Development of Risk-Based Listing Concentrations for Hazardous Constituents Contained in Spent Filter Aids and Triarylmethane (TAM) Wastewater Treatment Sludges

Background Document

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This document does not contain confidential business information

Table of Contents

Table of Contents (continued)

Table of Contents (continued)

Figures

Tables

List of Acronyms

1.0 Executive Summary

1.1 Purpose

The purpose of this document is to present EPA's assessment of potential risk from spent filter aids and triarylmethane (TAM) wastewater treatment sludges. EPA's risk assessment of spent filter aids and TAM sludges is designed to produce risk-based concentration limits for potentially hazardous constituents contained in these two waste streams. The risk-based concentrations resulting from the risk assessment described herein are constituent-specific numerical limits for hazardous constituents in TAM sludge and spent filter aids that are considered to be protective of human health. They can be defined as the maximum constituent concentrations at which adverse health effects from any single constituent do not exceed levels of concern.

1.2 Approach

This risk assessment uses both deterministic and probabilistic methodologies to develop risk-based concentrations for constituents of concern contained in two dye and pigment waste streams: (1) spent filter aids, diatomaceous earth, or adsorbents used in the production of azo, anthraquinone, or triarylmethane dyes, pigments, or FD&C colorants, hereinafter refered to as spent filter aids; and (2) wastewater treatment sludge from the production of triarylmethane dyes and pigments, hereinafter referred to as TAM sludges.

The only waste management practice considered in this risk assessment is disposal in municipal landfills. The filter aid waste quantities used for this assessment are from a distribution of waste volumes reported to be generated by the dye and pigment industry. For TAM sludge, only a single TAM sludge waste quantity is used.

The primary receptors considered in this analysis are adult and child receptors exposed via ingestion (i.e., drinking water) and noningestion (e.g., showering) pathways to water from groundwater wells contaminated by the leachate from the municipal landfill receiving filter aid or triarylmethane sludge waste streams. Specific constituents assessed in this analysis for both filter aid and triarylmethane waste streams are primarily those that have been detected or measured in samples obtained during EPA or industry sampling and analysis of any of the dye and pigment waste steams, and for which EPA-established health-based numbers are available.

Primary fate and transport models used for this analysis are a landfill partitioning model, which is used to estimate the concentration of leachate from the landfill and the emission rate for volatile constituents from the landfill, and the groundwater model, EPA's Composite Model for

Leachate Migration with Transformation Products, which is used to estimate the concentration of constituents of concern at the residential drinking water well.

The exposure assumptions used in this analysis are values from EPA's *Exposure Factors Handbook* (U.S. EPA, 1997c). The health benchmark data used are based upon the values presented in the Integrated Risk Information System (IRIS) online database of verified health benchmarks or in the Health Effects Assessment Summary Tables (HEAST).

1.3   Risk Characterization

The results of this analysis are presented in two forms: (1) as risk-based concentrations of hazardous constituents dye and pigment waste streams, and (2) as risk-based concentrations of hazardous constituents in the leachate generated from landfilled dye and pigment waste streams. These concentrations are calculated based on the groundwater pathway that is the most protective pathway for all constituents when disposed of in municipal landfills. The risk-based concentrations are calculated by assuming the residential drinking water well concentration is equal to EPA-established protective or health-based levels for each constituent for the most sensitive receptor (adult or child). Protective concentrations are those at which adverse health effects from any single constituent present in contaminated drinking water and/or water used for bathing or showering do not exceed a 1×10^{-5} individual lifetime cancer risk or a noncancer hazard quotient (HQ) of 1 at the upper end of the risk distribution.

1.4   Uncertainties

Uncertainty in quantifying potential risks to human health and the environment is inherent in the risk assessment process. It occurs because the risk assessment process is complex and variability is inherent in the environment. EPA typically classifies the major areas of uncertainty in risk assessment as parameter uncertainty, exposure scenario uncertainty, and model uncertainty.

Primary sources of parameter uncertainty specific to this assessment include waste characterization and waste volume data, waste management unit size and location parameters, and toxicologic benchmarks. To the extent possible, we address variability in physical/chemical properties of the wastes assessed by using all available waste characterization data for the analysis. Waste management unit parameter variability is addressed by using a national distribution of municipal landfill areas. Site characteristic variability (e.g., hydrogeological, meteorological, and soils data) is addressed through the use of distributions of location-related parameters for the sites modeled. Health benchmark data used are based on values that have been established and verified by EPA. These data are likely to be conservative because of the uncertainties and challenges associated with condensing toxicity data into a single quantitative expression.

Exposure scenario uncertainty derives from the use of default assumptions regarding population activity patterns, mobility, dietary habits, body weights, and other factors. Exposure factors used for this assessment are obtained from EPA's *Exposure Factor Handbook* (U.S. EPA, 1997c), which provides the current state-of-the-science regarding exposure modeling and assumptions.

Model uncertainty is associated with all models used in all phases of a risk assessment. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in the models due either to increased complexity or to a lack of data. The models used in this risk assessment are selected based on science, policy, and professional judgment, and all models used in the assessment have undergone some level of peer review. Even though the models used in this report have been widely used and accepted, they remain a significant source of uncertainty that could result in underestimating or overestimating risk.

2.0 Introduction

2.1 Background

The 1984 Hazardous Solid Waste Amendments (HSWA) to the Resource Conservation and Recovery Act (RCRA) require the U.S. Environmental Protection Agency (EPA) to make listing determinations for wastes from the production of dyes and pigments (see RCRA section 3001(e)(2)). On December 22, 1994, EPA proposed to list as hazardous five wastes generated during the production of dyes and pigments because certain ways of disposing of these wastes may present a risk to human health and the environment. These five wastes are wastewaters and wastewater treatment sludges from the production of azo dyes, wastewaters and wastewater treatment sludges from the production of azo pigments, and still bottoms or heavy ends from the production of triarylmethane (TAM) dyes or pigments (59 FR 66072-66114).

In the 1994 rule, EPA deferred action on three dye and pigment waste streams based on insufficient characterization data or lack of health-based levels (HBLs) for specific constituents of concern. The "deferred" dye and pigment waste streams are now being addressed through a separate proposed rulemaking. In support of this rulemaking, EPA has conducted a formal risk assessment for two of the deferred waste streams:

> \# Spent filter aids, diatomaceous earth, or adsorbents used in the production of azo, anthraquinone, or TAM dyes, pigments, or FD&C colorants

> \# Wastewater treatment sludge from the production of TAM dyes and pigments (excluding triarylmethane pigments using aniline as a feedstock).

2.2 Purpose

The purpose of this document is to describe EPA's assessment of risk from, and the development of risk-based listing concentrations for, hazardous constituents contained in spent filter aids and TAM wastewater treatment sludges. EPA's risk assessment of spent filter aids and TAM sludges is designed to produce risk-based concentration limits for potentially hazardous constituents contained in these two waste streams. The risk-based concentrations resulting from the risk assessment described herein are constituent-specific numerical limits for hazardous constituents in TAM sludge and spent filter aids that are considered to be protective of human health. They can be defined as the maximum constituent concentrations at which adverse health effects from any single constituent do not exceed a 1×10^{-5} (1 in 100,000) individual lifetime cancer risk or a noncancer hazard quotient (HQ) of 1 at the upper end of the risk distribution for any potential human exposure route.

There are several reasons for using a risk-based concentration approach for estimating risk from deferred dyes and pigments waste streams. First, these wastes are generated by an industry that uses batch processes to manufacture a variety of products in response to market demand for a wide variety of dye and pigment products. Batch operations may result in highly variable wastes at the same facility or different facilities. A concentration-based approach allows the variable wastes generated at these facilities to be evaluated individually for hazard, so only the truly hazardous wastes are designated as such.

Second, many manufacturers in the dye and pigment industries want to keep facility-specific product and waste information confidential. These manufacturers are concerned that release of such information could cause competitive harm. Development of risk-based constituent concentrations enables EPA to maintain most facility-specific information confidential, since information such as product formulations or concentrations of constituents in the wastes is not used directly in the risk modeling to generate the risk-based concentrations.

Third, a concentration-based listing approach may provide an incentive for hazardous waste generating facilities to modify their manufacturing processes or treat their wastes. For example, if a facility has a listed hazardous waste based on constituent-specific concentration levels established by EPA, the facility may decide to modify its manufacturing process or treat its waste in order to generate a nonhazardous waste.

2.3   Document Organization

The remainder of this document is organized into the following sections:

Section 3, Analytical Framework, presents an overview of the risk analysis.

Section 4, Risk Assessment Scenario, presents a detailed description and discussion of input parameters and assumptions used in this risk assessment.

Section 5, Estimating Exposure Point Concentrations, presents a detailed discussion of the risk assessment methodology and describes the fate and transport models and how they are used in the risk assessment.

Section 6, Exposure and Toxicity Assessments, presents the exposure factors and a discussion of health benchmarks used in the analysis.

Section 7, Risk Characterization, summarizes the results of this risk analysis.

Section 8, Uncertainty, discusses the variability and uncertainty associated with this risk assessment.

Section 9, References, lists all source citations in the document.

Appendixes A through I provide supporting documentation.

3.0 Risk Assessment Framework

This section presents an overview of the process used to estimate risk associated with management and disposal of filter aid and triarylmethane sludges generated during the manufacture of organic dye and pigment products. Details of the analysis are presented in Sections 4.0, Risk Assessment Scenario; 5.0, Estimating Exposure Point Concentrations; 6.0, Exposure and Toxicity Assessments; and 7.0, Characterizing Risk.

3.1 Establishing Risk Assessment Scenarios

The risk assessment scenario considered herein is disposal in unlined municipal landfills of filter aid and TAM sludge wastes generated during the manufacture of organic dye and pigment products. Each of these waste streams is considered separately in the risk assessment.

3.1.1 Characterization of Spent Filter Aid and TAM Sludge Waste Streams

Spent filter aids are diatomaceous earth or other similar adsorbents used in the production of azo, anthraquinone, and triarylmethane dyes, pigments, or FD&C colorants. This is a waste stream that may be generated by a facility manufacturing any dye or pigment product. Triarylmethane wastewater treatment sludges are generated from the production of TAM dyes and pigments. A noncommingled TAM sludge waste was reported as being disposed of in a municipal landfill, and this waste stream is included in the risk assessment.

Specific constituents assessed in this analysis for both filter aid and TAM waste streams are (1) those that have been detected or measured in samples obtained during EPA sampling and analysis of the dye and pigment waste steams, and (2) those for which EPA-established health-based numbers are available. The procedure used to identify the list of constituents analyzed in this assessment is described in the listing background document for this rulemaking (EPA, 1999). Certain constituents included in the risk assessment conducted for the 1994 proposed rulemaking on azo/benzidine dye and pigment waste streams are excluded from consideration in the risk assessment for the current rulemaking. These constituents are acetoacetaninide (AAA), acetoacet-o-toluidine (AAOT), and acetoacet-o-anisidine (AAOA). For the 1994 proposal, health-based numbers were derived for AAA, AAOT, and AAOA based on a structural activity relationship (SAR) analysis. The SAR analysis has since been reevaluated and revised based on comments received in response to the 1994 proposal. The revised analysis, which has been independently peer-reviewed, concludes that currently data are insufficient to make a quantitative estimation of the carcinogenic potential of AAA, AAOT, or AAOA or to establish provisional noncancer benchmarks. The revised toxicological analysis for these compounds and the peer-review documents are provided in Appendix A.

The only waste management practice considered in this risk assessment is disposal in municipal landfills. It is assumed that disposal in any municipal landfill described in EPA's distribution of municipal landfills is possible, with some geographic limitations reflecting the locations of the dye and pigment manufacturers. Therefore, the entire distribution of data available for each of the parameters needed to model potential risk associated with disposal of dye and pigment waste streams in municipal landfills is used. The primary source of data for key modeling inputs (e.g., surface area, active life, distance to well) is EPA's 1988 National Survey of Solid Waste Municipal Landfill Facilities (U.S. EPA, 1988).

The filter aid waste quantities managed in municipal landfills used for this assessment are from waste volumes reported to be generated by the dye and pigment industry. These volumes have been claimed as Confidential Business Information (CBI). Only the distribution of filter aid volumes based on the reported volumes is used for this analysis. One TAM sludge waste quantity is reported in the industry survey as being managed in a municipal landfill. Data on this waste stream are not claimed as CBI. The volume of this single waste stream, 57 metric tons, is used for the risk assessment.

Geographic locations where spent filter aids and TAM sludges are produced are reported in the dye and pigment industry survey and these data are used to generate location parameters needed to conduct the risk assessment. Geographic location is used to identify the soil, climate, and hydrogeologic parameters used in the fate and transport modeling. Location-related parameters required for the risk assessment and specific inputs and data distributions used to model risk from these units are described in detail in Sections 4.0 and 5.0 of this document, respectively.

3.1.2 Description of Receptors and Exposure Pathways

The primary receptors considered in this analysis are adults and children exposed via ingestion (i.e., drinking water) and noningestion (e.g., showering) pathways to water from groundwater wells contaminated by the leachate from the municipal landfill receiving filter aid or TAM sludge waste streams. Only receptors with residential drinking water wells are considered for the groundwater pathway. All community wells and other municipal water supplies are assumed to be treated and, therefore, not contaminated with the waste steams of concern. Residential wells are assumed to be located downgradient from the landfill and within the top 10 meters of the aquifer. Assumptions concerning distance of receptor wells from waste management units (WMUs) are based on data obtained from EPA's 1988 National Survey of Municipal Landfills (U.S. EPA, 1988). Detailed descriptions of these parameters are presented in Section 4.0.

Receptors from nongroundwater pathways are evaluated as part of a sensitivity analysis. As previously noted, the nongroundwater pathways are not further evaluated as part of the deterministic or probabilistic analysis conducted for this assessment because the sensitivity analysis showed potential risks to be below levels of concern. Receptors for nongroundwater pathways are assumed to be farmers and their children who live in proximity to the municipal landfill. Nongroundwater pathways for filter aid and TAM sludge wastes disposed of in municipal landfills result from the emission of vapors from these landfilled wastes. Exposure from both

direct (e.g., inhalation) and indirect (e.g., dietary) nongroundwater pathways is considered. The vapors from the landfill are dispersed and deposited through the air pathway directly to the receptor via the inhalation pathway and to plants and animals belonging to a farmer residing near the landfill. The air concentration of vapors is used directly to estimate the risk due to inhalation by the farmer and child. It is also used in the indirect exposure models to estimate deposition to soil and air-to-plant transfer of contaminant to fruits, vegetables, grain, and forage. Fruits, above and belowground vegetables, grain, forage, and silage are also contaminated by soil-to-plant transfer. Contaminants are then transferred to beef and dairy cattle through ingestion of grain, forage, and silage. Ultimately, the human receptors' food chain exposures result from ingestion of aboveground and belowground vegetables, fruits, beef, dairy products, and inadvertent ingestion of soil. All of these food chain pathways are considered in the sensitivity analysis for the nongroundwater risk assessment, which is described in detail in Appendix B.

3.2   Estimating Exposure Point Concentrations

3.2.1   Risk Assessment Methods

The risk assessment is conducted in three stages: (1) the sensitivity analysis, (2) the deterministic analysis, and (3) the probabilistic analysis. Sensitivity, deterministic, and probabilistic analyses are conducted for the groundwater pathways. For the nongroundwater pathways, EPA was able to use results from the sensitivity analysis to screen out nongroundwater risks relative to potential groundwater risks associated with disposal of wastes in landfills. Therefore, deterministic and probabilistic analyses are not conducted for nongroundwater pathways.

3.2.1.1 <u>Sensitivity Analysis</u>. The purpose of the sensitivity analysis is to identify the most sensitive or risk-driving parameters in the risk assessment model and to determine high-end and central tendency values for subsequent use in the deterministic analysis. The sensitivity of individual parameters is defined as the ratio of the predicted health risk when the parameter is set at its high-end value compared to the risk estimated when the parameter is set at its central tendency (50th percentile) value. The high-end parameter corresponds to its 90th or 10th percentile value, depending on whether a high or low value of that parameter results in a higher predicted risk. The sensitivity analysis is conducted by varying each parameter or set of linked parameters to high-end parameters one at a time while holding all other variables in the analysis at central tendency and comparing the risk results using a single high-end parameter to the results obtained when all values are set at central tendency. Using this method, the two most sensitive high-end parameters are identified and these parameters are set to their high-end values in the deterministic analysis.

For the nongroundwater pathway, EPA was able to use the results from the sensitivity analysis as a screening level analysis of nongroundwater risks. EPA was able to screen out nongroundwater risks by comparing the results of the nongroundwater sensitivity analysis to the results of the groundwater sensitivity analysis, which was performed using the same inputs for common parameters. In all cases, the groundwater risk analysis produced higher risk estimates for all constituents. Because the purpose of this analysis is to set risk-based concentration limits, only the pathways of most concern or those that present the highest risk are required to determine protective concentrations. Because, based on the results of the sensitivity analyses, groundwater

pathways are found to be most limiting (i.e., to present the highest risk) in all cases for all constituents of concern, only the groundwater pathway required further evaluation using deterministic and Monte Carlo analyses. In other words, risk-based concentrations based on groundwater pathway risks will also be protective of nongroundwater pathway risks. Based on this finding, no further modeling of nongroundwater risks was conducted.

