model is also included in EPA's Graphical Exposure Modeling System (General Sciences Corporation, 1990). AT123D solves the differential mass balance (advective-dispersive) equation that describes contaminant fate and transport in the saturated zone. Groundwater flow is assumed to be one-dimensional, steady, and uniform in the downgradient direction. Processes simulated by AT123D include: three-dimensional dispersion (longitudinal, lateral, and vertical); one dimensional uniform advection; linear, reversible equilibrium adsorption; and lumped first order decay (e.g., by biodegradation or other chemical transformation processes). The saturated zone is assumed to be homogeneous and isotropic in terms of its physical properties (porosity, bulk density, etc.). Chemical sources may be simulated as instantaneous injections of mass, or as time-varying mass rates into the groundwater system. The saturated zone may be infinite or finite in the lateral and vertical directions. The model has the ability to simulate a variety of contaminant source geometries oriented in different ways along the x, y and z directions. These can be used to estimate concentrations as a function of time at any location specified by different values of the x, y, z and t coordinates [2:C-1,3].

The AT123D model is based on the following assumptions: (1) the saturated zone is assumed to be homogeneous, isotropic and of uniform geometry, i.e., the thickness and width of the zone are considered to be uniform, (2) the water table is assumed to be steady without any fluctuations, (3) the flow direction is uniform, one-dimensional and steady-state, (4) contaminant decay is assumed to follow a lumped first order decay rate, (5) contaminant adsorption is considered to follow linear adsorption, (6) concentration in the liquid and solid phase of the aquifer are assumed to be in equilibrium at all times, and (7) the AT123D model simulates the dissolved phase contaminants only and is not applicable to simulate the transport of free product [2:C-10,15].

A complete description of the AT123D model is found in Appendix C of the DSS user's guide.

2.11.3.3 Gaussian (Air Dispersion) Model

The mathematical expressions used to estimate air dispersions are complex - considering the degree to which wind speed, wind direction, and atmospheric stability can vary. These variables, along with complicating factors such as topography and wake effects, necessitate the use of computer models for the calculation of contaminant concentrations [11:211]. The EPA states that Gaussian dispersion models can and have been successfully applied to the types of sources encountered at Superfund sites [11:211].

DSS these concentrations can either be specified by the user based on site-specific concentration measurements or estimated using either the box or the Gaussian dispersion model. [2:5-6]. Chemical emission rate is one required input to the Gaussian model. The jury model can be used to estimate this volatile emission rate and can provide this estimate as an input to the Gaussian model. This is a semi-analytic model that directly estimates contaminant leachate to the water table and volatilization to the atmosphere [2:5-7].

2.11.4 Exposure Modeling and Risk Calculation Under the DSS

Two sets of default exposure data are available in the DSS. These are the (1) EPA-recommended reasonable maximum exposure assumptions, and (2) average most-likely intake parameter values derived from EPA's Exposure Factors Handbook (1989) and Risk Assessment Guidance for Superfund Manual (1989) [2:6-3]. Exposure assumptions used in this study are presented in the next chapter (section 3.3).

If the models or the inputs vary in time (e.g., one time fuel spill), the model output consists of the receptor point concentration that varies with time. For example, application of AT123D might result in a time varying concentration in the receptor well. In order to estimate exposure (dose), however, an average concentration over the exposure duration is required [2:5-7]. These average concentrations used to estimate the dose are calculated internally by the DSS in accordance with the procedure described in the following (also see Figure 12):

> "The time-varying concentration values are analyzed to estimate the maximum running average concentration for up to 75 years (maximum likely human exposure duration) at 5-year intervals. This results in 15 values, i.e.,

> maximum 5-year running average concentration, maximum 10-year running average concentration, etc. Note, the maximum 5-year average concentration does not necessarily correspond to the average concentration for the first five years simulated. Rather, it corresponds to the five years surrounding the point in time when the maximum concentration occurred . . . These 15 concentrations are used to estimate the specific value used for estimating the dose . . . ." [2:8-9, 5-7,8]

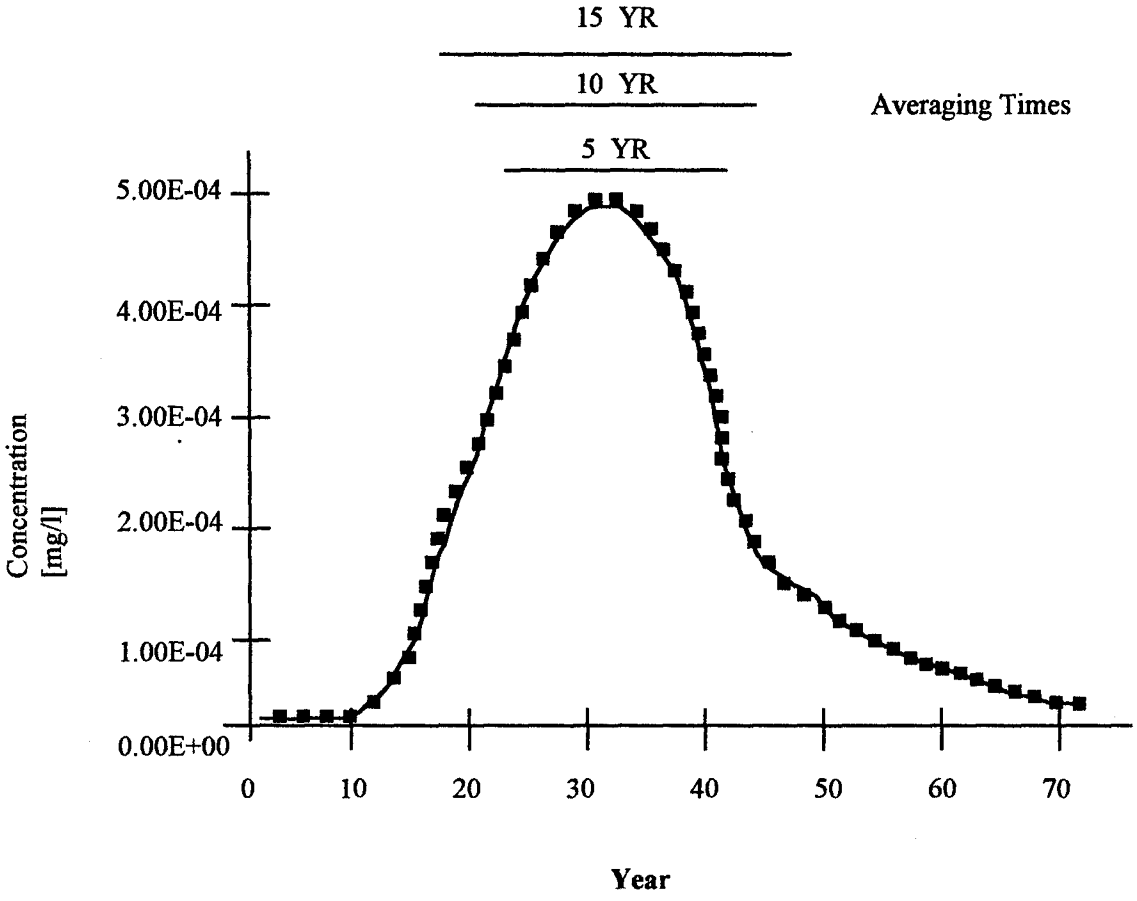


Figure 12: SCHEMATIC SHOWING THE ESTIMATION OF MAXIMUM AVERAGE CONCENTRATION FOR DIFFERENT AVERAGING PERIODS [2:8-9]

The final step in the risk assessment process is the risk calculation in which information on the toxicity of chemicals is combined with the intake estimates to compute the potential risk to human receptors [2:6-12]. The approach used by DSS to quantify risk is consistent with EPA's guidelines for hazardous waste sites as described in

(Part A) (1989). For each receptor, total risk and risk due to each chemical and each pathway of concern are estimated. Separate calculation methodologies are utilized by the DSS for evaluating carcinogenic risk and non-carcinogenic effects and each is described below.

For carcinogens, risk is quantified as the incremental probability of an individual developing cancer over a lifetime. Referred to as the individual excess lifetime cancer risk (IELCR), this risk is the excess risk incurred by individuals exposed to the chemicals by the pathways and routes found through the exposure assessment. This risk is quantified as the product of the slope factor (also referred to as the potency factor) and the lifetime average daily intake (LADD):

$$IELCR_{ij} = Sf_{ij}\ LADD_{ij} \tag{8}$$

where

$IELCR_{ij}$ = individual excess lifetime cancer risk for chemical i, exposure route j

Sf_{ij} = slope factor for chemical I, exposure route j $(mg/kg\text{-}day)^{-1}$

$LADD_{ij}$ = lifetime average daily dose for chemical I, exposure route j(mg/kg-day)

This approach to estimating risk is based on the linear low-dose cancer risk model described by the EPA, and is valid for risks below 0.01 (or 1 in 100 excess lifetime cancer risk, where the linear assumption of the low-dose model breaks down). DSS assumes that exposure to any amount of a carcinogen will increase the receptor's risk of developing cancer [2:6-12,13]. To estimate total risk to receptors exposed to multiple chemicals and exposure routes, cancer risk are assumed to be additive. This assumes there are no synergistic or antagonistic interactions between chemicals [2:6-13].