3.2.1.2 <u>**Deterministic Analysis**</u>. The deterministic method produces point estimates of risk or hazard based on single values for input parameters. For this analysis, high-end deterministic results are estimated using a double high-end risk assessment methodology. In this method, the two parameters identified to be most sensitive are set at their high-end values and all other parameters are set at central tendency. The point estimate in which all variables are set at central tendency is assumed to be the central tendency risk estimate. The high-end risk estimate is presumed by EPA to be a plausible estimate of individual risk for those persons at the upper end of the risk distribution. The intent of these descriptors is to convey estimates of exposure in the upper end of the distribution (i.e., above the 90th percentile) while avoiding estimates that are beyond the true distribution.

3.2.1.3 <u>**Probabilistic Analysis**</u>. In the probabilistic analysis, parameters are varied using a Monte Carlo methodology. Monte Carlo analysis provides a means of quantifying variability and uncertainty in risk assessments. Monte Carlo simulation is a statistical technique that calculates an individual risk value or hazard quotient (HQ) repeatedly using randomly selected inputs from each category of parameters that affect or determine risk. For each calculation, the Monte Carlo simulation uses parameter values that are randomly selected from the distribution of values available for each parameter. The range of values selected for the input parameters reflects the composition or probability density function of the distribution of values corresponding to each input parameter. The repetitive calculations take many randomly selected combinations of input parameters to generate a range of risk results. Based on the distribution of the output, a risk or hazard level representing the high end (e.g., 90th percentile) or central tendency (i.e., 50th percentile) can be determined. Although the simulation is internally complex, commercial software performs the calculations as a single operation, presenting a distribution of risk results. From these results, the percentile distribution of exposure point concentrations and risks can be determined for each risk assessment scenario. Potential groundwater pathway risks from disposal of filter aid and TAM wastes in municipal landfills are also assessed using the probabilistic risk assessment method.

3.2.2   Fate and Transport Modeling

The risk analysis employs several key fate and transport models. Models used to conduct the sensitivity analysis of both groundwater and nongroundwater pathways include the following:

> \#   EPA's Industrial Source Complex Short Term, version 3 (ISCST3), which is used to estimate the dispersion and deposition of vapors emitted from the municipal landfill

> \#   A landfill partitioning model based on the equations presented in a series of articles by Jury et al. (Jury et al., 1983, 1984, 1990), which is used to estimate the

concentration of leachate from the landfill and the emission rate for volatile constituents from the landfill

The indirect exposure model, which estimates the fate and transport of constituents through the environment and into the food chain to produce estimates of human health risk

The groundwater model EPACMTP or <u>EPA</u>'s <u>C</u>omposite <u>M</u>odel for Leachate Migration with <u>T</u>ransformation <u>P</u>roducts (U.S. EPA, 1996a, 1996b, 1997a, 1997b), which estimates the concentration of constituents of concern at the residential drinking water well.

Models used to conduct the deterministic and probabilistic modeling of groundwater pathways include: the landfill partitioning model, the EPACMTP groundwater model, and a shower model that is used to estimate dermal and inhalation exposure from use of contaminated tap water for showering and/or bathing.

       3.2.2.1   <u>**Air Model**</u>. The ISCST3 air dispersion and deposition model is used to estimate the vapor air concentrations and deposition rates needed to develop relative risk estimates associated with vapor air emissions from the municipal landfill. Air pathway risks are estimated using emissions of volatile constituents from the landfill partitioning model (described below) as inputs to the ISCST3 air model and using ISCST3 to estimate the air concentration and deposition of vapor for each constituent at receptor locations. As previously discussed, ISCST3 is used only for the sensitivity analysis to estimate risks from nongroundwater pathways.

       3.2.2.2   <u>**Indirect Exposure Model**</u>. The indirect exposure model is also used to conduct only the sensitivity analysis component of this assessment. This model estimates the fate and transport of constituents through the environment and into the food chain to produce estimates of human health risk to the farmer and farm child, the receptors assumed to be most highly exposed via the ingestion pathway. Risks to the farm family can occur through the ingestion of plants grown on affected soil and/or through ingestion of beef and dairy products from animals raised on contaminated feed. The indirect exposure model estimates exposure point concentrations in plant and animal tissue by using constituent-specific food chain biotransfer factors. For plants and beef and dairy products, empirically derived plant biouptake and food chain transfer factors are used to estimate the concentrations of hazardous constituents in plant and animal tissue.

       3.2.2.3   <u>**Landfill Partitioning Model**</u>. The landfill partitioning model uses partitioning equations developed for estimating volatilization of contaminants from soil (Jury et al., 1983, 1984, 1990). The Jury equations partition the waste in the landfill to waste, air, and pore water and calculate potential losses from leaching, volatilization, and degradation. The landfill partitioning model evaluates contaminant losses over three separate conditions. The first condition is the daily waste addition in which the waste is in direct contact with the atmosphere. The second condition is the active landfill cell in which the waste is covered by a 6-inch "daily" cover. The third condition is the closed landfill cell in which the waste is covered by a 2-foot-thick landfill cap. The model tracks the average annual landfill concentration and sums and tracks both annual emissions and leaching rates for the active life of the landfill and a set number of years after the

landfill has been closed. The landfill partitioning model is used for the sensitivity, deterministic, and Monte Carlo analyses.

3.2.2.4 Groundwater Model. The concentration of constituents of concern at the residential drinking water well are estimated using the groundwater model EPACMTP. The groundwater modeling is conducted using six surrogate compounds to represent the movement of all constituents of concern through the groundwater pathway. Use of surrogate compounds is necessary to minimize the modeling runs required to model the large number of constituents evaluated via the groundwater pathway for this assessment. For the groundwater modeling, organic constituents are grouped into six categories based on like chemical and physical properties. Sorption potential and hydrolysis rate are the key parameters used to group constituents. For computational efficiency, only the surrogate constituent in each category is modeled. The modeling results for the surrogate are then applied to each constituent in the category. Designation of constituent categories and identification of surrogate compounds are described in detail in Appendix C.

The EPACMPT model is used to conduct the sensitivity, deterministic, and Monte Carlo analyses for the groundwater pathway. Groundwater pathway modeling is performed to determine the groundwater exposure concentrations resulting from the release of waste constituents from the landfill. Precipitation that percolates through the waste unit generates leachate, which can infiltrate from the bottom of the landfill into the subsurface. The waste constituents dissolved in the leachate (as predicted by the partitioning model) are then transported via aqueous phase migration through the vadose zone to the underlying saturated zone and then downgradient to a groundwater receptor well. The exposure concentration is evaluated at the intake point of a hypothetical groundwater drinking water well or receptor well, located at a specified distance from the downgradient edge of the waste management unit.

The objective of the groundwater modeling performed for this listing determination is to compute the amount of dilution and attenuation a contaminant may undergo as it migrates from a landfill to a groundwater well. The amount of dilution and attenuation is expressed as a dilution/attenuation factor (DAF), which represents the ratio of the initial leachate concentration leaving the landfill to the groundwater receptor well concentration.

The groundwater model accounts for the following processes affecting contaminant fate and transport: advection, hydrodynamic dispersion, linear or nonlinear equilibrium sorption, chained first-order decay reactions, and dilution from recharge in the saturated zone. EPACMTP can be run in deterministic mode or Monte Carlo mode. In the deterministic mode, the most sensitive variables are set to their high-end values. All other parameters are set at central tendency. In the probabilistic Monte Carlo mode, parameter values are randomly selected from their respective statistical distributions. The Monte Carlo procedure allows assessment of the uncertainty associated with groundwater well concentrations that result from uncertainty and variability in climatic and hydrogeologic characteristics of waste management units across the range of locations associated with the dyes and pigments industry.

3.2.2.5 Shower Model. The shower model (RTI, 1998) is used to estimate exposures due to indoor household water uses. Dermal and inhalation risks from household water use are

estimated using the modeled groundwater concentration for each constituent of concern as the starting tapwater concentration. Exposure is assessed for three types of household exposures: (1) exposure from being in the shower stall during and immediately after showering (both dermal and inhalation exposures), (2) exposure from being in the bathroom after showering (inhalation exposure), and (3) exposure from being in the rest of the house (inhalation exposure). The shower model used in this analysis is based on the equations presented in McKone (1987) and the shower model construct described by Little (1992). The model estimates the change in the shower air concentration based on the mass of constituent lost by the water (fraction emitted or emission rate) and the air exchange rate between the various model compartments (shower, the rest of the bathroom, and the rest of the house). Risk due to inhalation is estimated for adults and children for constituents volatilized from groundwater. Dermal exposures during bathing or showering are also estimated, but for adults only. Dermal exposure factors are not available for children and there is a high degree of uncertainty associated with the estimated risk to children for this pathway.

3.3   Exposure and Toxicity Assessments

The exposure assumptions used in this analysis are values from the *Exposure Factors Handbook* (EFH) (U.S. EPA, 1997c). The recommended values for the central tendency and high-end intake rates are used in the deterministic analysis, and a distribution of values developed from the data presented in the EFH are used in the Monte Carlo analysis. These values are discussed in Section 6.0 of this document.

The health benchmark data used in the analysis are based on the values presented in the Integrated Risk Information System (IRIS) online database of verified health benchmarks or in the Health Effects Assessment Summary Tables (HEAST) document. The health benchmarks used in the analysis are presented in Appendix D, which documents all constituent-specific data used in the analysis. The studies used as the basis for these benchmarks have been reviewed and are summarized in Appendix E.

3.4   Characterizing Risk

The results of this risk assessment are summarized in Section 7.0 of this document. These results are presented in two forms: (1) as risk-based concentrations of hazardous constituents in dye and pigment waste streams, and (2) as risk-based concentrations of hazardous constituents in the leachate generated from landfilled dye and pigment waste streams. These concentrations are calculated based on the groundwater pathway, which is the most protective pathway for all constituents when disposed of in municipal landfills. The risk-based concentrations are calculated by assuming the residential drinking water well concentration is equal to an EPA-established protective or health-based level for each constituent for the most sensitive receptor (adult or child). Protective concentrations are those at which adverse health effects from any single constituent present in contaminated drinking water and/or water used for bathing or showering do not exceed a 1×10^{-5} individual lifetime cancer risk or a noncancer HQ of 1 at the upper end of the risk distribution. A summary of the resulting risk-limiting waste and leachate concentrations is presented in Section 7.0. Additional details of the results are presented in Appendix F.

4.0 Risk Assessment Scenario

The risk assessment scenario considered in the waste listing decision for deferred wastes from the manufacture of organic dye and pigment products is the disposal of spent filter aids and wastewater treatment sludges from the manufacture of triarylmethane dyes and pigments in municipal landfills in the vicinity of the manufacturing facilities. These waste streams are assumed to be disposed of in municipal landfills. Only those waste streams assumed to be managed in this way are assessed in this analysis.

4.1 Source Characterization

Waste streams assessed in this analysis are the spent filter aid waste stream and the TAM sludge waste stream. Spent filter aids may be generated in a facility manufacturing any of the three classes of dyes and pigment products of concern (azo, triarylmethane, and anthraquinone). TAM sludges are, by definition, restricted to facilities that manufacture TAM dyes or pigments. Filter aids are diatomaceous earth products or similar substances used to coat fixed filters to prevent their clogging during the filtration process, or other absorbents (e.g., activated carbon) used to purify products.

4.1.1 Generation

The spent filter aids waste streams evaluated in this risk assessment are generated at a number of dye and pigment manufacturing facilities; the number is not included at present because of business confidentiality concerns. The number of facilities at which TAM sludge is generated is also not included because of business confidentiality concerns.

4.1.1.1 Processes. Spent filter aids may be generated at any point in the dye and pigment manufacturing process at which liquids are filtered. TAM sludges are wastewater treatment sludge or wastewater storage tank sludge generated during the manufacture of TAM dye or pigment products.

4.1.1.2 Generation Rates. The generation rates for the filter aid waste stream have been claimed as Confidential Business Information (CBI) by most of the facilities that report generating these wastes. However, specific generation rates are not used in the analysis. Instead, the reported waste volumes are entered into a data set from which a distribution of volumes is developed (see Figure 4-1). The 50th and 90th percentile values for waste volumes are pulled from this distribution for use as waste generation rates in the sensitivity and deterministic risk analysis.

For the probabilistic analysis, all values in the data set are used. The spent filter aid generation rate distribution is as follows:

> \# 50th percentile = ___ tonne/yr.*
> \# 90th percentile = ___ tonne/yr.*

The generation rate for one noncommingled TAM sludge waste stream managed in a municipal landfill, as reported in the updated 3007 Industry Questionnaire, is not claimed as CBI and is used as reported. The TAM volume used in this analysis is 57 metric tons per year.

Relevant data are not included at present because of business confidentiality concerns.

Figure 4-1. Filter air waste volumes.

4.1.2   Physical/Chemical Properties

The waste's physical properties data needed for the risk analysis are fraction organic carbon (f_{oc}) and waste bulk density. These data are obtained from separate sources. The foc data are obtained from the sampling and analysis data (total organic carbon, or TOC) collected for the 1994 proposed rule for the dye and pigment industry (SAIC, 1994). The foc data for filter aid wastes are the results from sampling conducted on filter aids from all industry segments and various processes within the industry. EPA does not have sampling and analysis data for TAM sludge waste streams. Therefore, the foc for TAM sludges is assumed to be the same as for other dye and pigment industry wastewater treatment sludges that were sampled and analyzed for the 1994 proposed rule. The value for the dry bulk densities of spent filter aids and TAM sludges is obtained from the EPA survey of all hazardous waste generators (U.S. EPA, 1991). The generator survey requested information on the bulk density of various types of wastes, including spent filter aids and biological wastewater treatment sludge. The central tendency bulk density values from this survey are used for both the deterministic and Monte Carlo analyses because little variability is expected in this parameter and no waste-specific data are available from the sampling and analysis phase of this listing determination. Physical and chemical property data used in the deterministic and Monte Carlo analyses are presented in Table 4-1.

4.1.3   Chemical Composition — Identification of Chemicals of Potential Concern (COPCs)

The primary source of potential constituents of concern is the set of analytical data EPA collected to support the December 22, 1994, proposed rule. These data include sampling and analysis results for all dye and pigment wastes under consideration in the 1994 proposal. EPA used the analytical data to develop a list of chemicals of concern in spent filter aids and TAM sludges based on two primary factors: (1) whether the constituent could reasonably be attributed to the production of the dyes or pigments products at issue, and (2) whether the constituent has an EPA-established health benchmark. All constituents without health benchmarks were excluded from further evaluation. A more detailed description of the analytical data and the

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*Relevant data are not included at present because of business confidentiality concerns.

Table 4-1. Physical Property Data for Dye and Pigment-Deferred Waste Streams

Parameter	Central Tendency Value	High-End Value	Monte Carlo 50th and 90th Percentile Values	Reference
Spent filter aid				
f_{oc}	_____[a]	_____[a]	_____[a] _____[a]	Analytical Data for Proposed Rule
Bulk density	1.07	1.07	1.07	U.S. EPA, 1991
TAM sludge				
f_{oc}	_____[a]	_____[a]	_____[a] _____[a]	Analytical Data for Proposed Rule
Bulk density	1.07	1.07	1.07	U.S. EPA, 1991

[a]Relevant data are not included at present because of business confidentiality concerns.

process used to identify potential constituents of concern is provided in the Listing Background Document (EPA, 1999).

EPA also excluded certain constituents from consideration in this rule that were included in the evaluation of azo dye and pigment wastes in the 1994 proposed rule. These constituents are acetoacetanilide (AAA), acetoacet-o-toluidine (AAOT), and acetoacet-o-anisidine (AAOA). For the 1994 proposal, EPA derived health-based numbers for AAA, AAOT, and AAOA based on a structural activity relationship (SAR) analysis. EPA has since reevaluated and revised the SAR analysis based on comments received in response to the 1994 proposal. The revised analysis, which has been independently peer reviewed, concludes that the current data available are insufficient to make a quantitative estimation of the carcinogenic potential of these compounds or to establish provisional noncancer benchmarks. The revised toxicological analysis for these compounds and the peer-reviewed documents are provided in Appendix A.

EPA assessed 53 organic constituents, 31 noncarcinogenic compounds, and 22 carcinogenic compounds that are potential constituents of concern in wastes from the production of the dye and pigment products under consideration and have human health toxicity benchmarks. In addition, eight metals that may be present in dye and pigment waste streams based on analytical data are considered in this assessment. Risk-based limits are evaluated for all of these compounds for each waste stream. Constituents with cancer endpoints for the oral exposure route and constituents with noncancer oral benchmarks are presented in Table 4-2. The health benchmark data for each of these constituents are presented in Appendixes D and F.

Table 4-2. Carcinogenic and Noncarcinogenic Constituents Evaluated[a]

Carcinogenic		Noncarcinogenic	
Constituent	**CAS No.**	**Constituent**	**CAS No.**
Aniline	62-53-3	Acetone	67-64-1
Azobenzene	103333	Benzaldehyde	100-52-7
Benzene	71-43-2	p-Chloroaniline	106-47-8
Benzidine	92-87-5	Chlorobenzene	108-90-7
Bis(2-ethylhexyl)phthalate	117817	p-Cresol	106-44-5
Bromodichloromethane	75-27-4	1,2-Dichlorobenzene	95-50-1
Chloroform	67-66-3	N-N-Dimethylaniline	121-69-7
1,4-Dichlorobenzene	106467	Diphenylamine	122-39-4
3,3'-Dimethyoxybenzidine	119904	Ethylbenzene	100-41-4
1,2-Diphenylhydrazine	122667	Formaldehyde	50-00-0
Methylene chloride	75-09-2	Methyl isobutyl ketone	108-10-1
N-Nitrosodiphenylamine	86306	Naphthalene	91-20-3
o-Phenylenediamine	95-54-5	Phenol	108-95-2
o-Toluidine	95-53-4	p-Phenylenediamine	106-50-3
p-Toluidine	106490	Pyridine	110-86-1
		Toluene	108883
		1,2,4-Trichlorobenzene	120821
		Xylenes	1330207

[a] The complete list of constituents is not included at present because of business confidentiality concerns.

4.1.4   Waste/Source Management

The waste management practice assessed in this risk analysis is disposal in a municipal landfill near dye and pigment facilities generating spent filter aids or TAM sludge.

4.1.4.1 <u>Quantities Managed</u>. The filter aid waste quantities managed in municipal landfills used for this assessment are from a distribution of waste volumes. As previously noted, these volumes have been claimed as CBI. A distribution of volumes based on the reported volumes is used to generate 50th and 90th percentile values for waste volumes used for the sensitivity and deterministic risk analyses. For the probabilistic analysis, all values in the distribution of volume data set are used.

A dedicated (i.e., noncommingled) TAM sludge waste quantity is reported in the 3007 Questionnaire as being managed in a municipal landfill. Data on this waste stream are not claimed as CBI and the volume of this single waste stream is used for the risk assessment. The assessment assumes that the quantities are sent to a landfill each year and that the remainder of the landfill volume is filled with municipal waste.

4.1.4.2 <u>Waste Management Units Assessed</u>. This section discusses the key parameters used to characterize municipal landfills for this risk assessment. The landfills analyzed are assumed to comply with current municipal landfill management practices regarding landfill cover regulations in 40 CFR 258. These regulations require application of 6 inches of daily cover (soil) and application of a 2-foot soil cap for each annual cell. The landfills are not, however, assumed to have liner systems, because those requirements are not in effect for all existing landfills. The landfill is evaluated as an unlined landfill and does not include a landfill liner or leachate collections system.

Two primary models are used to conduct the deterministic and Monte Carlo assessments for dye and pigment waste streams, the landfill partitioning model and EPA's EPACMTP groundwater model, both of which are described in detail in Section 5.0. The municipal landfill parameters used in the partitioning model and groundwater models for the deterministic and probabilistic analysis are presented in Table 4-3. These parameters are also used in the sensitivity analyses for groundwater and nongroundwater pathways.

The municipal landfill parameters used in this risk assessment are not specific to any location. A national distribution of landfill areas is used for the surface area parameter. The lifetime of the landfill is assumed to be 30 years (U.S. EPA, 1988). There is no national distribution available for the third important parameter, landfill depth. Therefore, the values for municipal landfill depth used in this analysis are a distribution of permitted waste column depths for municipal landfills in the State of Texas. The Texas data are presented in Appendix G. The depth values were assessed for correlation with the associated landfill areas in the Texas permit data and found to correlate with a coefficient of 0.5. This correlation coefficient is used in the probabilistic analysis.

Table 4-3. Partitioning Model—Municipal Landfill Parameters

Parameter	Central Tendency	High End	Monte Carlo Parameter	Reference
Climate Station Loc. Filter Aids TAM sludge	_____[a] Jersey City, NJ	_____[a] Jersey City, NJ	All Locations Jersey City, NJ	3007 Industry Questionnaire
Area (m^2)	60,705	8,094	Empirical Data Distribution 50th percentile - 60,705 m^2 10th percentile -8,094 m^2	U.S. EPA, 1988
Depth	5.5 m	5.5 m	Empirical Data Distribution 50th percentile 10.7 m 10th percentile 3.7 m	Texas Landfill Permit data
Life of landfill (yr)	30		30	U.S. EPA, 1988
Area of active face (cell area) (m^2)	2023.5	269.8	Calculated	Calculated
Time uncovered (h)	12		12	Assumed
Thickness of daily cover (soil) (m)	0.15		0.15	U.S. EPA, 1994a
Thickness of cell cap (soil) (m)	0.60		0.60	U.S. EPA, 1994a

[a]Relevant data are not included at present because of business confidentiality concerns.

> **4.1.4.3 WMU Locations**. The municipal landfill locations receiving dye and pigment wastes are assumed to be located within an 11-mile radius of the waste-generating facility. For this reason, the geographic-specific parameters associated with the landfills are assumed to be similar to the parameters associated with the location of the waste-generating facility.

> **4.1.4.3.1 *Location-Related Parameters***. _____* facilities report producing spent filter aids and one facility reports producing noncommingled TAM sludges. Hydrogeologic and climate parameters used in the fate and transport modeling of the two waste streams are taken from the hydrogeologic and climate stations located nearest each of the facilities being evaluated. Facility locations for the filter aid waste streams are claimed as CBI. However, only the designation of the

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*Relevant data are not included at present because of business confidentiality concerns.

hydrogeologic codes and climate stations nearest the facilities are identified for the spent filter aid analysis. The only location at which dedicated TAM sludges are

**Table 4-4. Nearest Geographic Stations for
Dye and Pigment Manufacturing Facilities Reporting
Spent Filter Aid and TAM Waste Streams**

Met Station	Climatic Code	Hydrogeology Center	Hydrogeologic Code

Relevant data are not included at present because of
business confidentiality concerns.

reported to be generated is used in modeling conducted for TAM sludges. The hydrogeologic and climatic stations from which geographic data are obtained for this analysis and the location codes that correspond to those stations are presented in Table 4-4.

       Location-Related Parameters for Sensitivity and Deterministic Analysis.Landfill location was evaluated in the sensitivity analysis for spent filter aids to determine whether location is a sensitive parameter for the deterministic analysis. The sensitivity analysis is described in detail in Appendix H. Based on the results of the sensitivity analysis, location was determined not to be a sensitive parameter. Because location was not found to be a sensitive parameter, _____,* the central tendency location, is used for the deterministic analysis of spent filter aids. The location codes corresponding to _____,* are Climate Code _____* and Hydrogeologic Code _____*. Meteorologic and hydrogeologic data specific to these location codes are used for the deterministic analysis of filter aids. One location at which non-commingled TAM sludges are generated and disposed of (Jersey City, NJ) is used in all modeling conducted for TAM sludges.

       Because location was not found to be a sensitive parameter for filter aids, the central tendency location is used for the deterministic analysis of filter aids. It is important to note, however, that it is difficult to identify meaningful central tendency and high-end facility locations because of the numerous interrelated variables associated with each location. For instance, infiltration rate, unsaturated zone thickness, and aquifer thickness are among the most important groundwater pathway parameters tied to location. Other location-related parameters, such as windspeed and temperature, are important for nongroundwater pathways. Most locations are likely to have a mix of high-end, central-tendency, and low-end parameter values. In many cases,

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*Relevant data are not included at present because of business confidentiality concerns.

conditions that favor high-end exposure for one pathway may have the opposite effect on another pathway. For instance, locations with high precipitation may be high end for the groundwater pathway but will tend to reduce exposure from the air transport pathways.

Because these waste streams are modeled in municipal landfills, it was anticipated that the groundwater pathway would present the highest risk. Therefore, for the sensitivity analysis for filter aids, the central tendency and high-end locations were selected based on the groundwater pathway. On this basis, _____* was chosen to represent the central tendency case and _____* was chosen to represent the high-end case in the sensitivity analysis. Climate data from the LaGuardia meteorologic station (Climate Code 70) and hydrogeologic data from Edison, NJ, hydrogeology station (Hydrogeologic Code 2) were used for the TAM sludge analysis.

Location-specific variables used in the deterministic analysis include climate parameters, which are used to estimate infiltration rate used for each landfill, and hydrogeologic parameters needed to estimate infiltration rate and fate and transport in the subsurface. The meteorologic parameters included in the partitioning model are annual average precipitation, annual average runoff values, and annual average evapotranspiration values as shown in Table 4-5. Hydrogeologic parameters used in the EPACMTP groundwater model to simulate constituent fate and transport include saturated and unsaturated zone parameters. Values used for these parameters in the analysis are provided in Tables 4-6 and 4-7.

Table 4-5. Deterministic Partitioning Model Hydrogeologic and Climate Data

Parameter	Central Tendency Value	High-End Value	Reference
Spent filter aid			
Precipitation (cm/yr)	_____[a]	_____[a]	NOC, 1992
Evapotranspiration (cm/yr)	_____[a]	_____[a]	Geraghty et al., 1973
Runoff (cm/yr)	_____[a]	_____[a]	Geraghty et al., 1973
Infiltration (m/yr)	_____[a]	_____[a]	Calculated
TAM sludge			
Precipitation (cm/yr)	109	109	NOC, 1992
Evapotranspiration (cm/yr)	47.3	47.3	Geraghty et al., 1973
Runoff (cm/yr)	25.4	25.4	Geraghty et al., 1973
Infiltration (m/yr)	0.363	0.363	Calculated

[a]Relevant data are not included at present because of business confidentiality concerns.

*Relevant data are not included at present because of business confidentiality concerns.

**Table 4-6. Dyes and Pigments Listing Determination, Filter Aids Waste Stream
List of EPACMTP Input Parameters for Climate and Hydrogeology***

*Relevant data are not included at present because of business confidentiality concerns.

Table 4-7. Dyes and Pigments Listing Determination, TAM Waste Stream List of EPACMTP Input Parameters for Landfills

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic CT	Deterministic HE	Data Source	Monte Carlo Variable?	Monte Carlo[a] Min	Monte Carlo[a] Max	Data Source
				or Constant				or Constant		
Waste Parameters										
Waste concentration	mg/kg	Industry-specific	No	Concentration profile		RTI	Yes	Surrogate-specific		RTI
Leachate concentration	mg/L	Industry-specific	No	Surrogate-specific		RTI	Yes	Surrogate-specific		RTI
Bulk density of waste	g/cm^3	Industry-specific	No	1.07			No	1.07		
Site Parameters										
Area	m^2	National	Yes	60,705	(1) 420,888 (2) 8,090	EPA OSW Municipal Landfill Survey, U.S. EPA, 1988.	Yes	4,088	9.35E+6	Same
Recharge rate	m/yr	Regional	No	0.2944		HELP model data for the closest HWIR climate site; U.S. EPA, 1997a, 1997b.	Yes	0.051	0.75	Same

(continued)

US EPA ARCHIVE DOCUMENT

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic CT	HE	Data Source	Monte Carlo Variable?	Monte Carlo[a] Min	Max	Data Source
				or Constant				or Constant		
Infiltration rate	m/yr	Regional	No (Loc)	0.2944		HELP model data for the closest HWIR climate site; U.S. EPA, 1997a, 1997b.	Yes	0.013	0.36	Same
Source duration	yr	Derived	No	NA		Leaching profile provided by RTI's partitioning model	Yes	Surrogate- specific		Pulse source, which approximates leaching profile
Landfill depth	m	National	No	2.63		1986 EPA Survey of Industrial Subtitle D Waste Facilities (OPPI survey), as described in U.S. EPA, 1997a	Yes	0.914		Derived to fit waste amount in the landfill
Unsaturated Zone Parameters										
Saturated conductivity of the soil	cm/h	Soil-type-specific	No	14.59		Carsel and Parish (1988)	Yes	1.45E-4	26.15	Same

(continued)

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic CT	Deterministic HE	Data Source	Monte Carlo Variable?	Monte Carlo[a] Min	Monte Carlo[a] Max	Data Source
				or Constant				or Constant		
Alpha (moisture retention)	L/cm	Soil-type-specific	No	0.124		Carsel and Parish (1988)	Yes	1.09E-3	0.19	Same
Beta (moisture retention)	Unitless	Soil-type-specific	No	2.28		Carsel and Parish (1988)	Yes	1.07	2.36	Same
Residual water content	Unitless	Soil-type-specific	No	0.057		Carsel and Parish (1988)	Yes	0.017	0.112	Same
Saturated water content	Unitless	Soil-type-specific	No	0.41		Carsel and Parish (1988); constant for given soil type	Yes	0.41	0.45	Same
Unsaturated zone thickness (depth to water table)	m	Regional	No (Loc)	9.14		API HGDB (EPA, 1997a, 1997b)	Yes	0.31	610.0	Same
Percent organic matter	Unitless	Soil-type-specific	No	0.074		Carsel et al. (1988)	Yes	6.69E-3	1.62	Same
Bulk density	g/cm^3	Soil-type-specific	No	1.60		Carsel et al. (1988)	Yes	1.60	1.67	Same
Saturated Zone Parameters										
Particle diameter	cm	Empirical	No	0.025		Shea, 1974	Yes	4.07e-4	0.204	Same

(continued)

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic		Data Source	Monte Carlo Variable?	Monte Carlo[a]		Data Source
				CT	HE			Min	Max	
				or Constant				or Constant		
Effective porosity	Unitless	Derived	No	5.14E-2		Derived from particle diameter; McWorter and Sunada, 1977	No	0.0502	0.422	Same
Bulk density	g/cm³	Derived	No	1.58		Derived from porosity; Freeze and Cherry, 1979	No	1.16	1.8	Same
Saturated zone thickness (aquifer thickness)	m	Regional	No (Loc)	22.9		API HGDB (U.S. EPA, 1997a, 1997b)	Yes	0.31	914.0	Same
Hydraulic conductivity (K_x)	m/y	Regional	No (Loc)	284		API HGDB (U.S. EPA, 1997a, 1997b)	Yes	3.15	3.19E+6	Same
Hydraulic gradient	Unitless	Regional	No (Loc)	0.015		API HGDB (U.S. EPA, 1997a, 1997b)	Yes	2.0E-6	0.348	Same
Longitudinal dispersivity	m	Empirical, based on well location	No	8.29	5.29 HE(x); 8.29 HE(y)	U.S. EPA, 1997b, Gelhar et al., 1992	Yes	0.137	324	Same

(continued)

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic		Data Source	Monte Carlo Variable?	Monte Carlo[a]		Data Source
				CT	HE			Min	Max	
				or Constant				or Constant		
Transverse dispersivity	m	Derived	No	1.04	0.66 HE(x); 1.04 HE(y)	Derived from a_L (U.S. EPA, 1997b)	Yes	0.0171	40.5	Same
Vertical dispersivity	m	Derived	No	0.05	0.03 HE(x); 0.05 HE(y)	Derived from a_L (U.S, EPA, 1997b)	Yes	8.54E-4	2.03	Same
Groundwater temperature	°C	Regional	No	14.4		U.S. EPA, 1997a and 1997b, based on location	Yes	12.5		Same
Groundwater pH	Unitless	National	No	6.8		EPA's STORET database for HWIR '95 analysis; U.S EPA, 1997b	Yes	3.2	9.64	Same

(continued)

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic		Data Source	Monte Carlo Variable?	Monte Carlo[a]		Data Source
				CT	HE			Min	Max	
				or Constant				or Constant		
Fraction organic carbon	Unitless	National	No	0.000432		EPA's STORET database; Johnson distribution developed for HWIR '95 analysis; U.S. EPA, 1997b	Yes	1.98E-5	7.79E-3	Same
Receptor Well Parameters										
X-distance to well (X-well)	m	National	Yes	430	102	EPA OSW survey of landfills: U.S. EPA, 1997b	Yes	0.82	1601	Same
Y-distance to well (Y-well)	m	Uniform	Yes	118	0	CT = half-way to edge of plume HE = on plume centerline	Yes	1.1e-3	1617	Uniform distribution, constrained to lie within the areal extent of the plume, U.S. EPA, 1997b

(continued)

Table 4-7. (continued)

Parameter	Units	Variable Type	Sens. Analysis Variable?	Deterministic CT	Deterministic HE	Data Source	Monte Carlo Variable?	Monte Carlo[a] Min	Monte Carlo[a] Max	Data Source
				or Constant				**or Constant**		
Receptor well depth (Z-well)	m	Uniform	No	Sensitivity Analysis			Yes	1.78E-3	9.98	Uniform distribution throughout aquifer thickness or throughout upper 10 m of aquifer, whichever is less.
				8.65	1.80	Well placed at the vertical midpoint of the plume				
				Deterministic Analysis (2HE Analysis)						
					5	Well placed in middle of the upper 10 m of aquifer, since aquifer thickness is greater than 10 m				
Averaging time for groundwater concentration	yr	Constant	Yes	9	30	Corresponds to 9- or 30-year exposure duration in the risk calculation	No	9		Same

[a]The Min and Max values listed here for the Monte Carlo analysis are based on model results for Surrogate 1.

For the deterministic analysis of spent filter aids, the climate parameter values used for the central tendency location are median values from the distributions of values for Climate Code ___.* For the TAM deterministic analysis, median values from the distributions of data for Climate Code 70 are used. The hydrogeologic parameter values for the central tendency location for the spent filter aids analysis are the median values from the distributions of data for HydroGeoLogic Code ___.* For Jersey City, NJ, the single TAM location considered, the hydrogeologic parameters used are the median values from the distributions of data for Hydrogeologic Code 2. The hydrogeologic and climate data used in the EPACMTP groundwater model for the deterministic analysis are presented in Table 4-6 for spent filter aids and 4-7 for TAM sludge waste. The location-related data used in the partitioning model for the deterministic analysis are presented in Table 4-8.