Non-carcinogenic effects are evaluated by assuming that there is a threshold level of exposure (i.e., the RfD) below which it is unlikely that sensitive populations will experience adverse health effects. These effects are characterized by comparing the chronic daily intake for each chemical and exposure route to the chemical and route-specific RfD. This risk is quantified as the ratio of the chronic daily intake to the RfD:

$$HQ_{ij} = CDI_{ij} / RfD_{ij} \qquad (9)$$

where

HQ_{ij} = the hazard quotient for chemical I, exposure route j

CDI_{ij} = the chronic daily intake for chemical I, exposure route j (mg/kg-day)

RfD_{ij} = the reference dose for chemical I, exposure route j (mg/kg-day).

The hazard quotient is an index of the potential for non-carcinogenic effects. If the value exceeds unity, there is the potential for adverse non-carcinogenic health effects [2:6-13].

For receptors exposed to multiple chemicals and exposure routes, the potential for non-carcinogenic effects is quantified by the hazard index (HI), equal to the sum of the hazard quotients for each chemical and exposure route [2:6-13]. The hazard quotient is the ratio of a single substance exposure level over a specified time period to a reference dose for that substance, whereas the hazard index is the sum of more than one hazard quotient for multiple substances and/or multiple exposure pathways [2:6-13].

3.0 Methodology

3.1 Background

As mentioned previously, the quantitative assessment using the API DSS required the focus of this research effort to be on petroleum contaminated sites, as the DSS was designed to assess risk posed from fuel spills or leaking fuel tanks. A search was made among Installation Restoration Program (IRP) sites locally at Wright-Patterson AFB and among Air Combat Command (ACC) IRP sites in both the Spill Site (SS) and the Storage Tank (ST) site categories. ACC sites were sought due to the centralized access to relevant site reports at HQ ACC/CEVR at Langley AFB. The search was further narrowed by selecting only those SS and ST sites that had both RREM evaluations and DPM scores. Candidate sites were eliminated further by selecting sites that were not contaminated with any type of chlorinated solvents. The DSS is not designed to model the transport of these solvents as they tend to sink and form isolated pockets in the lower portions of the groundwater aquifers rather than migrating in the uppermost portions making transport difficult to model.

Site specific data was gathered from the most current reports available at ACC/CEVR. In general, they included Remedial Investigations/Feasibility Studies (RI/FSs), Contamination Assessment Reports (CARs), Preliminary Assessments/Site Investigations (PA/SI), Remedial Design/Remedial Action (RD/RA) reports, contaminant data gathered from site Automated DPM (ADPM) files, and site data gathered (for some sites) from a query of the IRP Information Management System (IRPIMS) database. A complete list of data sources by site, along with a general site description, can be found in Appendix A.

3.2 Data Requirements and Estimation Methods

3.2.1 Unsaturated Zone

The data requirements for the Jury (unsaturated zone) model are provided in the Table 5. Chemical input requirements and estimation methods are discussed in a subsequent section. Soil column data were derived using the following procedures:

θ - Sources for soil volumetric water content included the IRPIMS (average values), site study reports, and in the case of the Wright-Patterson AFB sites, RPM estimation.

η_e - The effective porosity was gathered primarily from site reports or ADPM files.

ρ_b - The soil bulk density used was the DSS default of 1.8 g/cm^3 (considered a typical value.

f_{oc} - The fraction of organic carbon content was typically not available among site data. In these cases, 0.1% was used as a conservative measure as little or no retardation of the contaminants of concern would occur [7].

thickness of incorporation - This refers to the thickness of contaminated soil (assuming a uniformly contaminated "block" of soil). In most cases, this figure, along with the *X and Y-dimensions of the source* and the *thickness of the (clean) soil cover* at each site was provided in associated ADPM data files for each site studied. In cases where this data had not been measured, these data were estimated using cross sectional views of sites available in associated site study reports.

unsaturated zone depth - This was provided as the estimated depth to groundwater in each site report used.

infiltration rate - The infiltration rate was provided (as measured) in only a few site reports. In most cases, it was assumed that that rate of runoff was approximately 5% of the rainfall rate. The annual infiltration rate was then estimated by subtracting the annual runoff and the evaporation from the annual rainfall rate. Rates of evaporation and precipitation for Air Force installations can be found in the <u>DPM User's Manual, Appendix B</u>. For some of the sites located in desert climates, however, this relationship could not be used. In these cases, site reports stated that heavy rainfall often occurs over a very short period of the year. A small average annual infiltration rate (on the order of a few inches) was assumed in these cases.

boundary layer thickness - This refers to the thickness of a layer of stagnant air (above the soil surface) through which vapor must move by molecular diffusion [2:B-9]. The thickness, d, was calculated using the following equation [10:562]:

$$d = D^{air}_{WV}\, \delta^*_{WV}\, (1\text{-}RH) / 2E\delta_{WL} \qquad (10)$$

where

E = water evaporation rate (m/d)

RH = atmospheric relative humidity

D^{air}_{WV} = the binary diffusion coefficient of water vapor in air (2 m^2/d)

δ^*_{WV} = water vapor density (g/m^3)

δ_{WL} = is the liquid water density (g/m^3).

Average relative humidity data for each installation, if not provided in site reports, was gathered from the International Station Meteorological Climate Summary, Version 3.0, which is a database maintained by the Federal Climate Complex in Ashville, North Carolina. The day and night average values from this source were combined and averaged to arrive at the value used. The evaporation rates of each installation were obtained from the DPM User's Manual (1993). The value of $\delta^*_{WV}/\delta_{WL}$ ratio was estimated for use in the above equation by using values of d, E, and RH provided in sample reference calculations.

Table 5
DATA REQUIRED TO IMPLEMENT THE JURY MODEL [2:5-8]

Parameter Definition	Units	API DSS Units
Module Control Parameters		
Simulation Time	[yrs]	[yrs]
Chemical Data		
Total Soil Concentration	[mg/kg]	[mg/kg]
Diffusion Coefficient in Air	[cm^2/sec]	[cm^2/sec]
Diffusion Coefficient in Water	[cm^2/sec]	[cm^2/sec]
Henry's Law Constant	[m^3Atm/mole]	[(m^3/L)/(mg/L)]
K_{oc}	[ug/g oc/ug/ml]	[ug/g oc/ug/ml]
Overall Decay Rate	[1/day]	[1/day]
Solubility	[mg/l]	[mg/l]
Data for the Soil Column:		
Volumetric Water Content	[-]	[-]
Effective Porosity	[-]	[-]
Soil Bulk Density	[g/cm^3]	[g/cm^3]
Fractional Organic Carbon	[mg/mg]	[mg/mg]
Thickness of Incorporation	[m]	[m]
Thickness of Soil Cover	[m]	[m]
Depth of Unsaturated Zone	[m]	[m]
X-dimension of the source	[m]	[m]
Y-dimension of the source	[m]	[m]
Thickness of boundary layer	[cm]	[cm]
Infiltration Rate	[cm/sec]	[cm/yr]

3.2.2 Saturated Zone

The data requirements for the AT123D (saturated zone) model are provided in the Table 6. Inputs were derived using the following procedures:

η_e - The effective porosity was gathered primarily from site reports or ADPM files.

hydraulic conductivity (K) - Average K-values were obtained from associated site reports.

hydraulic gradient - These values were obtained from site reports or scaled from groundwater level contours (m).

longitudinal, transverse, and vertical dispersivity - Figure used were 10, 1, and 1 meters respectively as the actual measurements for these parameters did not occur in past site investigations and the assumed values are reasonable in many cases [7].

f_{oc} - The fraction of organic carbon content was typically not available among site data. In these cases, 0.1% was used as a conservative measure as little or no retardation of the contaminants of concern would occur [7].

ρ_b - For the soil bulk density the DSS default of 1.8 g/cm^3 was used (considered a typical value).

thickness of aquifer - With the exception of a few of the sites, the aquifer thickness was provided in site reports; otherwise it was scaled from cross-sectional maps or a conservative value of 10 feet was used to match the well screen length.

Aquifer width - The aquifers at each site were assumed infinite in extent as this figure was not provided in site reports.

Receptor well data - The well screen length used was 10 ft as this would be the approximate length of a private well designed to serve the needs of a single family dwelling [7]. To determine the distance of the receptor well from the site, data from the ADPM files on each site was used. The distance to the nearest receptor well was estimated through the following relationship:

distance = (groundwater travel time)(gradient)(hydraulic conductivity) / (porosity)

If the groundwater travel time provided in the ADPM files was large, indicative of no current groundwater receptors, the nearest residence was used under the scenario that a well could be constructed for use at that location. In some cases, the measured distance from a site to downgradient wells was provided in the ADPM data files.