Location-Related Parameters for Monte Carlo Modeling. For the Monte Carlo analysis, location is a randomized parameter. For the spent filter aid analysis, specific locations from among the spent filter aid locations were randomly selected and the geographic variables associated with that location were used in the model. Each location is associated with one of the hydrogeologic and climate codes. Each hydrogeologic code is associated with a distribution of saturated and unsaturated zone parameters. Each meteorologic code is associated with a distribution of climate parameters. Once a specific location (and associated parameters) is selected, specific parameter values are randomly selected from the distribution of values for the climate and hydrogeologic codes corresponding to the location selected. For the probabilistic analysis for TAM sludges, hydrogeologic and meteorologic parameters were randomly selected from the distribution of parameters corresponding to the single TAM location modeled.

Table 4- 8. Monte Carlo Climate Inputs for Partitioning Model

Climatic Code		
Ave year precip (cm/yr)	___ [a]	
Ave year irrigation (cm/yr)	___ [a]	
Ave year evap. (cm/yr)	___ [a]	
Ave year runoff (cm/yr)	___ [a]	
Ave year infiltration (cm/yr)	___ [a]	

[a]Relevant data are not included at present because of business confidentiality concerns.

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*Relevant data are not included at present because of business confidentiality concerns.

Location-specific variables used in the probabilistic analysis include the meteorologic parameters needed to estimate infiltration rate and the hydrogeologic parameters used to estimate infiltration and subsurface fate and transport. The meteorologic parameters required are annual average precipitation values, annual average runoff values, and annual average evapotranspiration values. These parameters are presented in Table 4-8 for partition modeling.

4.2   Description of Receptors and Exposure Pathways

This section describes receptor and exposure pathways included in the deterministic and probabilistic assessment of risks from groundwater pathways only. EPA conducted a sensitivity analysis of nongroundwater paths and, based on the results of that analysis, determined that additional evaluation of nongroundwater pathways was unnecessary. The Agency was able to use results from the sensitivity analysis to determine that nongroundwater risks associated with each constituent were lower than risks for each constituent through the groundwater pathway. Therefore, deterministic and probabilistic analyses were not conducted for nongroundwater pathways. Receptors assessed in the nongroundwater sensitivity analysis were the farmer and farm child. The nongroundwater pathways assessed in the sensitivity analysis include inhalation of vapors and deposition of vapors on vegetation near municipal landfills receiving dye and pigment waste streams. The sensitivity analysis for the nongroundwater pathway is described in detail in Appendix B.

4.2.1   Human Receptors

Human receptors assessed in this risk analysis for the groundwater pathways are adults and children living near the municipal landfill receiving dye and pigment waste. These individuals are assumed to obtain all drinking water from a residential well located within the contaminated plume from this landfill. The receptor is assumed to be inhabitants of the individual residence nearest to the municipal landfill with a residential drinking water well. These individuals are also assumed to use this water for all household water uses, including bathing or showering.

4.2.2   Exposure Pathways

The selected receptor for the groundwater pathway is a hypothetical resident who obtains tap water for drinking and all other household water uses from a groundwater well. Therefore, the exposure point is a residential water well that is downgradient of a waste management unit containing dye and pigment industry wastes. The residential well location parameters assumed in the deterministic analysis are presented in Tables 4-8 and 4-9.

For the deterministic analysis, the receptor well for both TAM sludges and spent filter aid is located downgradient from the landfill at the high-end distance of 102 m. For spent filter aids, the lateral location of the well (Y-well) is placed at the central tendency value of half-way to the edge of the plume or 118 meters. For TAM sludges, the Y-well distance is set at high end or the plume center line for noncarcinogens and at the central tendency distance of 118 meters for carcinogens. For both spent filter aids and TAM sludges, the depth of the well intake point (Z-

Table 4-9. Exposure Pathways for Groundwater Risk Analysis

Pathways	Adult Farmer	Child of Farmer
Ingestion Pathways		
Ingestion groundwater	✓	✓
Inhalation Pathways		
Inhalation of vapors during household water use (bathing and showering)	✓	✓
Dermal Pathways		
Dermal exposure during bathing and showering	✓	[a]

[a] Dermal exposure factors are not available for children and there is a high degree of uncertainty associated with the estimated risk to children for this pathway.

well) was chosen to be the smaller of 5 m and half the aquifer thickness; thus Z-well for both waste streams is 5 m.

For the Monte Carlo analysis, the receptor well is located downgradient from the waste management unit at a radial distance of up to 1,610 m (EPA survey, as cited in U.S. EPA, 1997b). The lateral location of the well (Y-well) is assumed to be randomly distributed within the estimated lateral extent of the plume. The depth of the well is varied uniformly throughout the aquifer thickness or throughout the upper 10 m of the aquifer thickness, whichever is less. Thus, the well depth is never allowed to exceed 10 m below the water table.

The exposure pathways for the groundwater risk analysis are presented in Table 4-9.

5.0 Estimating Exposure Point Concentrations

5.1 Exposure Pathways for Human Receptors

The primary receptors considered in this analysis were adult and child receptors exposed to water from groundwater wells contaminated by leachate from municipal landfills receiving spent filter aids or triarylmethane sludge waste streams. The exposure pathways were ingestion of contaminated drinking water and showering and/or bathing with contaminated water. The evaluation of groundwater pathways included a sensitivity analysis and deterministic and probabilistic assessments of risk.

Only a sensitivity analysis is conducted for nongroundwater pathways. Receptors evaluated for nongroundwater pathways in the sensitivity analysis were farmers and farm children. Nongroundwater pathways evaluated in the sensitivity analysis were both direct (e.g., inhalation of vapors from landfills) and indirect exposure pathways (e.g., ingestion of food products contaminated by emissions of volatile constituents from the landfill).

5.2 Determining Exposure Point Concentrations for Human Receptors

5.2.1 Methodology for Determining Exposure Point Concentrations

5.2.1.1 Sensitivity Analyses. The initial phase of this risk analysis was a sensitivity analysis. The sensitivity analysis was conducted to identify and rank the most influential variable parameters in the analysis. The first step of a sensitivity analysis is to set all variable parameters at central tendency values and calculate central tendency risk estimates. Then, one at a time, each variable parameter is set to its high end value, risk is calculated, and the variation in the risk from the central tendency value is noted. The parameters having the greatest effect on the resulting risks are identified as the most sensitive variables. The two most sensitive parameters identified by this method are set to high end in the deterministic analysis. The sensitivity analyses for both the nongroundwater and groundwater pathways are described briefly in this section. A more detailed description of the sensitivity analysis for the nongroundwater pathways is presented in Appendix B. The groundwater pathway sensitivity analysis is described in more detail in Appendix H.

5.2.1.1.1 *Sensitivity Analysis for Nongroundwater Pathway*. The nongroundwater sensitivity analysis considered the emissions of volatile constituents from the landfill and their dispersion and deposition at receptor locations. The exposure pathways evaluated for the nongroundwater analysis were direct inhalation of the vapors and the ingestion of food chain products that are contaminated by the wet and dry deposition of vapors. Receptors for the nongroundwater pathways were assumed to be farmers and their children because these are considered to be the most highly exposed individuals.

TAM and filter aid waste streams were evaluated separately for the nongroundwater sensitivity analysis. Therefore, the waste-stream-specific parameters for volume, bulk density, and fraction organic carbon are specific to each waste stream. However, due to CBI constraints, a single set of constituents is evaluated for both TAM sludge and spent filter aids.

Parameters varied in the nongroundwater sensitivity analysis are listed below. Specific central tendency and high-end values for each parameter used in the nongroundwater sensitivity analysis are provided in Appendix B.

   #   Waste quantity (metric tons)

   #   Waste stream f_{oc}

   #   Geographic location
           -      Precipitation (cm/yr)
           -      Evapotranspiration (cm/yr)
           -      Runoff (cm/yr)

   #   Landfill dimensions
           -      Area (m^2)
           -      Depth (m)

   #   Distance to receptor (m)

   #   Exposure factors.

Models used to conduct the nongroundwater sensitivity analyses include the partitioning model, the EPA Industrial Source Complex Short Term, Version 3 (U.S. EPA, 1998a) air dispersion model and the Indirect Exposure Model. Air dispersion and exposure modeling for the nongroundwater sensitivity analysis is described in Appendix B. The partitioning model is described in detail below.

The partitioning model estimates volatile emissions and leachate concentrations from a landfill simultaneously to maintain a mass balance between the pathways. The parameters included in the partitioning model are:

   #   Waste quantity (metric tons)

   #   Waste stream f_{oc}

Geographic location
-        Precipitation (cm/yr)
-        Evapotranspiration (cm/yr)
-        Runoff (cm/yr)

Landfill dimensions
-        Area (m^2)
-        Depth (m).

For the first partition modeling run for the sensitivity analysis, all parameters were set to their median values for the central tendency case. Then, to determine the relative sensitivity of each parameter, each of the variable parameters was set to its high-end value while all the other parameters were kept at their central tendency values. The location parameter was considered a single parameter and all meteorologic variables associated with location were varied together as a unit from central tendency to high end. Each partitioning model run produced a volatile emission rate that was then used in the ISCST3 air dispersion model.

The ISCST3 modeling included some of the same parameters used in the partitioning model. It also included a variable for distance to receptor that is not included in the partitioning model. The parameters varied in the ISCST3 model were

Location
Landfill areas
Distance to receptor.

ISCST3 dispersion and deposition modeling was conducted four times for the nongroundwater sensitivity analysis. First, all parameters were set at central tendency. Then the model was run once with each of the three variable parameters set to high end and the other two fixed at central tendency. The location parameter was considered a single parameter, and all meteorologic variables associated with location were varied together from cental tendency to high end.

Parameters shared between the partitioning model and the air modeling were coordinated in the risk modeling. For example, the location parameter was used in the partitioning model to produce an emission rate. ISCST3 modeling for that emission rate was conducted using data for the same location. Landfill area was similarly coordinated between the two models. The variables used in the ISCST3 portion of the nongroundwater sensitivity analysis are presented in Appendix B.

The remaining variables considered in the nongroundwater sensitivity analysis were the following exposure factors:

Ingestion of soil (child only)
Ingestion of home-grown exposed vegetables
Ingestion of home-grown exposed fruit
Ingestion of home-grown root vegetables

Ingestion of home-produced beef
Ingestion of home-produced milk
Ingestion of home-caught fish.

Other exposures considered but not varied in the sensitivity analysis were inhalation of vapors and adult ingestion of soil. The sensitivity of the varied exposure parameters was determined by setting each parameter at high end while all other variable parameters were set at central tendency as described above.

The sensitivity analysis modeling produced a risk estimate associated with the use of a single high-end parameter for each modeling run. The most sensitive parameters were those that produced the highest estimate of risk when set at their high-end values. Originally, the sensitivity analysis was intended to be used solely to identify the most sensitive parameters for use in a deterministic analysis of nongroundwater risks from dye and pigment waste streams. However, EPA was able to use the results of the nongroundwater sensitivity analysis as a screening tool to screen out or eliminate nongroundwater risks as a primary concern for dye and pigment industry wastes.

Nongroundwater risks can be screened out by comparing the results of the nongroundwater sensitivity analysis to the results of the groundwater sensitivity analysis, which was done using the same inputs (as described in Section 5.2.1.1.2). In all cases, the groundwater risk analysis produced higher risk estimates for all constituents. Because the purpose of this analysis was to set risk-based concentration limits, only the pathways of most concern or those that present the highest risk were needed to determine protective concentrations. Based on the results of the sensitivity analysis, groundwater pathways were found to be most limiting (i.e., to present the highest risk) in all cases for all constituents of concern; therefore, only the groundwater pathway required further evaluation using deterministic and Monte Carlo analyses. In other words, risk-based concentrations set based on groundwater pathway risks will also be protective of nongroundwater pathway risks. Based on this finding, no further modeling of nongroundwater risks was conducted.

5.2.1.1.2 *Sensitivity Analysis for Groundwater Pathway*. This section briefly describes the groundwater pathway sensitivity analysis methodology and results. A more detailed description of this analysis is provided in Appendix H.

Only the spent filter aids waste stream was evaluated in the sensitivity analysis conducted for the groundwater pathway. Because of the similarity between the two waste streams, the sensitivity analysis results for the filter aids waste stream are assumed to be applicable to the TAM sludge waste stream; thus, a separate sensitivity analysis was not performed for the TAM sludge waste stream.

For the groundwater sensitivity analysis, constituents were evaluated in the partitioning model to determine potential leachate concentrations. Landfill leachate concentrations for each constituent were then modeled in the subsurface using EPA's EPACMTP groundwater model. EPACMTP modeling was conducted using six surrogate compounds to represent the movement of all constituents of concern through the groundwater pathway. Identification of surrogate

compounds provides a means of minimizing the modeling runs required to model the large number of constituents evaluated for this assessment. A detailed description of the methodology used to categorize and identify surrogates for the organic and inorganic constituents considered is provided in Appendix C.

For the groundwater modeling, organic constituents were grouped into six categories based on like chemical and physical properties. As described in greater detail in Appendix C, the factors used for this selection process are sorption potential (K_{OC}) and hydrolysis rate (λ). Once the constituents were grouped based on these factors, the constituent in each group with the lowest K_{OC} and λ was selected as the surrogate compound for the group and only the surrogate compound was modeled. The groundwater fate and transport modeling results for each surrogate were then applied to each constituent in the respective surrogate category. At this initial stage, metals of potential concern were also evaluated in the groundwater sensitivity analysis. Individual metals were assigned to the appropriate organic surrogate groups for groundwater modeling (as explained in Appendix C).

The following parameters were included in the EPACMTP groundwater modeling for each surrogate (specific central tendency and high-end values used for each parameter are provided in Appendix H):

Annual waste amount

Landfill area

Waste unit location, including
-      Infiltration and recharge rate
-      Soil type and properties
-      Depth to groundwater
-      Saturated zone thickness
-      Aquifer hydraulic conductivity
-      Hydraulic gradient

Distance to receptor well (X-well)

Distance of receptor well from the plume centerline (Y-well).

The EPACMTP sensitivity analysis modeling was conducted by performing a number of modeling runs for each surrogate. First, all parameters were set to their median values for the central tendency case. Then, each of the varied parameters was set to its high-end value and all the other parameters were kept at their central tendency values. Each high-end run results in a higher receptor well concentration and, thus, a lower dilution and attenuation factor (DAF) than the central tendency case for that surrogate. The two most sensitive modeling parameters were the two with high-end runs that resulted in the lowest DAFs.

The sensitivity analysis modeling results were tabulated and the parameters ranked in order of sensitivity for each surrogate. The two most sensitive parameters were identified for use

in the subsequent deterministic analysis. The two most sensitive modeling parameters for each surrogate are presented in Tables 5-1 and 5-2. A more detailed presentation and discussion of results is provided in Appendix H.

Because no separate sensitivity was conducted for the TAM sludges waste stream, the two most sensitive parameters were identified by extrapolating the results obtained for the filter aids waste stream. However, because the TAM sludges analysis is limited to one volume from one facility, waste amount and location were excluded from consideration as sensitive parameters for this waste stream. The third most sensitive parameter identified for the filter aids waste stream is the Y-well distance. Therefore, the two most sensitive parameters for the TAM sludges waste stream were assumed to be: (1) the X-well and Y-well distance to the receptor well for non-carcinogens, and (2) the X-well distance to the receptor well and exposure duration for carcinogens.

Table 5-1. Two Most Sensitive Parameters for Spent Filter Aids Waste Stream

Surrogate	Carcinogens	Noncarcinogens
1	Waste amount and exposure duration	Waste amount and x-well
2	Waste amount and exposure duration	Waste amount and x-well
3	Waste amount and exposure duration	Waste amount and x-well
4	Waste amount and exposure duration	Waste amount and x-well
5	Waste amount and exposure duration	Waste amount and x-well
6	Waste amount and exposure duration	Waste amount and x-well

Table 5-2. Two Most Sensitive Parameters for TAM Sludges Waste Stream

Surrogate	Carcinogens	Noncarcinogens
1	X-well and exposure duration	X-well and Y-well
2	X-well and exposure duration	X-well and Y-well
3	X-well and exposure duration	X-well and Y-well
4	X-well and exposure duration	X-well and Y-well
5	X-well and exposure duration	X-well and Y-well
6	X-well and exposure duration	X-well and Y-well

The results of the groundwater sensitivity analysis indicate that metals contained in dye and pigment waste streams do not pose risks of concern. In most cases, the metal constituents do not reach the residential well within the period of modeling (10,000 years). Details of the analysis and results are presented in Appendix H.

5.2.1.2 Deterministic Analysis for Groundwater Pathway. The purpose of the groundwater pathway deterministic analysis for the filter aids and TAM sludge waste streams was to estimate constituent-specific threshold leachate and waste concentrations from the scenario producing a reasonable high-end exposure concentration. The reasonable high-end exposure scenario was defined by setting the two most sensitive parameters (identified in the sensitivity analysis) to their high-end values while all other parameters were set to their central tendency values. In addition, central tendency risk was estimated for comparison with all parameters set to central tendency values.

The following sections describe the groundwater modeling methodology and results of the deterministic analysis. Section 5.2.1.2.1 describes the deterministic analysis performed for organic constituents; Section 5.2.1.2.2 discusses the deterministic analysis performed for metals.

5.2.1.2.1 *Analysis for Organic Constituents*. The groundwater pathway deterministic analysis requires the use of the same three basic models used in the sensitivity analysis:

> \#   Partitioning model to estimate leachate concentrations exiting the landfill

> \#   Groundwater transport model (EPACMTP) to estimate fate and transport of constituents to the residential well

> \#   Exposure models for estimating risk from ingestion and noningestion uses of contaminated groundwater.

Model Linearity. The deterministic risk analysis depends on the linear characteristics of each of the model segments in the analysis with respect to initial waste concentration. Linearity was examined in the partitioning model and the groundwater fate and transport model prior to performing the deterministic analysis. To check for linearity, all variables in the partitioning model were set at central tendency values, and initial waste concentrations of 10,000 mg/kg, 1,000 mg/kg, and 100 mg/kg were entered into the partitioning model for each of the groundwater surrogate constituents. The resulting annual leachate concentrations and the highest 9- and 30-year average leachate concentrations were verified as linear with the initial waste concentration. Then, the leachate concentrations were used as inputs to the EPACMTP groundwater fate and transport model (all other variables held in the EPACMTP analysis were set at central tendency values), and the resulting residential well concentrations were verified as linear with the initial leachate concentration. This result ensured that the relationship between the leachate concentration and the well concentration DAF was constant for all waste/leachate concentrations. Therefore, groundwater modeling could be performed for a single surrogate concentration, and the resulting DAF was assumed appropriate for all leachate concentrations for all constituents in the surrogate group under the same model conditions. This linearity is shown graphically in

Figure 5-1, which shows the leaching profiles for the six groundwater surrogates at initial waste concentrations of 100 ppm, 1,000 ppm, and 10,000 ppm.

Overview of Deterministic Analysis for Organics. The dyes and pigments integrated deterministic analysis is a multistep process:

Step 1: The initial waste concentration is set at 1,000 mg/kg in the partitioning model, and the model is run for each of the six groundwater surrogate compounds. The model generates annual leachate concentrations for each of the next 1,000 years for each surrogate.

Step 2: The EPACMTP model takes the 1,000 annual leachate concentrations for each of the six surrogate compounds and generates 9- and 30-year average groundwater DAFs (DAF^{GW}) for each surrogate.

Step 3: The partitioning model is then run for each individual constituent in each waste stream. The initial waste concentration is set at a fixed concentration (initial setting, 1,000 mg/kg) in the partitioning model, and the model is run to generate annual average leachate concentrations for each constituent in each waste stream.

Step 4: The appropriate surrogate DAF^{GW}s are applied to the constituent-specific leachate concentrations generated in Step 3 for each constituent within each of the six surrogate groups. This yields a constituent-specific average 9- or 30-year average residential well concentration for each constituent.

Step 5: The appropriate exposure factors and constituent-specific health benchmark values are used to estimate risk for each exposure route (ingestion of drinking water, inhalation during household use, and dermal exposure during bathing or showering) due to the initial waste concentration fixed in Step 3.

Step 6: The ratio of the risk estimated in Step 5 to the target risk (for carcinogens, the risk = 1E-05; for noncarcinogens, the HQ = 1) is then applied to the initial waste or leachate concentration to develop a new initial waste or leachate concentration that should yield a risk estimate close to the target risk level (see linearity discussion in Section 5.2.1.2.1). Steps 3 through 5 are then repeated iteratively until the initial concentration, which yields the exact target risk level, is determined. This becomes the allowable waste concentration. This same process is used to determine allowable leachate concentrations.

Step-by-Step Description of Deterministic Analysis for Organics.

Step 1: In the partitioning model, the initial waste concentration is set at 1,000 mg/kg, and the model is run for each of the six groundwater surrogate compounds. The model generates 1,000 years of annual leachate concentrations for each surrogate.

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Figure 5-1. Verification of model linearity.*

*Relevant data are not included at present because of business confidentiality concerns.

A spreadsheet calculation model was used to estimate the partitioning of constituents in the landfill. The partitioning model, described in Section 5.2.2.1, was used to simulate the fate and transport processes that occur within the landfill. The modeling simulates daily additions of waste during the active life of the landfill and for subsequent years after closure. The output of the model was 1,000 annual average leachate concentrations (for 30 years of the active life of the landfill and for 970 years after closure).

As the first step in the deterministic analysis, the initial waste concentration in the partitioning model was set at 1,000 mg/kg for each of the six surrogate compounds. For spent filter aids, the waste quantity was set to high end and all other inputs in the partitioning model were set to central tendency values. These values are presented in Section 4, Table 4-8. For TAM sludge waste, only a single waste volume and a single location were evaluated, so all values in the partitioning model were set to central tendency values. These values are presented in Section 4, Table 4-9. The partitioning model was run, and 1,000 annual leachate concentrations were generated for each of the six surrogate compounds for each waste stream.

Step 2: The EPACMTP model takes the 1,000 annual leachate concentrations for each of the six surrogate compounds and generates 9- and 30-year average DAF^{GW}s for each surrogate.

The annual average leachate concentration outputs were used as inputs to the EPACMTP groundwater model. To model the surrogate-specific leachate profiles accurately, the EPACMTP model was modified by adding a capability to accept an annually varying leachate concentration as input to the subsurface fate and transport simulation.

The deterministic analysis consisted of two modeling runs for each surrogate constituent: a central tendency run and a two-parameter high-end run. For the central tendency scenario, all input parameters were set to their central tendency (median) values. For the high-end scenario, the two most sensitive parameters were set to their high-end values, and the remainder of the input parameters were set to their central tendency values. The central tendency and high-end EPACMTP model input parameters are listed in Tables 4-8 and 4-9 in Section 4.0. Using the surrogate-specific leachate profiles as input, EPACMTP was run to obtain the receptor well concentrations.

Deterministic runs of the EPACMTP groundwater model were evaluated in terms of their predicted receptor well exposure concentration and the corresponding groundwater dilution and attenuation factor (DAF^{GW}). The receptor well exposure concentrations were calculated based on the maximum 9- and 30-year average receptor well concentrations over the next 10,000 years.

For constituents with noncancer endpoints, the DAF^{GW} is generally defined as the ratio of the 9-year maximum average leachate concentration and the corresponding maximum 9-year time-average receptor well concentration. However, when exposure duration is one of the high-end parameters for a carcinogen, then the DAF^{GW} is defined as the ratio of the 30-year maximum average leachate concentration and the corresponding maximum 30-year time-average receptor well concentration.

Different constituents have different leaching profiles. Nevertheless, a number of salient features can be distinguished. The leachate concentration increases with time during the operational life of the landfill (30 years). Thereafter, the leachate concentration decreases with time. How quickly the leaching drops off after the landfill is closed varies greatly between the different constituents. One of the key factors controlling leaching is the organic carbon partition coefficient (K_{OC}). Constituents with low K_{OC} values tend to leach out relatively easily. For a given waste concentration, their maximum leachate concentration will be higher than constituents that have a high K_{OC}, but their leaching also will tend to diminish more quickly than high K_{OC} constituents. In Figure 5-2, the compounds _____* (surrogate 1) and pyridine (surrogate 2) show the leaching behavior of low K_{OC} constituents. _____* (surrogate 5) depicts the behavior of a high K_{OC}, strongly sorbing constituent. Leaching will continue for this constituent at a nearly constant, albeit low, concentration level for an extended time period.

Modeling Results for Step 2. Tables 5-3 and 5-4 show the DAF^{GW} values calculated for carcinogens and non-carcinogens for central tendency and high-end scenarios for each surrogate in the filter aid and TAM waste streams, respectively.

Table 5-3. DAF^{GW} for Organic Surrogates in Filter Aids Waste Stream

Surrogate No.	Constituent Name	K_{oc} (L/kg)	λ (yr^{-1})	CT Case		W and X-Well		W and Duration
				C	NC	C	NC	C
1[a]	_____ [b]	___ [b]	___ [b]	[b]	[b]	[b]	[b]	___ [b]
2	Pyridine	2.2	0.0	12.3	12.2	5.1	5.1	15.8
3[a]	_____ [b]	___ [b]	___ [b]	[b]	[b]	[b]	[b]	___ [b]
4	Naphthalene	1,096	0.0	12.5	12.9	5.2	5.4	12.6
5[a]	_____ [b]	___ [b]	___ [b]	[b]	[b]	[b]	[b]	___ [b]
6[a]	_____ [b]	___ [b]	___ [b]	[b]	[b]	[b]	[b]	___ [b]

The header row "Two-Parameter High-End Cases" spans the "W and X-Well" and "W and Duration" columns.

C = Carcinogen.
NC = Noncarcinogen.
W and X-well = High-end waste amount and X-well
W and duration = High end waste amount and exposure duration
[a] These compounds were used in the original analysis but were later dropped from the list of constituents of concern evaluated for this analysis. They were nevertheless retained as surrogate constituents for the compounds assessed.
[b] Relevant data are not included at present because of business confidentiality concerns.

*Relevant data are not included at present because of business confidentiality concerns.

Figure 5-2. Comparison of leachate profile and effective leaching duration for each surrogate.*

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*Relevant data are not included at present because of business confidentiality concerns.

Table 5-4. DAF^{GW} for Organic Surrogates in TAM Sludges Waste Stream

Surrogate No.	Constituent Name	K$_{oc}$ (L/kg)	λ (yr^{-1})	CT Case		Two-Parameter High-End Cases		
						X- and Y-Well		X-well and Duration
				C	NC	C	NC	C
1[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
2	Pyridine	2.2	0.0	161.6	156.0	3.7	3.7	4.4
3[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
4	Naphthalene	1096	0.0	155.4	161.1	3.9	4.0	4.0
5[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
6[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]

C = Carcinogen.
NC = Noncarcinogen.
X-Well and Y-well = X-well and distance from the plume centerline.
X-Well and Duration = X-well and exposure duration.
Y-Well = Distance from the plume centerline.

[a] These compounds were used in the original analysis but were later dropped from the list of
   constituents of concern evaluated for this analysis. They were nevertheless retained as surrogate
   constituents for the compounds assessed.

[b] Relevant data are not included at present because of business confidentiality concerns.

        For a given scenario, there is little difference in DAF^{GW} values for the different surrogates.
The similarity of the DAF^{GW} values for all surrogates is not surprising because the surrogates are
primarily characterized by different K$_{OC}$ values. For this analysis, leachate profiles were used, and
the magnitude of the K$_{OC}$ value used in the partitioning model also affects the shape of the
leachate concentration profile. As the K$_{OC}$ value increases, the leachate profile (Figure 5-2)
approaches the profile of a long-duration pulse source. Long-duration pulse sources generally
produce higher receptor well concentrations.

        As expected, moving the receptor well closer to the landfill can greatly decrease the
DAF^{GW} value. This can be seen by comparing the DAF of the two-parameter high-end cases with
that of the central tendency case. For the filter aids waste stream, the DAF from the high-end case
was smaller (by a factor of about 2) than the DAF from the central tendency case. For the TAM
sludge waste stream, the DAF from the high-end case was smaller (by a factor of about 30) than
the DAF from the central tendency case.

The dramatic reduction in DAFs (from central tendency to high end) in the TAM sludge case was investigated in a series of additional test runs. This analysis shows that the reduction in DAF generally results from differences in the groundwater pathway parameters between the landfill location for TAM sludge and the central tendency landfill location for spent filter aids. The groundwater velocity was significantly smaller in the scenario modeled for the TAM sludge waste stream than in the scenario modeled for the filter aids waste stream. Thus, the ratio of groundwater velocity to infiltration rate for the TAM sludge (14.7) was smaller than that for the filter aids (_____)*. The smaller ratio for TAM sludge causes the contaminant plume to reach the bottom of the aquifer faster than in the scenario modeled for filter aids. This is more so for the central tendency case than for the high-end distance to receptor well (X-well) case because, as time goes on, the relatively large recharge rate (which is equal to the infiltration rate) tends to drive the vertical plume center farther down to the bottom of the aquifer.

For the TAM sludge location, the aquifer thickness was 22.9 m. The depth of the well intake point (Z-well) was either 5 m or half the aquifer thickness, whichever is smaller; thus, Z-well for TAM sludge was 5 m. By the time the plume reached the central tendency receptor well location (X-well = 430 m and Y-well [distance from plume centerline] = 118 m), the maximum groundwater concentration had occurred near the bottom of the aquifer. That is, the receptor well (Z-well = 5 m) was actually located above most of the plume. For the high-end X-well location (X-well = 102 m and Y-well = 94 m), the well intake point was close to the vertical center of the plume. This explains the unusually high DAF^{GW} for the central tendency case relative to that for the high-end X-well case for the TAM sludge waste stream.

Step 3: The partitioning model is then run for each individual constituent in each waste stream. The initial waste concentration is set at a fixed concentration (initial setting, 1,000 mg/kg) in the partitioning model, and the model is run to generate 9- and 30-year average leachate concentrations for each constituent in each waste stream.

As the third step of the deterministic analysis, the initial waste concentration in the partitioning model was set at 1,000 mg/kg for each of the 53 individual constituents of each waste stream. For spent filter aids, the waste quantity was set to the high-end value and all other inputs in the partitioning model were set at central tendency values. These values are presented in Section 4, Table 4-8. For the TAM sludge analysis, there was only a single waste volume and a single location, so all values in the partitioning model were set at central tendency values. These values are presented in Section 4.0, Table 4-9. The partitioning model was run, and the 9- and 30-year maximum average leachate concentrations were generated for each of the 53 individual constituents for each waste stream.

Step 4: The appropriate surrogate DAF^{GW}s generated in Step 2 are applied to the constituent-specific leachate concentrations generated in Step 3 for each constituent within each of the six surrogate groups. This yields a constituent-specific 9- or 30-year average residential well concentration.

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*Relevant data are not included at present because of business confidentiality concerns.

Residential well concentrations were estimated for each constituent by combining the constituent-specific leachate concentrations with the appropriate DAF^{GW}. Each of the individual constituents in each waste stream was represented by one of the six surrogate compounds in the groundwater model. Those constituents in each surrogate group with noncancer endpoints used the constituent-specific 9-year maximum average leachate concentration from the partitioning model divided by the 9-year minimum DAF^{GW} for the appropriate surrogate. Because one of the high-end parameters identified for constituents with cancer endpoints is exposure duration, these constituents used the constituent-specific 30-year maximum average leachate concentration from the partitioning model divided by the appropriate minimum 30-year DAF^{GW}. These calculations yielded constituent-specific residential well concentrations.

Step 5: The appropriate exposure factors and constituent-specific health benchmark values are used to estimate risk for each exposure route (ingestion of drinking water, inhalation during household use, and dermal exposure during bathing or showering) due to the initial waste concentration fixed in Step 3.

The residential well concentrations estimated in Step 4 were used to estimate risk for each constituent due to ingestion of drinking water, inhalation of tap water during household use, and dermal exposure during bathing or showering. The drinking water ingestion pathway was evaluated using the standard risk equations, the constituent-specific health benchmarks, and the following variable exposure factors: drinking water intake rate, body weight, and exposure duration for adults and children. The household inhalation pathways include the time during showering, the time in the bathroom after showering, and all remaining time in the house not in the bathroom. The shower model used to evaluate inhalation and dermal pathways is described in Section 5.2.2. The exposure factors and constituent-specific benchmarks used in these risk estimates are presented and discussed in Section 6.0 of this document.

The risks determined by Step 5 correspond to the waste concentration set in Step 3 of this analysis. For the initial evaluation, this concentration was 1,000 mg/kg. For noncancer endpoints, the risks for inhalation and oral routes were not considered additive if the target organ for the endpoints was not the same for both routes. For cancer endpoints, inhalation and oral exposures were considered additive. Dermal risk was considered additive to oral risk in all cases because it is based on the same health benchmark, with an appropriate adjustment factor. Dermal exposures are only estimated for adults in this risk assessment because of the lack of readily available, appropriate exposure factors for children. Dermal exposure factors are not available for children and there is a high degree of uncertainty associated with the estimated risk to children for this pathway.

Step 6: The risk estimated in Step 5 is compared to the target risk (for carcinogens, the risk = 1E-05; for noncarcinogens, the HQ = 1), and a new waste or leachate concentration is estimated that will yield the target risk or HQ.

All risk was initially estimated for the fixed concentration of 1,000 mg/kg for each constituent. The resulting risk was compared to the target risk for the most sensitive receptor. The target risk is 1E-05 for constituents with cancer endpoints and the HQ is 1 for constituents with noncancer endpoints. The ratio of the risk estimated in Step 5 to the target risk (for

carcinogens, the risk = 1E-05; for noncarcinogens, the HQ = 1) was then applied to the initial waste concentration to develop a new initial waste concentration that should yield a risk estimate close to the target risk level (see linearity discussion in Section 5.2.1.2.1). Steps 3 through 5 were then repeated iteratively until the initial concentration, which yields the exact target risk level, was determined. This became the allowable waste concentration. This same process was used to determine allowable leachate concentrations.