Table 6
DATA REQUIREMENTS FOR AT123D MODEL INCORPORATED IN THE APIDSS [2:1-13]

Parameter Definition	Units/Options
Model Control Parameters	
Option to indicate if the media is infinitely wide (y-direction)	YES/NO
Option to indicate if the aquifer is infinitely deep (z-direction)	YES/NO
* Option to indicate the type of release/source	Click on selection
instantaneous release	-
constant release rate	-
pulse release rate	-
* Simulation time (max = 100 yrs)	(yr)
*** Source Parameters**	
Length in the x-direction	(m)
Length in the y-direction	(m)
Thickness in the z-direction	(m)
Duration of the pulse (only for pulse source)	(yr)
Instantaneous chemical release (only for instantaneous release case)	(kg)
Constant chemical release rate (only for constant release case)	(kg/yr)
Media Parameters	
Effective porosity	(cm^3/cm^3)
Hydraulic conductivity	(m/yr)
Hydraulic gradient	(m/m)
Longitudinal dispersivity	(m)
Transverse dispersivity	(m)
Vertical dispersivity	(m)
Bulk density of soil	(g/cm^3)
Fractional organic carbon content	(g/g)
** Thickness of aquifer	(m)
** Width of aquifer	(m)
***Chemical Parameters**	
Organic carbon partition coefficient	[(μg/g)/ (μg/ cm^3)]
Overall decay rate	(1/day)
Molecular diffusion coefficient	(cm^2/s)
Receptor Well Data	
x-coordinate of well	(m)
y-coordinate of well	(m)
z-coordinate of top of well	(m)
z-coordinated of bottom of well	(m)

* Not required where AT123D is run in conjunction with the SESOIL or JURY model.
** required only when aquifer is finite in z- or y- directions respectively

Note default values of chemical parameters are available from the database in the APIDSS.

3.2.3 Air

The data requirements for the Gaussian (air dispersion) model are provided in the Table 7. Inputs were derived using the following procedures:

area of source - This was determined from source dimensions provided in ADPM files or scaled from site maps.

emission rate - The emission rate was estimated as an output of the Jury model that serves as input to the dispersion model.

decay rate - Default values were used from the DSS chemical database.

distance from source to receptor - For the off-site receptor, this distance matched the distance to the receptor well assuming it was at an off-site residence; for the on-site worker and resident, a value of 1 meter was used, placing the receptors at the site in these two cases.

average wind speed - The average wind speed was obtained from site reports in most cases, and estimated from a mean annual wind speed map in the DPM User's Manual when not available in the site reports.

Stability class frequencies - These represent the fractions of time that winds are unstable/stable and they were obtained from measurements recorded at nearby airports or climatological stations. This data was obtained from files in Risk Assistant (1991), a microcomputer-based software system for risk analysis (developed by Hampshire Research Institute, Alexandria, Virginia).

Fraction of the time wind blows toward receptor - This could not be estimated and it was simply assumed that the wind blew toward the receptor half the time.

Vertical dispersivity - This was estimated internally using the input stability class data and the Pasquill-Gifford relationship [2:H-2,3].

Table 7
DATA REQUIREMENTS FOR THE GAUSSIAN DISPERSION MODEL [2:H-5]

	PARAMETER DEFINITION	UNITS	*APIDSS UNITS
Source Data			
	Area of source	[m^2]	[m^2]
Chemical Data			
	Emission rate	[g/s]	[kg/yr]
	First order decay rate in air	[1/s]	[1/s]
Receptor Data			
	Distance from source to receptor	[m]	[m]
Meteorologic Data			
	Mean annual wind speed	[m/s]	[m/s]
	Frequency of occurrence for each stability class	[-]	[-]
	Fraction of the time wind blows in the direction of the receptor from the source	[-]	[-]
	Vertical dispersivity **	[m]	[m]

* Conversions handled internally by the APIDSS
** Estimated based on empirical Pasquill-Gifford relationship

3.3 Site Chemicals of Concern

3.3.1 Contaminants of Concern

This study focused on the BTEX components of JP-4, a mixture of gasoline and kerosene distillates, as these have been the primary chemicals of concern in past DOD studies of petroleum contamination. The BTEX are among the VOC fuel constituents whose physical characteristics make them moderately to highly mobile in the environment. At most of the sites studied, sampling took place only for BTEX in the soils and/or groundwater. Another reason for focus on these fuel components is that for

RREM site evaluations on sites contaminated solely with POL products, only the BTEX compounds in each media, together with corresponding BTEX standards, are used to determine contaminant hazards associated with each site[15:17].

A breakdown (by percent mass) of the major components of JP-4 (Jet fuel, No. 4) is found in Table 8. Not listed among these are the Polynuclear Aromatic Hydrocarbons (PAHs), which are a large portion of the SVOCs in fuels that are of high molecular weights and found only in the parts-per-million range in JP-4 [7]. As SVOCs, the PAHs are less mobile and more persistent in the environment than the VOCs. Benzene, however, is usually the primary driver of risk at petroleum contaminated sites as it is more abundant and a more mobile carcinogen than the PAHs [7].

Table 8
MAJOR COMPONENTS OF JP-4 [17:32-33]

Fuel Component	Percent by Weight	Fuel Component	Percent by Weight
n-Butane	.012	p-Xylene	0.35
Isobutane	.066	3, 4-Dimethylheptane	0.43
n-Pentane	1.06	4-Ethylheptane	0.18
2,2-Dimethylbutane	0.10	4-Methyloctane	0.86
2-Methylpentane	1.28	2-Methyloctane	0.88
3-Methylpentane	0.89	3-Methyloctane	0.79
n-Hexane	2.21	o-Xylene	1.01
Methylcyclopentane	1.16	1-Methyl-4-ethylcyclohexane	0.48
2,2-Dimethylpentane	0.25	n-Nonane	2.25
Benzene	0.50	Isopropylbenzene	0.30
Cyclohexane	1.24	n-Propylbenzene	0.71
2-Methylhexane	2.35	1-Methyl-3-ethylbenzene	0.49
3- Methylhexane	1.97	1-Methyl-4-ethylbenzene	0.43
trans-1,3-Dimethylcyclopentane	0.36	1, 3, 5-Trimethylbenzene	0.42
cis-1, 3-Dimethylcyclopentane	0.34	1-Methyl-2-ethylbenzene	0.23
cis-1,2-Dimethylcyclopentane	0.54	1, 2, 4-Trimethylbenzene	1.01
n-Heptane	3.67	n-Decane	2.16
Methylcyclohexane	2.27	N-Butylchclohexane	0.70
2,2,3,3-Tetramethylbutane	0.24	1, 3-Diethylbenzene	0.46
Ethylcyclopentane	0.26	1-Methyl-4-propylbenzene	0.40
2,5-Dimethylhexane	0.37	1, 3-Dimethyl-5-ethylbenzene	0.61
2,4-Dimethylhexane	0.58	1-Methyl-2-i-propylbenzene	0.29
1,2,4-Trimethylcyclopentane	0.25	1, 4-Dimethyl-2-ethylbenzene	0.70
3,3-Dimethylhexane	0.26	1, 2-Dimethyl-4-ethylbenzene	0.77
1,2,3-Trimethylcyclopentane	0.25	n-Undecane	2.32
Toulene	1.33	1, 2, 3, 4-Tetramethylbenzene	0.75
2,2-Dimethylhexane	0.71	Naphthalene	0.50
2-Methylheptane	2.70	2-Methylundecane	0.64
4-Methylheptane	0.92	n-Dodecane	2.00
cis-1,3-Dimethylcyclohexane	0.42	2, 6-Dimethylundecane	0.71
3-Methylheptane	3.04	Unidentified	0.68
1-Methyl-3-ethylcyclohexane	0.17	2-Methylnaphthalene	0.56
1-Methyl-2-ethylcyclohexane	0.39	1-Methylnaphthalene	0.78
Dimethylcyclohexane	0.43	n-Tridecane	1.52
n-Octane	3.80	2, 6-Dimethylnaphthalene	0.25
1, 3, 5-Trimethylcyclohexane	0.99	n-Tetradecane	0.73
1, 1, 3-Trimethylcyclohexane	0.48		
2, 5-Dimethylheptane	0.52		
Unidentified	0.98		
Ethylbenzene	0.37		
m-Xylene	0.96		

The percent mass (in gasoline) of the following PAHs and gas additives were determined. These are among the chemicals in the DSS database and the figures below were used to approximate the percent mass of these components in JP-4 [1:914, 17:33, and 12:2108]:

Benzo(b)fluoranthene - 3.9 mg/L

Ethylene Dichloride - 210 mg/L

Ethylene Dibromide - 190 mg/L

Naphthalene - 0.5% by weight (in JP-4)

Pyrene - mole fraction of 1E-6 (used with molecular weights to arrive at mass fraction)

Benzo(a)pyrene - mole fraction of 1E-7.

The percent mass of the following chemicals in JP-4, also among the chemicals in the DSS database, were used to estimate the mass of each of these components in the original spill [17:32]:

Benzene - 0.5%

Toluene - 1.33%

Ethylbenzene - 0.37%

Xylenes - 2.32%.

The toxicities and carcinogenic classifications of the chemicals addressed at each site in this study are shown in Table 9.

Table 9
CHEMICAL TOXICITY DATA FROM DSS DATABASE [2:9-11,12]

Chemical	**Classification (1)**	**Inhalation RfD (mg/kg-d)**	**Oral RfD (mg/kg-d)**
Benzene	A	Pending	Pending
Ethylbenzene	D	2.86E-01	0.1
Toluene	D	1.14E-01	0.02
Xylenes (mixed)	D	0.2	2
Benzo (a) pyrene	B2	ND	ND
Benzo (b) floranthene	C	ND	ND
Naphthalene	D	ND	0.04
Pyrene	D	ND	0.03
Ethylene Dibromide	B2	ND	ND
Ethylene Dichloride	B2	ND	ND

ND = No Data

Note: 1.

Classification	Description
A -	Human Carcinogen
B1 -	Probable human carcinogen, based on limited human data
B2 -	Probable human carcinogen, based on sufficient evidence in animals and inadequate evidence in humans
C -	Possible human carcinogen
D -	Not classifiable as to human carcinogenicity
E -	Evidence of non carcinogenicity for humans

Separate DSS model runs were made with BTEX contamination only (benzene being the only carcinogen) and then with the PAHs and gasoline additives listed previously so that comparisons could be made with current DOD models considering BTEX contamination only (emulating the DOD models) and then under a potentially more threatening contamination scenario involving all 10 contaminants of interest.

3.3.2 Contaminant Concentrations and Mass Determinations

3.3.2.1 BTEX

Based on the original estimated JP-4 spill quantity and the percent mass of each of the BTEX constituents at each site, the maximum mass of BTEX possible at each site was calculated. An average density for gasoline of 0.74 g/mL was used to estimate the mass of JP-4 at each site from reported spill volumes [1:917]. Soil maximum contaminant concentration measurements and the estimated volume of contaminated soil (both available at most of the sites) were used to calculate the actual mass (of BTEX) currently at the site. At three sites where this mass exceeded the maximum possible (using benzene, the only carcinogen among the BTEX, as the threshold chemical constituent) based on the original spill, the areas of these sites were reduced until the mass matched the maximum that would be found immediately after the fuel release. Where maximum soil contaminant concentrations were not available or were uncertain, the original spill mass was spread evenly over the entire estimated contaminated volume to arrive at average concentration measures for the BTEX constituents. Where groundwater contaminant concentrations were available, the mass of contaminant in the groundwater was estimated based on the maximum aqueous concentrations given and the plume size (given or scaled from site drawings in associated reports) and assuming the concentration is evenly distributed throughout the first foot of groundwater [7]. Where groundwater concentrations were not available, average expected aqueous concentrations were calculated for BTEX constituents based on average (experimentally measured) aqueous phase concentrations for BTEX components from gasoline (Cline and others,1991) and were multiplied by the estimated plume volume to estimate the mass of BTEX in groundwater [1:917]. These groundwater contaminant mass estimates were input into the DSS groundwater model (AT123D) and the associated risk calculated was added to that associated with current vadose zone contamination.

3.3.2.2 PAHs/Gas Additives

Of interest are some of the SVOCs that could be present as the result of JP-4 spills or leaks as these might contribute to excess lifetime cancer risk under the exposure scenarios considered. The percent mass of each of these components normally found in gasoline was used to estimate what might be expected in the original JP-4 spill quantities. Since sampling for these materials did not occur during site characterization, the maximum possible mass that could be at each site (based upon the original spill quantity) was used and spread over the entire contaminated region of soil to provide an estimation of the soil concentration for entry into the DSS. This method likely overestimated both the actual mass present (e.g., some would volitalize or degrade) and the spread of these contaminants as they are less mobile than the VOCs. However, because they are less mobile and more persistent, a more isolated area of contamination at higher concentrations than were estimated in this study could be present. The DSS was run assuming the contaminants from these fuel components were in the vadose zone only and the results were added to previous BTEX results as an extension of the DSS risk estimation for each site (except where contaminated soil had been removed and only groundwater contamination now exists, in which case BTEX were the only contaminants considered).

3.4 Summary of Model Applications

As mentioned previously, a limitation of the DSS is that the fate and transport models can only simulate a single source, e.g., the only contaminated medium at time = 0 is the soil (unsaturated zone). However, at most of the contaminated sites, this is not the existing site condition as contamination has reached groundwater. Also, at a few sites it appeared (as discussed in relevant site reports) that contaminated soils had been removed but groundwater contamination remained.

To more closely reflect existing site conditions and model site risks, risk associated with the current unsaturated zone contamination was determined by running the Jury (Vadose zone) model. This model was linked to both the AT123D (saturated zone) and Gaussian dispersion models allowing for migration of the contaminant from the vadose zone into the groundwater and the atmosphere (through volatilization) to be characterized.

In sites where current groundwater contamination also existed, the AT123D model was run separately (standalone run) with the input being a simulated "instantaneous release" of contamination spread over a plume (plume size estimated from scaled drawings or placed directly under vadose zone contamination) where the contaminants were assumed to be in the top one foot of groundwater. The risk associated with this portion of the contaminant mass was then added to that previously calculated from the vadose zone contamination to arrive at a total risk estimate (excess cancer risk) posed by the site if no cleanup efforts were undertaken.

3.5 Exposure Scenarios

As discussed previously, consideration of both current risk scenarios and potential future risk scenarios is the recommended approach of EPA risk assessment guidance. Current risk scenarios considered were for both an on-site worker and for an off-site resident child receptor. The future risk scenario evaluated is that of an on-site resident child receptor. Because residential land use is most often associated with the greatest exposures, it is generally the most conservative choice to make when deciding what type of alternate land use may occur in the future [20:6-7]. Each scenario, as well as exposure parameters used, is discussed below.

Under the on-site worker current risk scenario, the parameters entered into the intake parameter database are shown in the first column of Table 10. With the exception of the soil ingestion rate (where the most likely case value seemed most appropriate for an adult receptor), parameters used were default maximum exposure case values found in the DSS database for the adult worker. These values were derived from the EPA's Superfund Exposure Assessment Manual (1988) and are in line with the conservative approach recommended by EPA risk assessment guidance. The site-specific time outdoors parameter was conservatively estimated as 7 hr/day considering that most sites in this study were storage tank farm areas and a seemingly worst case scenario was that most of the normal work day was spent on site.

Under the off-site resident current risk scenario, the parameters entered into the intake parameter database are shown in the second column of Figure 10. With the exception of a few cases (discussed below) parameters used were default maximum exposure case values found in the DSS database for the child resident. As in the off-site receptor case, these values were derived from the Superfund Exposure Assessment Manual and are in line with the conservative approach recommended by EPA risk assessment guidance. The site-specific time outdoors parameter was conservatively estimated as 8 hr/day considering a young child spending several hours of each day involved in outdoor play. In addition, the inhalation rate was increased to reflect a child involved in moderate activity as this is the recommended measure provided in the Superfund Exposure Assessment Manual [21:123]. Both the total skin surface area and skin surface area (arms and hands) were reduced (from adult values) to reflect that of the average child receptor [21:127].

Under the on-site resident future risk scenario, the parameters entered into the intake parameter database are shown in the last column of Figure 10. These are identical to the parameters found in the second column that were used for the off-site child receptor.

Table 10
INTAKE PARAMETERS DATABASE [2:6-5,6]

Parameter	Units	On-site Worker	On-site Child	Off-Site Child
Common to all Routes				
Body Weight	[kg]	70	15	15
Lifetime	[years]	70	70	70
Ingestion of Drinking Water				
Exposure Duration	[years]	30	30	30
Exposure Frequency	[days/year]	365	365	365
Ingestion Rate	[liters/day]	2	1.4	1.4
Bioavailability	[mg/mg]	Chemical-Specific		
Inhalation During Shower				
Exposure Duration	[years]	30	30	30
Exposure Frequency	[days/year]	365	365	365
Exposure Time	[hours/day]	0.333	0.333	0.333
Inhalation Rate	[m^3/hour]	0.89	0.89	0.89
Bioavailability	[mg/mg]	Chemical-Specific		
Dermal Exposure in Shower				
Exposure Duration	[years]	30	30	30
Exposure Frequency	[days/year]	365	365	365
Exposure Time	[hours/day]	0.333	0.333	0.333
Total Skin Surface Area	[cm^2]	18150	9400	9400
Permeability Coefficient	[cm/hour]	Chemical-Specific	-	-
Inhalation of Volatile Soil Emissions				
Exposure Duration	[years]	30	30	30
Exposure Frequency	[days/year]	365	365	365
Time Outdoors	[hours/day]	7	8	8
Inhalation Rate	[m^3/hour]	1.3	2.1	2.1
Bioavailability	[mg/mg]	Chemical-Specific	-	-
Dermal Contact with Soil				
Exposure Duration	[years]	30	30	30
Exposure Frequency	[days/year]	365	365	365
Skin Surface Area (arms and hands)	[cm^2]	3120	1615	1615
Adherence Factor	[mg/cm^2]	1.45	1.45	1.45
Absorption Coefficient	[mg/mg]	Chemical Specific	-	-
Ingestion of Soil				
Exposure Duration	[years]	30	30	30
Soil Contact Frequency	[days/year]	365	365	365
Soil Ingestion Rate	[mg/day]	10	200	200
Fraction Soil Contaminated	[mg/mg]	1	1	1
Bioavailability	[mg/mg]	Chem-Specific	-	-

3.6 Model Outputs

Though the DSS can provide cancer risk by route and/or by chemical and the hazard index (HI) by chemical and/or by route, the output of interest is the total cancer risk posed by the site and the total HI for the site. Cancer risk is the driver for site ranking as there is zero threshold for intake of carcinogenic chemicals. In other words, any dose of a known carcinogen can increase the receptor's risk of developing cancer, whereas a threshold dose for noncarcinogens exists below which no ill-effects are expected. Very low doses received through exposure to carcinogens are of great concern, but the HI serves more as an indicator of an immediate hazardous condition at a single site indicating (if evaluated at one or greater) a need for immediate attention. The HI, therefore, would not serve as an appropriate indicator of cleanup priority among several sites as the cancer risk associated with the sites does.