5.2.1.2.2 *Analysis for Metals*. Deterministic analyses for metals in spent filter aids and TAM sludge wastes were also conducted for this assessment. In the EPACMTP analyses for most of the metals, the constituents never reached the residential well. Therefore, these analyses are not included in this section. The documentation of the deterministic analyses for metals is presented in Appendix I.

5.2.1.3 **Monte Carlo Analysis for the Groundwater Pathway**. Like the deterministic analysis, the purpose of the groundwater pathway Monte Carlo analysis for the filter aids and TAM sludges waste streams is to estimate constituent-specific threshold leachate and waste concentrations from the scenario producing a reasonable maximum exposure concentration. In the probabilistic analysis, all parameters are varied using a Monte Carlo methodology.

The following sections describe the groundwater modeling methodology and results of the probabilistic analysis. Section 5.2.1.3.1 describes the Monte Carlo analysis performed for organic constituents; and Section 5.2.1.3.2 discusses the Monte Carlo analysis performed for metals.

5.2.1.3.1 *Analysis for Organic Constituents*. The groundwater Monte Carlo analysis for organic constituents used the same models as the deterministic analysis. The relationships between the models and the steps in the analysis are unchanged. However, maintaining the relationship between models throughout the probabilistic analysis is more difficult. Care has been taken in this analyses to ensure that all parameters shared between models were varied consistently. The Monte Carlo analysis required many more computations than the deterministic analysis, and the computations were far more complex. Consequently, simplifying modifications were made to the modeling methodology to reduce the computational resources required to conduct the Monte Carlo groundwater modeling. Those modifications are described below, followed by a step-by-step description of the process used to conduct the probabilistic modeling.

Analysis of the Leachate Concentration Curve. The concentration of waste constituents in the leachate emanating from a landfill is expected to vary as a function of a number of parameters, including the waste concentration, annual waste mass, waste management unit size (area), and infiltration rate. The leachate concentration also varies with time. Generally, the leachate concentration increases during the active life of the landfill and then diminishes after the landfill is closed. The shape of the leachate concentration curve versus time (leachate profile) and the magnitude of the maximum leachate concentration, however, are highly constituent-specific.

Because of the prohibitive computational resources required to conduct the Monte Carlo groundwater modeling using the landfill-depleting source option and the entire leaching profile (of annual leach concentrations over a 1,000-year period) for each of the 2,000 iterations, a simpler methodology was developed. The deterministic and sensitivity analyses showed that, for organics,

approximately the same DAF^{GW} results if the landfill modeling is conducted with a pulse source in which the highest 9-year average leachate concentration is used compared to using the entire leaching profile. That is, if the time-averaged receptor well concentration used in the risk calculation was directly related, by a nearly constant factor (DAF^{GW}), to the corresponding time-averaged value of the leachate concentration, it was not necessary to know the entire leachate concentration history, only the maximum 9-year average leachate concentration.

A series of model runs were conducted to verify this observation. For runs in which the pulse source was used, the leaching duration was determined to preserve the contaminant mass balance. This concept is illustrated in Figure 5-2. The graphs in this figure show the actual leachate concentration profile for the six representative constituents (surrogates). Superimposed on these graphs is the equivalent pulse source condition for each constituent. As shown in Figure 5-2, the maximum 9-year average leachate concentration captures the most significant component of the actual leachate concentration profile for each constituent.

The results presented in Tables 5-5 and 5-6 demonstrate that the DAF values obtained using the leaching profile and the equivalent pulse source are essentially the same. Thus, the maximum 9-year average leachate concentration is used in the Monte Carlo groundwater analysis.

Figure 5-2 shows that the effective pulse source duration varied considerably between different constituents. However, Table 5-6 shows that the resulting DAF^{GW} values were not sensitive to this finding; the DAF^{GW} values were essentially the same for every constituent. In other words, even the shortest pulse duration, such as that for pyridine (surrogate 2), was already sufficiently long to drive the DAF^{GW} to its steady-state value. This analysis shows that the EPACMTP model may appropriately be run in finite pulse source mode, with the pulse duration determined from the actual leachate profiles used for the deterministic analysis. This methodology adjusts leaching duration to account for the constituent-specific K_{OC} and results in an approximate conservation of mass.

Step-by-Step Description of the Monte Carlo Analysis. The Monte Carlo simulation procedure essentially involves the same steps used for the deterministic analysis. The models used in the Monte Carlo analysis are identical to those used for the deterministic analysis. They include the partitioning model to estimate leachate concentrations exiting in the landfill, the groundwater transport model (EPACMTP) to estimate the fate and transport of constituents to the residential well, and the exposure models for estimating risk from ingestion and noningestion uses of contaminated groundwater.

However, because distributions of variable parameters are used in the Monte Carlo analysis, integration of the models to maintain consistency throughout the process is more complex. In order to conduct a valid Monte Carlo analysis, the values selected for parameters common to both the landfill partitioning model and the groundwater model must be fully synchronized in each Monte Carlo realization. As in the deterministic analysis, the source

Table 5-5. EPACMTP Modeling Performed Using Leaching Profile

Surrogate	Constituent Name	K_{OC} (L/kg)	λ (yr^{-1})	Maximum 9-year Average C_L[a] (mg/L)	Maximum 9-year Average C_{RW}[b] (mg/L)	DAF^{GW}
1[c]	_____[d]	___[d]	___[d]	___[d]	___[d]	___[d]
2	Pyridine	2.2	0.0	36.3	7.10	5.1
3[c]	_____[d]	___[d]	___[d]	___[d]	___[d]	___[d]
4	Naphthalene	1,096	0.0	1.01	0.194	5.2
5[c]	_____[d]	___[d]	___[d]	___[d]	___[d]	___[d]
6[c]	_____[d]	___[d]	___[d]	___[d]	___[d]	___[d]

[a] Leachate concentration.
[b] Receptor well concentration.
[c] These compounds were used in the original analysis but were later dropped from the list of constituents of concern evaluated for this analysis. They were nevertheless retained as surrogate constituents for the compounds assessed.
[d] Relevant data are not included at present because of business confidentiality concerns.

Table 5-6. EPACMTP Modeling Performed Using Equivalent Pulse Source

Surrogate	Constituent Name	K_{OC} (L/kg)	λ (yr^{-1})	Apparent Leaching Duration	Maximum 9-year Average C_L (mg/L)	Maximum 9-year Average C_R (mg/L)	DAF^{GW}
1[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
2	Pyridine	2.2	0.0	30	36.3	7.08	5.1
3[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
4	Naphthalene	1,096	0.0	681	1.01	0.198	5.1
5[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]
6[a]	_____[b]	___[b]	___[b]	___[b]	___[b]	___[b]	___[b]

[a] These compounds were used in the original analysis but were later dropped from the list of constituents of concern evaluated for this analysis. They were nevertheless retained as surrogate constituents for the compounds assessed.
[b] Relevant data are not included at present because of business confidentiality concerns.

partitioning model and the EPACMTP model are run sequentially, and all parameters common to both models (e.g., landfill area and infiltration rate) are set to the same values in both models to maintain consistency and conserve contaminant mass. The integrated probabilistic analysis is a multistep process. The steps in the analysis are described as follows. These steps parallel those undertaken for the deterministic analysis described previously.

Step 1: The initial waste concentration is set at 1,000 mg/kg in the partitioning model, and the model is run in the Monte Carlo mode for each of the six groundwater surrogate compounds. The model generates 1,000 maximum 9-year average leachate concentrations for each of the six groundwater surrogate compounds.

In the Monte Carlo analysis, the initial waste concentration was set at 1,000 mg/kg in the partitioning model for each of the six groundwater surrogate compounds, and the other variable parameters in the model (landfill area, depth, waste quantity, and location-dependent parameters) were varied randomly. The results of the 1,000 partitioning model runs for each surrogate were 1,000 iterations of the analysis (lines of data), each of which contains the maximum 9-year average leachate concentration and the randomly selected parameters (including location codes) used in the partitioning model that yielded the leachate concentration.

Step 2: The EPACMTP model takes the 1,000 maximum 9-year average leachate concentrations for each of the six surrogate compounds and the shared parameters used by them to generate 2,000 9-year average DAF^{GW}s for each surrogate.

For each realization of the Monte Carlo analysis for each of the six surrogate compounds, the EPACMTP model picked a random record from the file generated by the partitioning model and input the values of the common parameters into that record. The hydrogeologic and climate region codes were then employed to choose random values for other saturated and unsaturated zone parameters appropriate to the landfill location associated with the chosen record.

The receptor well location was treated was a Monte Carlo parameter. The downgradient distance (X-well) to the well was obtained from the nationwide distribution of distances from a municipal landfill to the nearest drinking water well (U.S. EPA, 1997b). This distribution has lower and upper limits of 0.6 m and 1,635.0 m (1 mile), respectively. The location of the well with respect to the centerline of the plume was assigned randomly within the plume boundaries. The depth of the well below the water table (Z-well) was treated as a uniform random distribution with a minimum value of 0.0 m (that is, the water table) and a maximum value of 10 m below the water table or the thickness of the aquifer, whichever was smaller.

The output of the EPACMTP model runs in the Monte Carlo mode was a series of 2,000 iterations of the analysis (lines of data) for each of the six surrogate compounds. Each line of data included the shared parameters used in the partitioning and groundwater models (landfill area, infiltration rate, location code, the maximum 9-year average leachate concentration, and the maximum 9-year average receptor well concentration). The leachate concentration and associated well concentration were used to calculate a distribution of 2,000 DAF^{GW}s, one associated with each line of data inputs. The data set (2,000 lines of data) containing the shared parameters and the associated DAF^{GW}s were output for use in the remaining portion of the analysis.

Step 3: The partitioning model is then run for each individual constituent in each waste stream. The initial waste concentration is set at a fixed concentration (initial setting, 1,000 mg/kg) in the partitioning model, and the model is run to generate 2,000 iterations of 9-year average leachate concentrations for each individual constituent in each waste stream.

The partitioning model was run for each individual constituent in each waste stream, with the waste concentration initially fixed at 1,000 mg/kg. Parameters in the partitioning model not essential to the EPACMTP model (waste f_{oc}) were varied randomly. The other parameters in the partitioning model that are shared with the EPACMTP groundwater model (i.e., landfill area, landfill depth, infiltration, and location) were entered as inputs into the partitioning model from the EPACMTP output file to maintain consistency in the shared parameters between the models. The EPACMTP output file for each surrogate was used as the input file to the partitioning model for each individual constituent represented by that groundwater surrogate. Each line of data from the EPACMTP surrogate-specific output file was used as input data to the partitioning model, and 2,000 constituent-specific 9-year average leachate concentrations were calculated.

Step 4: The appropriate surrogate DAF^{GW} is applied to the constituent-specific leachate concentrations generated in Step 3 for each constituent within each of the six surrogate groups. This yields 2,000 constituent-specific maximum 9-year average residential well concentrations.

Each line of the EPACMTP output file not only contains inputs required by the partitioning model but also a corresponding DAF^{GW}. Thus, each of the 2,000 iterations of constituent-specific 9-year maximum average leachate concentrations estimated in Step 3 was matched with a corresponding DAF^{GW} for the appropriate groundwater surrogate compound. Dividing the 2,000 constituent-specific leachate concentrations calculated in Step 3 by the corresponding 2,000 DAF^{GW} yielded 2,000 maximum 9-year average residential well concentrations.

Step 5: The appropriate distributions of exposure factors and constituent-specific health benchmark values are used to generate 2,000 estimates of risk for each exposure route (ingestion of drinking water, inhalation during household use, and dermal exposure during bathing or showering) due to the initial waste concentration fixed in Step 3.

The residential well concentrations estimated in Step 4 of this analysis were integrated directly with randomly generated exposure parameters for adults and children to yield 2,000 risk estimates for adults and children for the ingestion of drinking water, inhalation of tap water during household use, and dermal exposure during bathing or showering. The risk due to ingestion of drinking water was estimated using a simple equation that includes the variable well concentration and the exposure factors for the adult and child receptors (i.e., drinking water intake, body weight, and exposure duration). The exposure due to inhalation of constituents volatilized during household tap water use was estimated using a model developed specifically for this purpose. The dermal exposure model is discussed in detail in Section 5.2.3.

The exposure variables included in this model were the inhalation rate, body weight, and exposure duration. The only Monte Carlo variables in the risk equations for dermal exposures to tap water were the constituent concentration, body weight, and exposure duration. The body weight and exposure duration variables were considered simultaneously in all exposure routes so that total exposures were consistent across pathways; if it was appropriate, the risk was summed. Adding risk was only appropriate if the health benchmarks for the two routes were based on the same endpoint. Dermal exposure was based on the oral benchmark so that risk due to dermal exposure could be added to risk due to oral exposure in all cases. For cancer endpoints, risks also were considered additive across routes. For constituents with noncancer endpoints, the target organ for the health benchmark had to be considered; only if the organ is the same can the HQs be added for different exposure routes. A complete discussion of exposure factors used in this risk analysis is presented in Section 6.0.

Step 6: The 90th and 95th percentiles of the 2,000 risk estimates generated in Step 5 are compared to the target risk (for carcinogens, risk = 1E-05 and for noncarcinogens, HQ = 1). A new waste concentration is estimated that will yield the target risk or HQ at the 90th and 95th percentiles. These concentrations are set as the fixed waste concentrations in Step 3, and Steps 3 through 6 are repeated until the estimated risk is the target risk for all constituents. The same process is used to generate 90th and 95th percentile leachate concentrations.

The 90th percentile of the 2,000 risk estimates with an initial waste concentration of 1,000 mg/kg was determined. The 90th percentile risk was compared to the target risk for the most sensitive receptor. The target risk is 1E-05 for constituents with cancer endpoints, and the HQ is 1 for constituents with noncancer endpoints. The ratio of the target risk and the initial risk estimate was used with the initial waste concentration of 1,000 mg/kg to estimate a new starting waste concentration. The new waste concentration was entered into the partitioning model in Step 3, and Steps 3 through 6 were repeated until the estimated risk was equal to the target risk level for each constituent in each waste stream. The final waste concentrations yielding the target risk were the risk-limiting concentrations. The waste concentration corresponding to the target risk at the 95th percentile was determined in the same manner. The constituent concentrations corresponding to the target risk were the risk-based listing concentrations derived based on Monte Carlo analysis.

5.2.2   Contaminant Fate and Transport Modeling

The following contaminant fate and transport models were used to conduct the deterministic and probabilistic analyses for this assessment:

- #   Landfill partitioning model to estimate leachate concentrations exiting the landfill

- #   Groundwater transport model (EPACMTP) to estimate fate and transport of constituents to the residential well

- #   Exposure models to estimate risk from ingestion and noningestion uses of contaminated groundwater.

These models are described in detail below.

5.2.2.1 <u>**Landfill Partitioning Model**</u>. A spreadsheet calculation model was used to determine contaminant loss from a landfill due to volatilization and leaching. The model uses partitioning equations developed for estimating volatilization of contaminants from soil (Jury et al., 1983, 1984, 1990). For this risk assessment, the equations were adapted to represent the management practices and design criteria required by regulation for municipal landfills. Runoff losses were assumed to be zero because landfills are assumed to have berms or other control devices sufficient to prevent runoff. Aerobic degradation was assumed not to occur in landfills.

Key assumptions for the partitioning model are that waste will be collected 350 d/yr and each daily addition volume will be placed in a daily pile in the landfill. The model evaluated contaminant losses from the landfill over three separate conditions: (1) losses from the daily pile, which is uncovered for a portion of the day; (2) losses from the daily pile after cover is applied; and (3) losses after closure of the landfill cell when the waste is covered by a 2-ft thick landfill cap.

Daily piles were assumed to be uncovered for a period of 12 h prior to the application of daily cover. After the 12-h period, the waste was assumed to be covered by a 6-in daily cap as required by the municipal landfill regulations (40 CFR 258). The amount of contaminant lost during the uncovered duration was calculated, and the total contaminant concentration remaining in the waste was calculated and used as the starting concentration for the covered daily waste addition.

On each successive day, the daily waste addition was placed in piles assumed to be adjacent to the previous day's waste addition. For each daily addition of waste, 12-h emissions were estimated from the newly added uncovered pile, and vapor emissions were estimated through the daily soil cover for waste added on previous days. The model estimated partitioning of the waste through the daily cover until new waste was added to that daily pile.

Additions of new daily piles continued until the area of the cell was filled with a layer of daily waste. When a layer was completed, a second layer was begun by placing the next daily waste addition on top of the initial daily wastepile. Once waste was added on top of a daily addition, volatilization losses from that pile were assumed to be minimal. This process continued until the annual cell was filled. The number of daily piles in a cell was estimated based on the area of the landfill cell divided by the area of each daily pile.

At the end of each year, the annual cell was assumed to be capped and a new cell started. The closed landfill cell was assumed to be covered by a 2-ft thick landfill cap as required under the municipal landfill regulations. The emission rate was greatly reduced by the cap; however, it was not zero.

In the first year, a mass of the constituent was added to the landfill, and the model estimated the mass partitioned to each media (air, leachate, and soil) for each of the emission scenarios. As noted previously, no biodegradation was included within the landfill. The losses to the air and leachate were summed over these three conditions for the year and subtracted from the initial mass of the constituent added to the landfill. The mass of the constituent remaining after accounting for these losses was carried forward and summed with the new annual waste mass added to the landfill. (Note: A check mechanism was included in the model to ensure that the solubility limit of the constituent was not exceeded in the leachate.)