Some pathway outputs were not considered as valid exposure routes under both the on-site worker and the off-site child receptor scenarios and were, therefore, not added into the total cancer risk. For example, in the case of the on-site worker, the groundwater ingestion routes and showering exposure routes were not considered appropriate as it was assumed that water was consumed/used off-site. In the off-site child receptor scenario, the soil ingestion and dermal contact pathways were eliminated as routes as the child would not have access to the site under this scenario.

Predetermined outputs from the models under study include site DPM scores (0 to 100), RREM media (risk categories for soil, groundwater, surface water, and sediment evaluations) and RREM site evaluations (high, medium, or low risk category for entire evaluations) and RREM site evaluations (high, medium, or low risk category for entire site). Sites were ranked from what was estimated as highest to lowest risk by each of the three models. The comparative analysis of these rankings was accomplished according to the procedure outlined in the following section (output analysis).

As site RREM evaluations result in placing a site in the high, medium, or low category, a procedure was needed to distinguish between sites within each category in order to develop a complete ranking of the sites. To rank order the sites within each category, the media level categories were observed, and the site of the same overall category (e.g., medium) with another but with more medias at the same (medium) category was considered higher in priority (e.g., a overall medium with two medium media categories is higher priority than a overall medium with one medium media category). It was also assumed that a low in a media category is higher risk than a media category that was not evaluated, as not evaluated would indicate the particular media does not provide a viable pathway or it does not exist at a site (e.g., contaminated soils removed, or no nearby surface water, etc.).

3.7 Output Analysis Methods

As discussed above, the data to be analyzed are rankings of the same group of sites accomplished by the three site ranking methods used. Of interest is a measure of how these ranking methods agree with one another and a test of the hypothesis that there is no association between the methods (with an associated test significance level). A measure of the association of all three ranking methods can be made through use of the Kendall Coefficient of Concordance [6:300-301]. A pairwise measure (e.g., measure of association between DSS and RREM) can be made using the Spearman Coefficient of Rank Correlation [6:275]. These methods are discussed below with an example.

3.7.1 Kendall Coefficient of Concordance for Complete Rankings

Under the study scenario, the data to be analyzed consist of k complete sets of rankings of n objects. Each ranking is made by a comparison of objects within that set only. The data as collected or observed consist of nk ranks or measurements on at least an ordinal scale. In the simplest case, the same group of n objects is presented to each of k observers; and the observers score or rank each of the objects according to some criterion. An example is a set of rankings of n beauty contestants by a panel of k judges. A measure of the agreement between the k sets of rankings of these k related measurements describes the association between the k characteristics. This quantity, called the Kendall coefficient of concordance, can also be used to test the null hypothesis that the k characteristics are independent.

The important thing about the data is that the comparisons that are of interest, are relevant, or make sense, are comparisons within each of the k groups or characteristics. Hence, the rankings are assigned separately within each of the k groups.

An example is provided below of a beauty contest where each member of a panel of three judges is asked to rank the eight contestants [3:302-304]:

	Contestant								
Judge	A	B	C	D	E	F	G	H	Sum
1	2	1	3	5	4	8	7	6	36
2	1	2	4	5	7	6	8	3	36
3	3	2	1	7	5	8	6	4	36
Sum R_j:	6	5	8	17	16	22	21	13	108

If there were no agreement between the three sets of rankings, the column totals would be approximately equal for each contestant. If they were exactly equal, each would sum to 13.5, the expected value of each column sum if there is no association, or, in other words, the judges are assigning ranks independently. The following is a measure of the departure from equal column sums

$$S = (6 - 13.5)^2 + (5 - 13.5)^2 + (8 - 13.5)^2 + \ldots + (13 - 13.5)^2 = 306.00.$$

The sum of the squares of deviations of these column sums (with perfect agreement) from the column sums expected (with no association) is

$$(3 - 13.5)2 + (6 - 13.5)2 + (9 - 13.5)2 + \ldots + (24 - 13.5)2 = 378.0.$$

The relative agreement can be measured by the ratio of the actual sum of squares to the sum of squares under perfect agreement, or

$$(306.00) / (378.00) = 0.810.$$

The ratio formed in this way would equal 1 for perfect agreement and tend toward 0 for no agreement. It can be shown that the above ratio is equivalent to the following expression involving k sets of n ranks:

$$W = (12 \Sigma_{j=1}^{n} R_j^2 - 3k^2n(n+1)2) / nk^2(n^2 - 1) \qquad (11)$$

where R_j = the column sums (as in the previous example) [6:304].

This is the relative measure of association, or the degree of communality or agreement, which is called the Kendall coefficient of concordance.

If any of the observations within a set to be ranked are tied, the mid-rank method should be used to assign ranks. This requires replacing the denominator of W in the Equation 11 above by:

$$k^2n(n^2 - 1) - k(\Sigma t^3 - \Sigma t)$$

where

t is the number of observations tied at any rank in any set of rankings, and the summation is over all sets of t tied ranks and all k sets of rankings [6:305].

For any k sets of rankings, the Kendall coefficient can be used to test the null hypothesis of no association between rankings. The hypothesis is written as

H_o: No association $\qquad H_A$: Association exists.

If no association exists (that is, if the null hypothesis is true), then the ranks have been assigned independently to the n objects for each of the k sets of rankings, and the column sums tend toward equality, so W is close to 0. If there were association, then most of the large ranks would appear together in the same columns, and small ranks would also be clustered in the same columns. This results in W becoming close to 1. Accordingly, large values of W call for rejection of H_o in favor of the alternative H_A [6:305-6].

The Kendall coefficient of concordance W may be used as a test statistic. The test statistic that is easiest to use is Q as calculated from

$$Q = k\,(n - 1)\,W. \tag{12}$$

Since the test statistic Q is a monotonically increasing function of W, and specifically Q is large when W is large and zero when W is zero, the appropriate P-value is the right-tail probability. The distribution of the test statistic Q can be approximated by the chi-square distribution with n - 1 degrees of freedom when large values of n and k preclude the use of tables for the statistic Q. A small P-value (i.e., $P < .02$) indicates that the data do not support the null hypothesis of independence (association implying a lack of independence), and we conclude that there is a relationship between the k rankings (e.g., the rankings of the three judges in the previous beauty contest example above) [6:306].

3.7.2 Spearman Coefficient of Rank Correlation

The Spearman coefficient of rank correlation is defined as

$$R = 1 - (6 * \Sigma D_i^2) / n*(n^2-1) \tag{13}$$

where D_i is the difference between any two paired ranks. When $|R| = 1$ it reflects perfect association, and the sign of R indicates the type or direction of association [6:277]. Perfect agreement means that large values of one variable are associated with large values of the other (R close to one), whereas perfect disagreement is indicated by an inverse association (R close to -1). When R = 0, there is no association.

Using only the top two rows (judges) of the previous example:

$$\Sigma D_i^2 = (1)2 + (-1)2 + \ldots + (-1)2 + (3)2 = 26.$$

and

$$R = 1 - 6(26) / 8(63) = 0.690$$

indicating a direct relationship (positive) and a relative measure of association between rankings of the judges of 0.690.

The null hypothesis that can be tested in this case is, as in the case of comparing the three methods at once, that the variables that measure the two characteristics in the population are independent. However, in this case, the hypothesis may be any one of the following:

Two-sided alternative

H_o: No association H_A: Association exists

One-sided alternative

H_o: No association H_{A+}: Direct association

H_o: No association H_{A-}: Direct association with the test statistic being the Spearman coefficient of rank correlation (R) and the P-value being found in tables for associated R-values [6:279-80]. The equation for R noted above may also be modified slightly to account for ties among ranks.

4.0 RESULTS

4.1 Data Summaries and Analysis

As discussed in the previous chapter, a measure of association between the three ranking methods is desired. Such a measure could indicate, at least for the sites under study, how closely the site rankings compare between the strictly quantitative DSS model, the semi-quantitative DPM, and the primarily qualitative approach of the RREM. This could serve as at least an initial indicator of the validity of the DOD site ranking approaches for this class of contaminated sites.

Tables 11 through 13 below show the DPM site scores, the RREM risk categories, and the DSS excess lifetime cancer risk levels for each site as well as the rankings produced by each method.