This process continued for the life of the landfill, which is assumed to be 30 years. At the end of the active life of the landfill, all cells were capped, but leaching and limited air emissions were assumed to continue and were modeled for an additional 40 years after landfill closure. Potential release mechanisms for the municipal landfill are graphically presented in Figure 5-3.
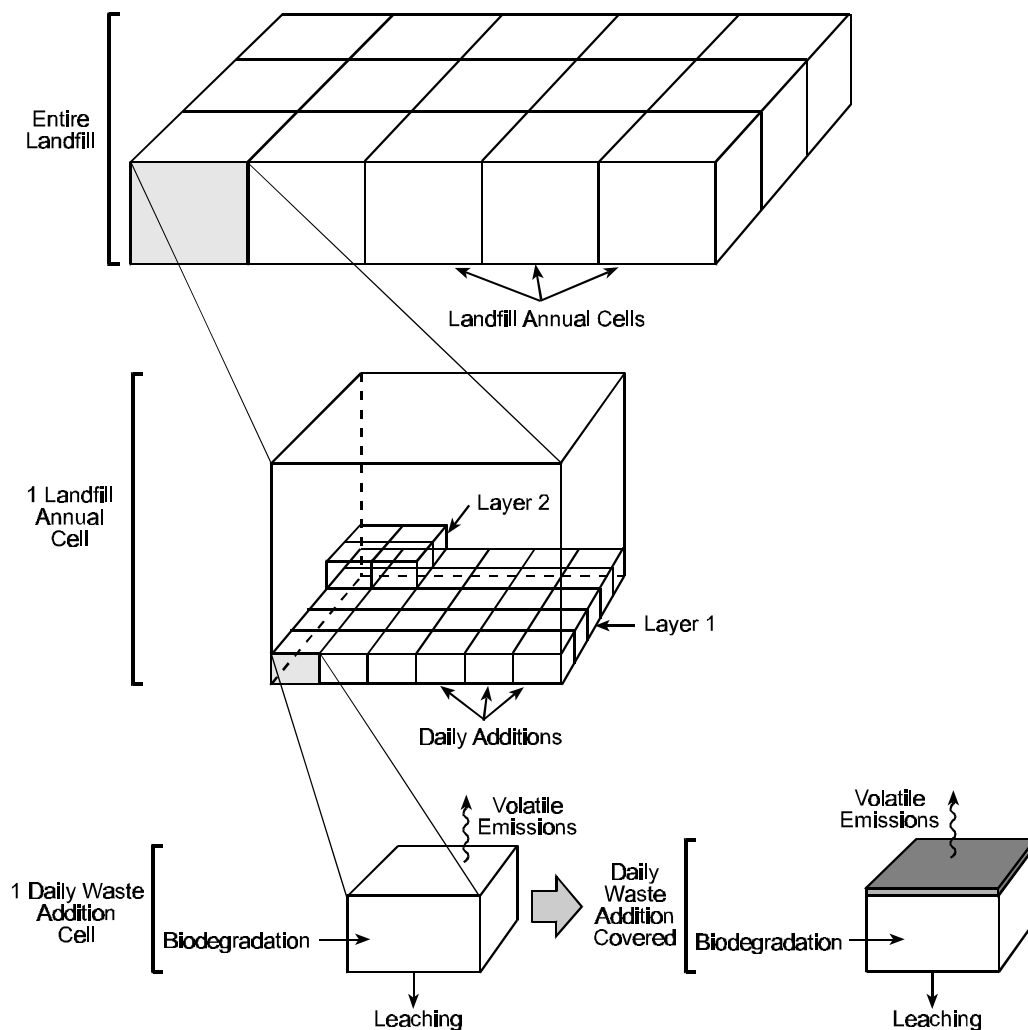

Figure 5-3. Landfill partitioning schematic.

The partitioning model uses a similar set of equations to estimate losses under the three sets of conditions that occurred in the landfill and sums the results. Partitioning was estimated for each constituent based on these physical and chemical properties: Henry's law constant, diffusivity coefficients in air and water, and the soil partitioning coefficient (K_d) (for organic compounds $K_d = f_{OC} \times K_{OC}$). The model assumed linear partitioning and first-order rate losses and used a finite difference (numerical) integration approach to solve the mass balance equations.

The model summed the air emissions and leachate concentrations from each segment of the model (uncovered daily addition, covered daily addition piles, and capped cells) into annual air emission rates and leachate concentrations over the total landfill area and maintained a continuous mass balance within the landfill system. The model also calculated maximum average leachate and air emission rates for 9 and 30 years.

The leachate was generated over the entire area of the landfill each year. The mass of the constituent leached from the landfill annually was dissolved in the total annual volume of the leachate estimated for the entire landfill area. Thus, as the landfill was filling up, the leachate concentrations were increasing. The leachate concentration reached a peak for all constituents after the last waste addition. For very volatile or soluble constituents, the decline was rapid. For constituents that are significantly sorbed within the landfill, the leaching occurred slowly at low concentrations for a long period of time. This model maintained a mass balance within the landfill over time.

To maintain consistency and mass balance in the groundwater risk assessment for the dye and pigment analysis, the annual leachates generated by this model were used as input concentrations in the deterministic groundwater model and 9-year average leachate values were used as inputs for the probabilistic groundwater model. Leachate concentrations were estimated for a total of 1,000 years. At the end of 70 years (30 years of active life and 40 years postclosure), however, the estimated emissions and leachate concentrations for most constituents were reduced to near zero.

5.2.2.1.1 *Partitioning Model Theory*

Equilibrium Partitioning. The total concentration of the contaminant in the soil can be expressed as the sum of the masses of the contaminant adsorbed on the soil or waste particles, dissolved in the liquid, and in the air spaces divided by the total mass of the contaminated soil, as follows:

$$C_T = C_s + {}_w C_w / {}_b + {}_a C_a / {}_b \tag{5-1}$$

where

C_T = total contaminant concentration in landfill (mg/kg = g/Mg)
C_s = concentration of contaminant adsorbed on soil (mg/kg = g/Mg)
${}_w$ = water-filled soil porosity (m^3_{water}/m^3_{soil})
C_w = concentration of contaminant in liquid ($\mu g/cm^3 = g/m^3$)
${}_b$ = soil dry bulk density ($g/cm^3 = Mg/m^3$)

$\quad_a$ = air-filled soil porosity (m^3_{air}/m^3_{soil})

C_a = concentration of contaminant in air $(\mu g/cm^3 = g/m^3)$.

The adsorbed contaminant concentration is assumed to be linearly related to the liquid phase concentration, as follows:

$$C_s = K_d C_w \tag{5-2}$$

where

K_d = soil-water partition coefficient $(cm^3/g = m^3/Mg) = K_{oc} f_{oc}$ for organic compounds

K_{oc} = soil organic carbon partition coefficient (cm^3/g)

f_{oc} = organic carbon content of soil (g/g).

The contaminant concentration in the vapor phase is assumed to be linearly related to the liquid phase concentration, as follows:

$$C_a = H' C_w \tag{5-3}$$

where

H' = dimensionless Henry's law constant = $41 \times H$

H = Henry's law constant at 25 °C $(atm\text{-}m^3/mol)$.

Equations 5-2 and 5-3 assume linear equilibrium partitioning between the adsorbed contaminant, the dissolved contaminant, and the volatilized contaminant. Combining Equations 5-1, 5-2, and 5-3 yields

$$C_T = C_s [1 + \ _w/(K_d \ _b) + \ _a H'/(K_d \ _b)] \ . \tag{5-4}$$

The total contaminant concentration, C_T, represents the measured soil concentration. The adsorbed soil concentration, however, is needed to calculate the equilibrium liquid and air contaminant concentrations (Equations 5-2 and 5-3). Equation 5-4 can be rearranged to calculate the adsorbed soil contaminant concentration given the total contaminant concentration as follows:

$$C_s = \frac{C_T \ K_d \ _b}{(K_d \ _b + \ _w + \ _a H')} \ . \tag{5-5}$$

Overall Mass Balance and Contaminant Half-Life. For a constant volume system, assuming first-order rate loss mechanisms, the mass balance can be expressed as

$$(\ C_T /\ \ t) = - (k_{app,air} + k_{app,leach} + k_{app,runoff} + k_{app,bio} + k_{app,hyd}) \ C_T \tag{5-6}$$

where

t	=	time, s.
$k_{app,air}$	=	apparent first-order rate constant for volatilization (s^{-1})
$k_{app,leach}$	=	apparent first-order rate constant for leaching (s^{-1})
$k_{app,runoff}$	=	apparent first-order rate constant for rain runoff (s^{-1})
$k_{app,bio}$	=	apparent first-order rate constant for biodegradation (s^{-1})
$k_{app,hyd}$	=	apparent first-order rate constant for hydrolysis (s^{-1})

For small time steps (time steps in which C_T changes by only a few percentage points), Equation 5-6 can be approximated as follows:

$$(M_{s,t+\Delta t} - M_{s,t})/(\Delta t) = - (k_{app,air} + k_{app,leach} + k_{app,runoff} + k_{app,bio} + k_{app,hyd}) \, M_{s,t} \qquad (5\text{-}7)$$

or

$$\Delta M_{tot} = \Delta M_{air} + \Delta M_{leach} + \Delta M_{runoff} + \Delta M_{bio} + \Delta M_{hyd} \qquad (5\text{-}8)$$

where

$M_{s,t+\Delta t}$	=	mass of contaminant in soil at time $t+\Delta t$ (g)
$M_{s,t}$	=	mass of contaminant in soil at time t (g)
Δt	=	time step of calculation (s)
ΔM_{tot}	=	total mass of contaminant removed from soil over time step (g = $M_{s,t} - M_{s,t+\Delta t}$)
ΔM_{air}	=	mass of contaminant lost over time step due to volatilization (g)
ΔM_{leach}	=	mass of contaminant lost over time step due to leaching (g)
ΔM_{runoff}	=	mass of contaminant lost over time step due to runoff (g)
ΔM_{bio}	=	mass of contaminant lost over time step due to biodegradation (g)
ΔM_{hyd}	=	mass of contaminant lost over time step due to hydrolysis (g).

Due to the simplified nature of the numerical integration used, any number of competing loss mechanisms can be included in the model, because each of the loss mechanisms can be evaluated separately and then summed together. The overall apparent first-order disappearance rate is simply the sum of all of the individual first-order rate constants as follows:

$$k_{app,overall} = k_{app,air} + k_{app,leach} + k_{app,runoff} + k_{app,bio} + k_{app,hyd} \quad . \qquad (5\text{-}9)$$

Unfortunately, there are few controlled biodegradation rate studies that actually measure biodegradation rates while accounting for competing removal mechanisms. Typically the literature reports a contaminant disappearance rate (or half-life) in soil, which includes losses via volatilization, leaching, and hydrolysis, as well as biodegradation. That is, data for $k_{app,overall}$ are generally more prevalent than data specifically for $k_{app,bio}$. As a result, the model first calculates $k_{app,overall}$ from the soil half-life and then integrates Equation 5-6 to calculate the total mass lost from the system, as follows:

$$\Delta M_{tot} = M_{s,t} \, [1 - \exp(- k_{app,overall} \, \Delta t)] \quad . \qquad (5\text{-}10)$$

The mass lost by each loss mechanism is then calculated, and $\Delta M_{bio} + \Delta M_{hyd}$ is calculated by the difference. If ΔM_{tot} as calculated by Equation 5-10 is less than $\Delta M_{air} + \Delta M_{leach} + \Delta M_{runoff}$, then Equation 5-8 is used to calculate $\Delta M_{bio} + \Delta M_{hyd}$ by the difference.

Mass Loss Through Volatilization. The primary mechanism of contaminant loss to the atmosphere is the diffusion of volatilized contaminant to the soil surface. During periods of evaporation, the flux of water vapor enhances contaminant transport to the soil surface. Consequently, the total contaminant flux to the atmosphere is

$$J_{air,t} = J_{vol,t} + J_{evaptr,t} \tag{5-11}$$

where

$J_{vol,t}$ =   contaminant flux to the atmosphere due to diffusion, g/m^2-s
$J_{evaptr,t}$ =   contaminant flux to the atmosphere due to evaporative transport, g/m^2-s.

Because both leaching and enhanced volatilization due to evaporation losses were being modeled, the Jury et al. (1983) model was not used directly because, to be consistent with the model derivation, this solution should calculate V_E (evaporative convective velocity) based on the overall water flux, which would be a V_E term based on the infiltration rate. Instead, the losses were segregated, diffusional volatile losses were estimated for $V_E = 0$, leaching losses were estimated assuming an equilibrium concentration in the infiltrate, and the increased volatilization losses due to periods of evaporation were estimated using terms from the Jury model solution that were primarily attributable to convective contaminant flux. This solution is an estimation methodology, but, for small time steps, it is accurate compared to the more complete set of the Jury et al. (1983) model equations.

Emissions with No Soil Cover. Assuming that there is no soil cover, no stagnant boundary air layer at the ground surface, and no net water flux ($V_E = 0$), the Jury et al. (1990) simplified finite source model for diffusional volatilization can be written as

$$J_{vol,t} = C_T \left(\frac{0.01\ D_A}{t} \right)^{\frac{1}{2}} \left[1 - \exp\left(\frac{-d_s^2}{0.04\ D_A\ t} \right) \right] \tag{5-12}$$

where

D_A =   apparent diffusivity (cm^2/s)
          =   3.14
t =   time (s)
d_s =   depth of uniform soil contamination at $t = 0$ (i.e., depth of daily addition [m]);

and

$$D_A = \left[\frac{\left(\theta_a^{\frac{10}{3}} D_i H' + \theta_w^{\frac{10}{3}} D_w \right)}{n^2 \left(\rho_b K_d + \theta_w + \theta_a H' \right)} \right] \tag{5-13}$$

where

D_i =   diffusivity in air (cm^2/s)
D_w =   diffusivity in water (cm^2/s)
n =   total soil porosity $(L_{pore}/L_{soil}) = 1 - (\rho_b/\rho_s)$
ρ_s =   soil particle density (g/cm^3).

As discussed in Jury et al. (1984), volatilization with evaporation is a complex problem, but evaporation always increases the overall volatilization rate. Jury et al. (1984) presents an equation for the convection of contaminants caused by the flux of water in the soil. The convective volatilization flux caused by evaporation is then calculated by isolating the first half of the overall volatilization flux equation (Jury et al., 1983), which is the primary term for convective transport. The other terms are nearly identical to Equation 5-12, and approach Equation 5-12 in the limits of small time or small convective velocity. The evaporative flux can, therefore, be estimated as follows:

$$J_{evaptr,t} = \tfrac{1}{2} C_T \rho_b (0.01 V_E) \left[erfc\left(\frac{V_E t}{(4 D_A t)^{\frac{1}{2}}} \right) - erfc\left(\frac{(100 d_s + V_E t)}{(4 D_A t)^{\frac{1}{2}}} \right) \right] \tag{5-14}$$

where

V_E =   evaporative convective velocity (cm/s)
$erfc(x)$ =   complementary error function

and

$$V_E = \left(\frac{-E}{(365 \times 24 \times 3600) \times (\rho_b K_d + \theta_w + \theta_a H')} \right) \tag{5-15}$$

where

E =   average annual evaporation rate (cm/yr).

(Note: The minus sign is introduced here because upward movement is in the negative direction.)

The total mass loss to the air can be calculated as follows:

$$\Delta M_{air} = (J_{evaptr,t} + J_{vol,t})(A\Delta t) \tag{5-16}$$

where

A = area of contaminant source, m^2.

Emissions with Soil Cover. Jury et al. (1990) also provided simplified equations for the volatilization flux for a contaminated soil (waste) layer buried below a clean layer of soil. This method basically evaluates the contaminant flux for the depth of the system (contaminated layer + cover), and then subtracts the flux attributable to the top layer (i.e., the flux, assuming the contaminant is present only in the cap). The volatilization contaminant flux for buried waste can then be estimated as

$$J_{vol,t} = C_T \left(\frac{0.01\ D_A}{t} \right)^{\frac{1}{2}} \left[\exp\left(\frac{-d_{cover}^2}{0.04\ D_A\ t} \right) - \exp\left(\frac{-(d_{cover} + d_s)^2}{0.04\ D_A\ t} \right) \right] \tag{5-17}$$

where

d$_{cover}$ = depth of daily soil cover or landfill cap (m).

Using the same rationale, the evaporative contaminant flux is equal to the contaminant flux, assuming the entire system contains the contaminant and subtracting the emissions attributable to the top soil layer. The evaporative contaminant flux can then be estimated as:

$$J_{evaptr,t} = \frac{1}{2}C_T \ \ _b(0.01V_E)\left[\text{erfc}\left(\frac{100d_{cover} + V_E t}{(4\ D_A\ t)^{\frac{1}{2}}} \right) - \text{erfc}\left(\frac{100(d_{cover} + d_s) + V_E t}{(4\ D_A\ t)^{\frac{1}{2}}} \right) \right] \tag{5-18}$$

Mass Loss Through Leaching. The mass flux loss of a contaminant due to leaching is estimated by assuming the leachate is in equilibrium with the soil (i.e., Equation 5-2 applies).

$$J_{leach,t} = \frac{C_T \ \ _b\ (0.01\ V_L)}{(\ _b\ K_d + \ _w + \ _a\ H')} \tag{5-19}$$

where

$J_{leach,t}$ = contaminant flux in leachate at time t, g/m^2-s
V_L = (P + I - R - E)/(365 × 24 × 3600) = leachate rate (cm/s)
P = annual average precipitation rate (cm/yr)

       I   =   annual average irrigation rate (cm/yr)

       R   =   annual average runoff rate (cm/yr).