Table 11
DPM AND RREM RANKS

Site ID/Location	Short Hand Designator	DPM Score	DPM Ranks	Overall RREM	RREM Sub Categories				RREM Ranks
					GW	SW	S	Sed	
ST-11 / Minot	A	0.2	15	L	-	-	L	-	14.5
ST-02 / Pope	B	2.5	10	M	M	-	L	-	9
SS-24 / Little Rock	C	3.2	9	M	M	-	-	-	12
SS-27 / Wright-Pat	D	39.2	2	H	M	L	H	-	1.5
SS-63 / Wright-Pat	E	30.9	3	M	M	L	M	-	5
SS-28 / Wright-Pat	F	39.6	1	H	M	L	H	-	1.5
SS-10 / Ellsworth	G	27.0	5	M	M	-	L	-	9
SS-08 / Ellsworth	H	27.7	4	M	M	-	L	-	9
ST-14 / Ellsworth	I	25.0	6	L	-	-	L	-	14.5
SS-20 / Moody	J	1.1	13	M	M	-	-	-	12
ST-18 / Moody	K	2.0	11	M	M	-	-	-	12
ST-12 / Moody	L	4.5	8	H	H	-	L	-	4
ST-15 / Moody	M	1.4	12	H	H	-	M	-	3
SS-30 / Nellis	N	0.9	14	M	M	L	L	L	6.5
ST-27 / Nellis	O	21.4	7	M	M	L	L	L	6.5

Table 12
DSS CURRENT AND FUTURE EXCESS LIFETIME CANCER RISKS AND RANKS (BTEX)

	Current Risk Scenario Receptors				Future Risk Scenario Receptor	
Sites	**On-Site Worker**	**Off-Site Resident Child**	**Highest Risk**	**Rank**	**On-Site Child Resident**	**Rank**
A	4.8 E-8	2.94 E-14	4.80 E-8	14	3.75 E-4	11
B	1.09 E-11	3.1 E-8	3.10 E-8	15	1.89 E-7	15
C	1.15 E-4	2.84 E-6	1.15 E-4	6	7.27 E-3	3
D	1.28 E-5	3.3 E-5	3.30 E-5	9	2.65 E-4	12
E	6.09 E-6	1.78 E-5	1.78 E-5	10	2.44 E-4	13
F	1.08 E-4	2.46 E-4	2.46 E-4	4	4.17 E-3	5
G	1.21 E-4	6.13 E-4	6.13 E-4	1	8.69 E-2	1
H	4.73 E-4	4.76 E-4	4.76 E-4	2	8.34 E-3	2
I	3.08 E-4	1.77 E-5	3.08 E-4	3	2.84 E-3	6
J	1.49 E-6	4.86 E-12	1.49 E-6	13	3.94 E-5	14
K	5.16 E-6	1.68 E-11	5.16 E-6	11	1.29 E-3	9
L	3.69 E-5	1.2 E-10	3.69 E-5	8	8.23 E-4	10
M	1.21 E-4	3.93 E-10	1.21 E-4	5	1.59 E-3	8
N	2.85 E-6	1.53 E-10	2.85 E-6	12	2.43 E-3	7
O	1.13 E-4	6.49 E-5	1.13 E-4	7	4.4 E-3	4

Table 13
DSS CURRENT AND FUTURE EXCESS LIFETIME CANCER RISKS AND RANKS (BTEX, PAHs, GAS ADDITIVES)

	Current Risk Scenario Receptors				Future Risk Scenario Receptor	
Sites	**On-Site Worker**	**Off-Site Resident Child**	**Highest Risk**	**Rank**	**On-Site Child Resident**	**Rank**
A	3.74 E-6	3.75 E-14	3.74 E-6	13	2.99 E-1	3
B	3.85 E-5	1.81 E-1	1.82 E-1	1	1.36 E-1	5
C	1.17 E-4	3.38 E-10	1.17 E-4	10	6.96 E+0	1
D	2.31 E-5	9.54 E-2	9.54 E-2	4	2.21 E-2	11
E	6.1 E-6	1.87 E-3	1.87 E-3	7	3.26 E-3	13.5
F	1.13 E-4	3.13 E-2	3.13 E-2	5	1.4 E-2	12
G	1.21 E-4	7.83 E-5	1.21 E-4	8.5	8.7 E-2	8
H	4.97 E-4	9.81 E-2	9.81 E-2	3	1.29 E-1	7
I	3.45 E-4	1.58 E-1	1.58 E-1	2	2.04 E-1	4
J	3.28 E-6	8.24 E-9	3.28 E-6	14	3.26 E-3	13.5
K	5.16 E-6	1.68 E-11	5.16 E-6	12	1.29 E-3	15
L	3.83 E-5	1.2 E-10	3.83 E-5	11	8.11 E-2	9
M	1.21 E-4	3.93 E-10	1.21 E-4	8.5	3.46 E-2	10
N	2.85 E-6	1.53 E-10	2.85 E-6	15	2.36 E+0	2
O	1.14 E-4	8.61 E-3	8.61 E-3	6	1.35 E-1	6

There are uncertainties inherent in the DSS risk numbers due to the uncertainty of some of the associated site data entered into the model as well as assumptions made in exposure scenarios. Some of these uncertainties will be discussed in more detail later in this section. However, an attempt was made at arriving at conservative exposure scenarios and assumptions on site contamination and conditions to provide risk estimates at each site that would more likely err on the side of caution.

A summary of the site rankings produced by each method under both the current risk and future risk scenarios is provided in table 14. Each scenario was modeled for BTEX contamination and also for the combination of BTEX plus the PAHs and gas additives used in this study (denoted as BTEX$^+$).

Table 14 SUMMARY OF SITE RANKINGS PRODUCED BY EACH MODEL

SITES

Ranking Models	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**
DPM	15	10	9	2	3	1	5	4	6	13	11	8	12	14	7
RREM	14.5	9	12	1.5	5	1.5	9	9	14.5	12	12	4	3	6.5	6.5
DSS CR-BTEX	14	15	6	9	10	4	1	2	3	13	11	8	5	12	7
DSS CR-BTEX+	13	1	10	4	7	5	8.5	3	2	14	12	11	8.5	15	6
DSS FR-BTEX	11	15	3	12	13	5	1	2	6	14	9	10	8	7	4
DSS FR-BTEX+	3	5	1	11	13.5	12	8	7	4	13.5	15	9	10	2	6

Notes: 1. CR and FR refer to current and future risk scenarios, respectively.
2. BTEX+ refers to BTEX and remaining study contaminants (PAHs and gas additives)

coefficient of concordance were calculated along with associated P-values for inference testing. The results of this analysis are provided in table 15.

Table 15
VALUES FOR KENDALL COEFFICIENT OF CONCORDANCE (W) AND HYPOTHESIS TESTING RESULTS

	Current Risk Scenario			**Future Risk Scenario**		
	W	**P-Valve**	**Hypothesis Test Results (1)**	**W**	**P-Valve**	**Hypothesis Test Results (1)**
BTEX	0.7377	0.006	Reject H_o	0.2546	0.70995	Do not reject H_o
BTEX+	0.7664	0.00377	Reject H_o	0.3375	0.43676	Do not reject H_o

Notes: 1. The null hypothesis, H_o, claims that no association exists between the ranking methods or they are independent. The alternative hypothesis, H_A, says that an association exists.

As presented in Table 15, under the current risk scenario, the null hypothesis that no association exists between the models is rejected under both contamination scenarios. However, for the future risk scenario the null hypothesis of no association between the models cannot be rejected. It was anticipated that this might be the case under the future risk scenario as neither the DPM or the RREM are used to consider the hypothetical case of risk to some future (currently non-existent) receptor. The future risk scenarios, therefore, will not be addressed statistically any further.

A shortcoming of the analysis above is that, using the Kendall coefficient, the alternative hypothesis does not distinguish any direction of relationship (direct or inverse). Furthermore, with more than two sets of rankings, a completely inverse relationship cannot be defined [6:304].

As the above analysis (of Table 15) indicates, there appears to be a relationship between the rankings established by the three methods (under current risk scenarios) for this sample of fifteen JP-4 contaminated sites. Of interest is the direction of this relationship, which can be indicated only through pairwise comparisons using of another measure of association, the Spearman coefficient of rank correlation.

From Table 14 and using the procedures outlined in Section 3.7.2, four pairwise measurements of association (in the Spearman rank correlation values) were calculated along with associated P-values for inference testing. The results of this analysis are provided in Table 16.

Table 16
PAIRWISE COMPARISON OF MODELS (CURRENT RISK ONLY)
USING SPEARMAN COEFFICIENT OF RANK CORRELATION (R)

Model Pair	R	P-Value (rt tailed)	Hypothesis Test Results (1)	Relationship Level Indicated
DSS BTEX/DPM	+ 0.600	0.012	Reject H_o	Moderate to strong
DSS BTEX+/DPM	+ 0.696	0.004	Reject H_o	Moderate to strong
DSS BTEX/RREM	+ 0.175	0.256	Do Not Reject H_o	None to weak
DSS BTEX+/RREM	+ 0.193	0.235	Do Not Reject H_o	None to weak
DPM/RREM	+0.488	0.034	Reject H_o	Weak to Moderate

Note: 1. The null hypothesis, H_o, claims that no association exists between the ranking methods. The alternate hypothesis, H_{A+} in this case, is that a direct association exists between the methods.

The results in Table 16 above indicate that, for the group of 15 sites in this study, the null hypothesis of no association is rejected in favor of the alternate when the quantitative DSS model's rankings are compared with those of the semi-quantitative DPM. A moderate to strong direct correlation exists between the DSS and the DPM rankings for this sample of petroleum contaminated sites (for both contamination scenarios).

However, the null hypothesis that no association exists between the quantitative DSS model's rankings and those of the primarily qualitative RREM model cannot be rejected. At best, only a weak correlation exists between the RREM and the validation model, and the associated p-values indicate that one cannot reject the null hypothesis that no correlation exists for this sample of 15 sites.

The results of the rank comparisons of the DPM (using all chemicals of concern) versus the DSS and those of the RREM (BTEX only) versus the DSS are shown graphically in Figures 13 and 14, respectively. As indicated earlier in Table 16, at a minimum, a moderate correlation is shown between the DPM and DSS site rankings (Figure 13). However, Figure 14 illustrates graphically that, at best, a weak correlation exists between the RREM and the DSS validation model.

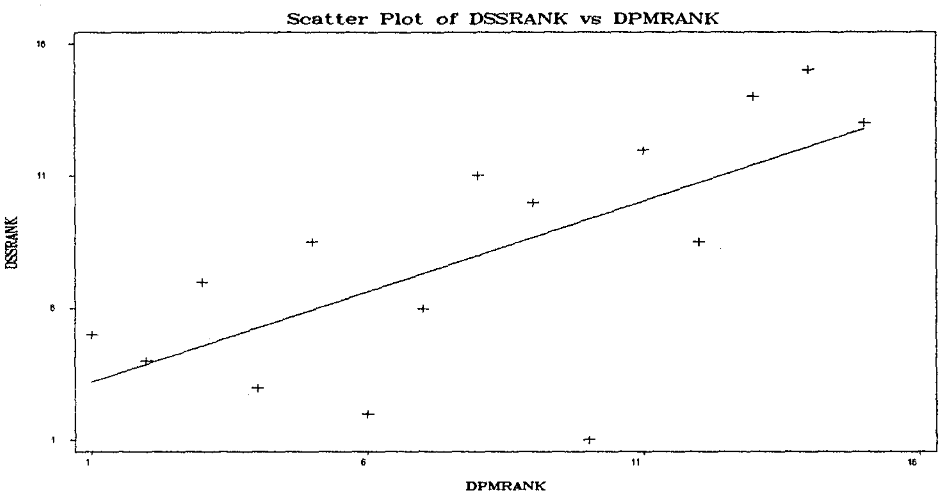


Figure 13: Scatter Plot of DPM Versus DSS Ranks (BTEX+)

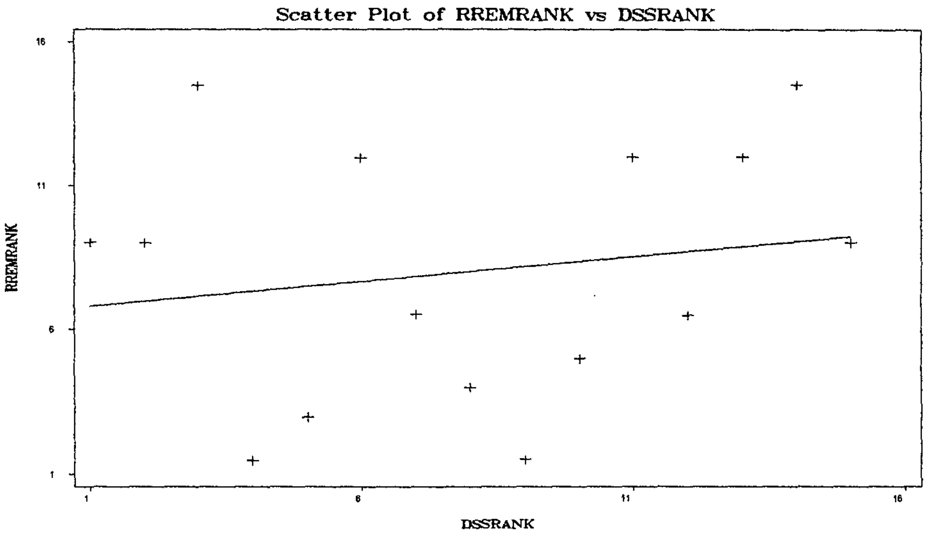


Figure 14: Scatter Plot of RREM versus DSS (BTEX)

4.2 Non-cancer Risks

Though they obviously made no contribution to the carcinogenic risk used to rank the sites, chemicals that are noncarcinogenic (e.g., ethylbenzene, toluene, xylenes, etc.) were included in this study to more closely match the DOD models.

4.3 Uncertainties in Results

Though this discussion is not all inclusive, some of the more notable sources of uncertainty in the DSS results include the following model entries: contaminant concentration and mass estimates, the size of each site, the fraction of organic carbon content in the soil at each site, the hydraulic conductivity at each site, the dispersivity values used for each site, and the receptor exposure scenarios. These uncertainties were associated, to some degree, with each site in the study. Each is discussed briefly below.

exposure concentrations and duration and, therefore, under or overestimate associated site risks. Perhaps the greatest potential for error occurred with those sites with little contaminant data where the masses of the fuel components currently at the sites were based on those in the original spill. This might tend to overestimate site risks.

For most sites, a minimal value (0.1%) for the fraction of organic carbon content was used to eliminate the modeling of any possible retardation effects of the soil on the fuel organics. This is unfortunate in that the result was likely higher contaminant mobility than might actually occur at some of the sites, affecting both the exposure durations and the concentration levels experienced at receptor points.

Some of the sites reported wide ranges of soil hydraulic conductivity (K) values. The DSS model's simplifying assumption provided for a constant value for K, but where average values used were derived from measurements several orders of magnitude apart, the actual ability of the soil to transmit water (and aqueous phase contamination) is uncertain.

Dispersivity values used could potentially contribute to under or overestimation of risk posed by each site. These parameters effect the shape of the groundwater plume and the mass of the contaminants that move past a downgradient receptor well.

Finally, the exposure scenarios chosen, as well as the off-site receptor points established for some of the sites, could contribute to either an over or an underestimation of actual site risks. Conservative exposure assumptions were chosen, however, to provide for cautious measures of risk at each site.

5.0 RESEARCH CONCLUSIONS

The research problem was that the validity of the present DOD model for ranking contaminated sites for cleanup, the RREM, and its predecessor, the DPM, had been questioned by some risk assessment professionals. In addition, no validation efforts had occurred to reduce doubts about the usefulness of these models.

The objective of this thesis effort, therefore, was to serve as an initial validation effort for the RREM and the DPM models by comparing their rankings against those established by a more rigorous quantitative risk assessment approach.

The scope of this study was narrowed to a small group of 15 petroleum contaminated sites. This was because the quantitative model used in this study, the DSS, was developed to address this class of contaminated sites. Each site was contaminated through either spills or leaks of JP-4 (Jet Fuel, No. 4), a gasoline-kerosene distillate fuel, into the soil. At most of the sites, fuel contamination is now in both the unsaturated (vadose) and saturated (groundwater) subsurface zones.

Approximately 110 sites were considered initially; however, most of the sites were quickly eliminated from further study simply because they had not been evaluated under the RREM or too little site data had been collected to perform the quantitative modeling required in this study.

Admittedly, this research involved a fairly small sample of 15 petroleum contaminated IRP sites. However, the sites were chosen randomly from among those with strictly petroleum contamination, with the only selection criteria being that each site readily available to provide for quantitative modeling of each site. Naturally, both imperfect site characterization data and the assumptions associated with this study add a factor of uncertainty in the results.

For this sample of 15 petroleum contaminated sites, a moderate to strong correlation between the DPM and the quantitative (DSS) validation model suggests that the DPM captures some key elements of a quantitative approach. However, no to weak correlation between the RREM and the validation model suggests the RREM rankings are questionable. No agreement between the models under the future risk scenario suggests neither DOD model is addressing this hypothetical, worst-case scenario of an on-site resident child receptor. This can be explained by the fact that the DPM and the RREM are used to consider only risk to current receptor populations.

The said benefit of the RREM over the DPM is that it is easily understood by stakeholders (e.g., local public officials) in the restoration process. The drawback, however, is that the quality of site ranking assessments may suffer causing improper resource allocation. A better approach could be to use the DPM as a site ranking tool but to package this model in a way that would make it more understandable to stakeholders.

6.0 FUTURE RESEARCH POSSIBILITIES

Some possibilities for continued research of the DOD's site prioritization tools are discussed below.

6.1 Continue Validation Effort

Additional validation efforts may allow broader inferences to be drawn on the relationships between the ranking models than could be hypothesized with the small sample from the single class of contaminated sites that was used in this study. Such a study might potentially include several hundred sites if site information could be gained DOD wide directly from site managers. The researcher could then simply enter the information and relevant exposure scenarios into the DSS or a comparable quantitative model. Such an effort could provide support and credibility for the conclusions of this thesis, or, on the other hand, might dispute them.

6.2 Sensitivity of Site Risk to Input Parameter Variation

Application of the DSS's Monte Carlo Simulation options would allow a researcher to study how variations in a site parameter, such as the average soil organic carbon content, affect the resulting risk level at a site. Such information could indicate which site parameters should be measured with greatest precision (e.g., those with the greatest affect on risk level under a given exposure scenario) during site characterization. This could also be used to suggest improvements in the DOD models.

6.3 Restoration Funding Trends Since DOD's Switch to RREM

Of the three factors used to arrive at the RREM category for a site, two (the MPF and RF) are strictly subjective evaluations of the model user. A general feeling from informal conversations with remedial project managers (RPMs) during this research was that simply having enough concentration levels of a particular contaminant at a site could practically guarantee at least medium risk level evaluation for the site, implying that model users could make unrealistically conservative assumptions in the subjective portions of the RREM to inflate evaluations. An incentive for this is that DOD policy is to fund all medium and high risk sites for restoration to at least a level that would place them in the low risk category. With pressures that come to bear from local and state regulators at installations and the competition for funding so intense, a overly conservative approach could be taken at installations in an effort to secure cleanup funds and appease regulators and the local community.

DPM site scores can range from 0 to 100. Among the fifteen sites in this study, eight sites had DPM scores under 4.5, and five of these eight had scores of 2.0 or less. If DPM is assumed to have some validity, such low scores would most likely predict low risk sites under the RREM. Of these eight sites, two were evaluated as high risk, and five were placed in the medium risk category under the RREM. Though this certainly is not proof of inflationary scoring, there seems to be an overall upward trend in the estimation of the severity of site risks in this study when the RREM is used as the ranking tool instead of its predecessor, the DPM.

One possibility for future research is to compare DPM scores against RREM evaluations to assess any changes/trends in restoration funding since the DOD stopped using the DPM in favor of the RREM. This might indicate whether the overall level of risk of DOD sites is shifting under the RREM from what it was under the DPM evaluations, serving as a check of possible inflationary scoring under the RREM.

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APPENDIX: Site Descriptions and Data Sources

The following is a list of primary sources for site data for each of the study sites. Secondary sources included ADPM data files for each site, the IRPIMS database, Risk Assistant (software) and the International Station Meteorological Climate Summary (Version 3.0) for associated weather data, and IRP remedial project managers (RPMs) at some bases.

Moody AFB Sites

The following report served as the primary source of information for the four Moody sites studied:

> RCRA Facility Investigation Report
> Moody AFB, Georgia, November 1993.

At *Moody site ST-15*, Underground Waste Fuel Storage Area, an estimated 1,500 gallons of waste JP-4 was spilled. The area had one 5,000 gallon UST that was active from 1976 until 1984 when it was removed.

At *Moody site SS-20*, Wingtip Tank Storage Area, an estimated 80 gallons was spilled. The UST involved was removed in March 1990 and replace with clean fill.

At *Moody site ST-18*, Bulk Fuel Storage Area, the spill/leak quantity is unknown. The area involved four ASTs with JP-4 combined capacity of 1.3 million gallons. One AST had apparently leaked for several years. Several tons of contaminated soil removed from area.

At *Moody site ST-12*, North POL Site, an estimated 1,000 gallons of fuel lost from overfilling. Additional impacts suspected from former UST waste sludge pit that was removed.

Wright-Patterson AFB Sites

The following report served as the primary source of information for the three Wright-Patterson sites studied:

> IRP Draft Remedial Investigation Report for OU2 at
> Wright-Patterson AFB, Ohio, 14 November 1994.

At *Wright-Patterson site SS-28*, Spill Site 3, an estimated 2,500 gallons fuel was spilled in March 1981.

At *Wright-Patterson site SS-63*, Spill Site 10, an estimated 150 gallons was lost from a hydrant leak.

At *Wright-Patterson site SS-27*, Spill Site 2, an estimated 8,300 gallon spill occurred in 1976.

Ellsworth AFB Sites

The following report served as the primary source of information for the three Ellsworth sites studied:

> Final Corrective Action Plan for the Flightline Refueling Area,
> (Site 1, SS-08, ST-10, and ST-14), Ellsworth AFB, South Dakota,
> December 1994.

The three Ellsworth sites listed below are in fairly close proximity to one another in the flightline refueling area. It was noted that drinking water is not obtained downgradient from this base, with local residencies obtaining their water from 800 foot deep wells.

At *Ellsworth site SS-8*, (SP-1) Pumphouse No. 7 (ST-8), an estimated 50,000 gallons fuel was lost.

At *Ellsworth site ST-10*, (SP-3) Hydrant Fuel Leaks (OU-10), an estimated 10,000 gallons was lost from a hydrant leak.

At *Ellsworth site ST-14*, Pumphouse #6 / Jet Fuel Spill (ST-14), an estimated 12,000 gallon have been lost.

Nellis AFB Sites

The following reports served as the primary sources of information for the two Nellis sites studied:

> Final IRP Site 27 Remedial Investigation
> Nellis AFB, Nevada, November 1994.
>
> IRP Preliminary Assessment /Feasibility Study
> Report at 15 Sites: LF-5, LF-5A, DP-6, LF-7,
> DP-8, DP-9, DP-14, SS-30, RW-31, OT-8,
> OT-9, LF-10 OT-11, OT-12, SD-13. Volume 1,
> January 1994.

At *Nellis site ST-27*, Site 27, an estimated 3000 gallons of JP-4 was lost. Approximately 2900 cubic yards of contaminated soil were removed in 1989 along with four USTs. The volume of contaminated soil remaining is estimated at 314,000 cubic yards (most below 45 feet below ground surface).

At *Nellis site SS-30*, Spill, an estimated 600 gallons were spilled in this area. The soil contamination begins around 25 feet below the surface.

Minot AFB Site

The following report served as the primary source of information for the Minot site studied:

> Draft Final IRP RI/FS, Stage 3, Technical Report
> For the POL Storage Area, Volume 1, June 1994.

At *Minot site ST-11*, Eight USTs at Nine Locations, an estimated 15,000 gallon JP-4 spill occurred (1982) with most of the fuel dispersed by the fire department and left to evaporate or soak into the ground.

Little Rock AFB Site

The following report served as the primary source of information for the Little Rock site studied:

> Site Investigation Report, UST Sites
> Preliminary Draft Report, Volume1.
> Little Rock AFB, Jacksonville, Arkansas,
> 20 May 92.

At *Little Rock site ST-24*, AAFES Service Station, an estimated 10,000 gallons of fuel was lost. Four 10,000 gallon USTs were removed in 1991 along with 2,841 cubic yards of soil. Significant groundwater contamination is said to remain from the UST losses.

Pope AFB Site

The following report served as the primary source of information for the Pope site studied:

> Draft Phase II Remedial Investigation Report
> for Pope AFB, North Carolina. June 92

At *Pope site ST-02*, POL Bulk Storage Area, an estimated 900 gallons of JP-4 was lost in a surface spill in 1983. The groundwater has not yet been impacted by this spill.

Vita

David M. Hunter (Mike) [redacted]. He graduated from Lee High School located in Huntsville in 1980. Following graduation, he worked at various "odd jobs" and attended a local university part-time until marrying Jama L. Blair in July 1984 and enlisting in the United States Air Force in September of that same year. He received training as an Electronic Computer and Switching System Technician at Keesler and Sheppard Air Force Bases and served in that capacity at Sembach Air Base, Germany, from July 1985 to July 1988. Then Staff Sergeant Hunter entered the Airman Education and Commissioning Program, attending Arizona State University, Tempe, Arizona and receiving a Bachelor of Science in Industrial and Management Systems Engineering in May 1991. Upon completion of Officer Training School in September 1991, 2Lt Hunter was assigned to the 325th Civil Engineering Squadron at Tyndall AFB, Florida, where he served as a contract programmer and as an environmental engineer before being selected to attend the Class of 1995 GEEM Program at the Air Force Institute of Technology's School of Engineering. He and his wife Jama are the proud parents of three children, a daughter, Erin [redacted] and two sons, Paul ([redacted]) and Blair ([redacted]

[redacted]

REPORT DOCUMENTATION PAGE

Form Approved
OMB No. 0704-0188

1. AGENCY USE ONLY *(Leave blank)*

2. REPORT DATE
December 1995

3. REPORT TYPE AND DATES COVERED
Master's Thesis

4. TITLE AND SUBTITLE
A COMPARISON OF REMEDIATION PRIORITIES DEVELOPED BY THE DEFENSE PRIORITY MODEL, THE RELATIVE RISK EVALUATION METHOD, AND A QUANTITATIVE RISK ASSESSMENT APPROACH

5. FUNDING NUMBERS

6. AUTHOR(S)
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7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES)
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2750 P Street
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8. PERFORMING ORGANIZATION REPORT NUMBER
AFIT/GEE/ENV/95D-08

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)

10. SPONSORING/MONITORING AGENCY REPORT NUMBER

11. SUPPLEMENTARY NOTES

12a. DISTRIBUTION/AVAILABILITY STATEMENT
Approved for public release; distribution unlimited

12b. DISTRIBUTION CODE

13. ABSTRACT *(Maximum 200 words)*

The DOD site ranking models addressed in this study are the Relative Risk Evaluation Method (RREM), and its predecessor, the Defense Priority Model (DPM). The RREM has been criticized as relying too much upon qualitative factors that could, perhaps, create inconsistencies between users of the model. One criticism of the DPM, not used after 1993, is that some of it's models appear theoretically weak. The problem is that these two DOD approaches for prioritizing sites for remediation have not been validated. The research objective was as an initial validation effort for the RREM and the DPM models by comparing their rankings of a sample of contaminated sites against those of a rigorous, quantitative risk assessment model. Site specific data obtained from study reports for 15 petroleum contaminated sites was entered into a quantitative model. For this sample of sites, the moderate to strong correlation between DPM and the validation model suggested that DPM captures some key elements of a quantitative approach. However, no to weak correlation between RREM and the validation model suggested RREM rankings are questionable for this sample.

14. SUBJECT TERMS
Contaminated Site Ranking, Defense Priority Model, Relative Risk, Risk Assessment

15. NUMBER OF PAGES
123

16. PRICE CODE

17. SECURITY CLASSIFICATION OF REPORT
Unclassified

18. SECURITY CLASSIFICATION OF THIS PAGE
Unclassified

19. SECURITY CLASSIFICATION OF ABSTRACT
Unclassified

20. LIMITATION OF ABSTRACT
UL