The leaching flux rate can be converted to a mass loss, as follows:

$$\Delta M_{leach} = (J_{leach,t})(A \Delta t) \quad . \tag{5-20}$$

        5.2.2.1.2 *Model Application for Landfill*. The waste added to the landfill is assumed to be homogeneous and temporally consistent. One landfill cell is assumed to be filled per year. The user inputs the annual waste quantity and the contaminant concentration of the waste of interest, the waste density, the dimensions of the entire landfill, and the landfill's life expectancy (i.e., number of cells in the landfill). From this information, the dilution effect of the waste added to the landfill can be calculated. For example, the average contaminant concentration of a daily waste addition is calculated as follows:

$$C_T = \frac{\left(\dfrac{C_{waste} \times Q_{waste}}{N_{daily}} \right)}{_b \times A_{daily} \times d_{daily}} \tag{5-21}$$

where

      C_{waste} =   concentration of contaminant in waste (mg/kg = g/Mg)

      Q_{waste} =   annual waste disposal rate (Mg/yr)

      N_{daily} =   number of daily additions per year

      A_{daily} =   area of a daily waste addition, m^2

      d_{daily} =   depth of a daily waste addition, m.

        5.2.2.2 **Fate and Transport Modeling using EPACMTP**. EPACMTP (U.S. EPA, 1996a, 1996b, 1996c, 1997b) is a computer simulation model for modeling the subsurface fate and transport of contaminants leaching from a land disposal site (e.g., landfill, surface impoundment, wastepile, or land application unit). Fate and transport processes accounted for in the model are: advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and chemical and biological decay processes. The composite model consists of two coupled modules: (1) a one-dimensional module that simulates infiltration and dissolved contaminant transport through the unsaturated zone, and (2) a saturated zone flow and transport module that can be run in either 3-D or quasi 3-D mode.

        EPACMTP has been published in an international refereed journal (Kool et al., 1994) and has been reviewed by EPA's Science Advisory Board (U.S. EPA, 1996). This review commends the Agency for its significant improvements to the model and states that EPACMTP represents the state of the art for such analyses.

        The methods used in EPACMTP to analyze flow and transport in the unsaturated and saturated zones are described below. Figure 5-4 is a schematic diagram of the model.

5.2.2.2.1 *Flow in the Unsaturated Zone (Vadose Zone)*. Flow in the unsaturated zone is assumed to be steady-state, one-dimensional vertical flow from the bottom of the landfill toward the water table. The lower boundary of the unsaturated zone is assumed to be the water table. The flow in the unsaturated zone is predominantly gravity-driven. Therefore, it is reasonable to model flow in the unsaturated zone as one-dimensional in the vertical direction. It is also assumed that transverse dispersion (both mechanical dispersion and molecular diffusion) is negligibly small compared with the width of the landfill. In addition, this assumption is conservative because it allows the leading front of chemicals to arrive at the water table more quickly and, in the case of finite source, with greater peak concentration. The flow rate is assumed to be determined by the long-term average infiltration rate through the landfill.

5.2.2.2.2 *Transport in the Unsaturated Zone*. Contaminant transport in the unsaturated zone is assumed to occur by advection and dispersion. The unsaturated zone is assumed to be initially contaminant-free, and contaminants are assumed to migrate vertically downward from the landfill. EPACMTP can simulate both steady-state and transient transport in the unsaturated zone.

5.2.2.2.3 *Flow in the Saturated Zone (Aquifer)*. The saturated zone module of EPACMTP is designed to simulate flow in an unconfined aquifer with constant thickness. The model assumes regional flow in a horizontal direction with vertical disturbance resulting from recharge and infiltration from the overlying unsaturated zone and landfill, respectively. The lower boundary of the aquifer is assumed to be impermeable. Flow in the saturated zone is assumed to be steady-state. EPACMTP accounts for different recharge rates beneath and outside the landfill area. Groundwater mounding beneath the landfill is represented in the flow system by increased head values at the top of the aquifer. This approach is reasonable as long as the height of the mound is small relative to the thickness of the saturated zone.

5.2.2.2.4 *Transport in the Saturated Zone*. Contaminant transport in the saturated zone is assumed to be a result of advection and dispersion. The aquifer is assumed to be initially contaminant-free, and contaminants are assumed to enter the aquifer only from the unsaturated zone immediately underneath the landfill, which is modeled as a rectangular, horizontal plane
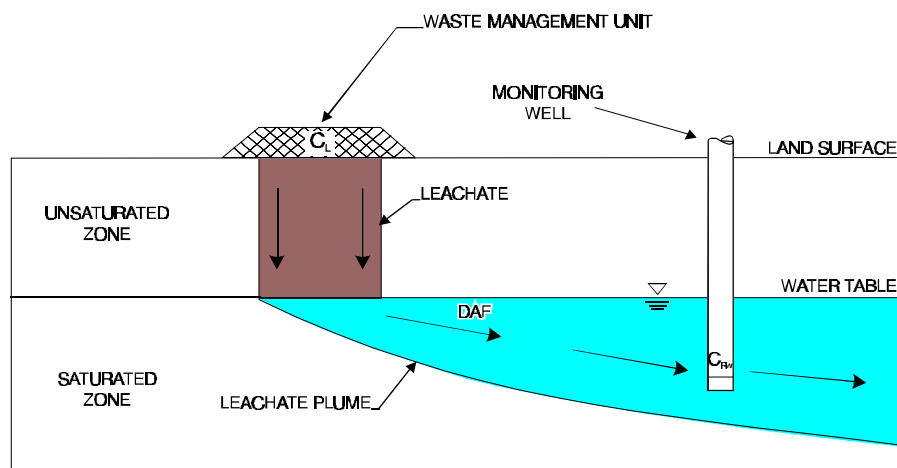

Figure 5-4. Schematic diagram of EPACMTP model.

source. EPACMTP can simulate both steady-state and transient three-dimensional transport in the aquifer. For steady-state transport, the contaminant mass flux entering at the water table is constant with time; for the transient case, the flux at the water table is constant or varies as a function of time.

EPACMTP also accounts for chemical and biological transformation processes. All transformation reactions are represented by first-order decay processes. These transformation processes can be lumped together and specified as an overall decay rate or specified with separate first-order decay coefficients for chemical decay and biodegradation. EPACMTP also has the capability to determine the overall decay rate from chemical-specific hydrolysis constants using soil and aquifer temperature and pH values. In the event that the daughter products of transformation are hazardous and their chemical-specific parameters are known, the model can also account for the formation and subsequent fate and transport of these daughter products.

The groundwater pathway analysis accounts for equilibrium sorption of waste constituents onto the soil and aquifer solid phase. For organic constituents, a partition coefficient (k_d) is calculated as the product of the constituent-specific organic carbon partition coefficient (k_{oc}), and the fraction organic carbon (f_{oc}) in the soil and aquifer.

The output of EPACMTP is a prediction of the contaminant concentration arriving at a downgradient groundwater receptor well. This can be either a steady-state concentration value, corresponding to the continuous source scenario, or a time-dependent concentration, corresponding to the finite source scenario. In the latter case, the model can calculate either the peak concentration arriving at the well or a time-averaged concentration corresponding to a specified exposure duration (e.g., a 9-year average residence time).

EPACMTP has the capability to perform Monte Carlo simulations to account for parametric uncertainty or variability. The flow and transport simulation modules of EPACMTP are linked to a Monte Carlo driver, which permits a probabilistic evaluation of uncertainty in model input parameters as described by specified (joint) probability distributions.

5.2.2.3 <u>Model for Dermal and Inhalation Exposure to Contaminated Tap Water</u>. Exposure to contaminants in groundwater through noningestion pathways (i.e., showering and bathing) were modeled using a dermal and inhalation exposure model (U.S. EPA, 1997d; SAIC, 1999). For the noningestion pathway modeling, the maximum modeled groundwater concentration for each constituent of concern was used as the starting concentration. Both dermal and inhalation risks were estimated based on this starting concentration.

Exposures were estimated for all indoor household water uses. Exposure was assessed for three house compartments: (1) exposure from being in the shower stall during and immediately after showering (both dermal and inhalation exposures), (2) exposure from being in the bathroom after showering (inhalation exposure), and (3) exposure from being in the rest of the house (inhalation exposure). It was assumed that all water uses for each household compartment occurred while the individual was in that compartment. Average air concentrations for each house compartment were calculated and used to assess exposure from inhalation. Dermal exposures

were calculated based on the starting groundwater concentration. Figure 5-5 depicts this exposure scenario.

The model used in this analysis is based on the equations presented in McKone (1987). The model estimates the change in the shower air concentration based on the mass of constituent lost by the water (fraction emitted or emission rate) and the air exchange rate between the various model compartments (shower, the rest of the bathroom, and the rest of the house) following the same basic model construct described by Little (1992). The resulting differential equations are solved using finite difference numerical integration.

The basis for estimating the concentration of constituents in the indoor air is the mass transfer of constituent from water to shower air.

This equation estimates the overall mass transfer coefficient from tap water to air from showering:

$$ K_{ol} = \quad \times \left(\frac{2.5}{D_w^{2/3}} + \frac{1}{D_a^{2/3}H} \right)^{-1} \tag{5-22} $$
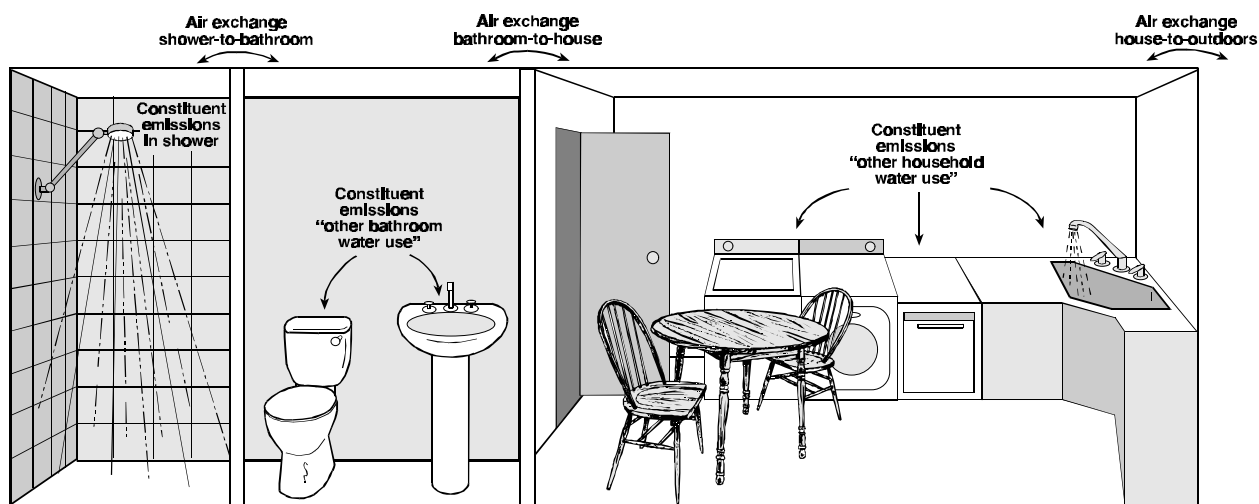
Figure 5-5. Dermal and inhalation exposure scenario.

where

K_{ol}	=	overall mass transfer coefficient (cm/s)
	=	proportionality constant $(cm/s)^{-1/3}$
D_w	=	diffusion coefficient in water (cm^2/s)
D_a	=	diffusion coefficient in air (cm^2/s)
H'	=	dimensionless Henry's law constant $(=41*H_{LC})$.

       The constituent emission rate is estimated from the change in the shower water concentration as the water falls, which is calculated using the overall mass transfer coefficient as follows:

$$\partial c / \partial t = - K_{ol}(A/V)(c - y_s/H)$$
(5-23)

where

c	=	liquid phase (droplet) constituent concentration $(\mu g/cm^3$ or mg/L)
t	=	time (s)
A	=	total surface area for mass transfer (cm^2)
V	=	total volume of water within the shower compartment (cm^3)
y_s	=	gas phase constituent concentration in the shower $(\mu g/cm^3$ or mg/L)
H	=	dimensionless Henry's law constant.

       Consequently, in addition to the overall mass transfer coefficient, the emission rate of a contaminant within the shower is dependent on the surface-area-to-volume ratio of the shower water (within the shower) and the concentration driving force between the water and the shower air.

       The shower emissions can be modeled based on falling droplets as a means of estimating the surface-area-to-volume ratio for mass transfer and the residence time of the water in the shower compartment. Equation 5-23 can then be integrated assuming the compound concentration in the gas phase is constant over the time frame of the droplet fall. The time required for a droplet to fall equals the nozzle height divided by the water droplet velocity. The ratio of the surface area to volume for the droplet is calculated as $6/d_p$ (i.e., by assuming a spherical shape). By assuming the drops fall at terminal velocity, the surface-area-to-volume ratio and the residence time can be determined based solely on droplet size. A droplet size of approximately 1 mm (0.1 cm) was selected. The terminal velocity for the selected droplet size is approximately 400 cm/s. The fraction of constituent emitted from a water droplet at any given time can then be calculated by integrating Equation 5-23 and rearranging as follows:

$$f_{em} = 1 - C_{out}/c_{in} = (1 - f_{sat})(1 - e^{-N})$$
(5-24)

where

f_{em} = fraction of constituent emitted from the droplet (dimensionless)
c_{out} = droplet constituent concentration at shower floor/drain (mg/L)
c_{in} = droplet constituent concentration entering the shower (mg/L)
f_{sat} = $y_s/(H\ c_{in})$ = fraction of gas phase saturation (dimensionless)
N = dimensionless overall mass transfer coefficient = $K_{ol}(6/d_p)(h/v_t)$
d_p = droplet diameter = 0.1 (cm)
h = nozzle height (cm)
v_t = terminal velocity of droplet = 400 (cm/s).

The gas phase constituent concentration in the shower is then calculated for each time step for the duration of the shower. The air exchange rate between the shower and the bathroom is included in the estimation of the gas phase concentration of the constituents in the shower:

$$y_{s,t+1} = y_{s,t} + [Q_{gs} \times (y_{b,t} - y_{s,t}) \times (t_{t+1} - t_t) + E_{s,t}]/V_s \tag{5-25}$$

where

$y_{s,t+1}$ = gas phase constituent concentration in the shower at the end of time step (mg/L)

$y_{s,t}$ = gas phase constituent concentration in the shower at the beginning of time step (mg/L)

Q_{gs} = volumetric gas exchange rate between shower and bathroom (L/min)

$y_{b,t}$ = gas phase constituent concentration in the bathroom at the beginning of time step (mg/L)

$(t_{t=1}-t_t)$ = calculation time step

$E_{s,t}$ = mass of constituent emitted from shower between time t and time t+1 (mg)

V_s = volume of shower stall (L).

The shower model also provides direct estimates of the bathroom and whole house exposure. The risk from inhalation exposures in the remainder of the house is generally several orders of magnitude less than the risk from inhalation exposures in the bathroom and during showering. The gas phase constituent concentration in the bathroom may be estimated by Equation 5-26 for each time step of the exposure duration:

$$y_{b,t+1} = y_{b,t} + \{[Q_{gs} \times (y_{s,t} - y_{b,t})] - [Q_{gb} \times (y_{b,t} - y_{h,t})] + (I_b \times C_{in} \times f_{em,b})\} \times \left(\frac{t_{t+1} - t_t}{V_b}\right) \tag{5-26}$$

where

$y_{b,t+1}$ =   gas phase constituent concentration in the bathroom at end of time step (mg/L)

$y_{b,t}$ =   gas phase constituent concentration in the bathroom at beginning of time step

Q_{gs} =   volumetric gas exchange rate between bathroom and house

$y_{h,t}$ =   gas phase constituent concentration in the house at beginning of time step (mg/L)

I_b =   bathroom water use (L/min)

C_{in} =   constituent concentration in tap water (mg/L)

$f_{em,b}$ =   fraction of constituent emitted from bathroom water use (unitless)

$(t_{t+1} - t_t)$ =   calculation time step (min)

V_b =   volume of bathroom (L).

The gas phase constituent concentration in the remainder of the house may be estimated by Equation 5-27 for each time step of the exposure duration:

$$y_{h,t+1} = y_{h,t} + \{[Q_{gb} \times (y_{b,t} - y_{h,t})] - [Q_{gh} \times (y_{h,t} - y_{a,t})] + (I_h \times C_{in} \times f_{em,h})\} \times \left(\frac{t_{t+1} - t_t}{V_h} \right) \qquad (5\text{-}27)$$

where

$y_{h,t+1}$ =   gas phase constituent concentration in the house at end of time step (mg/L)

$y_{h,t}$ =   gas phase constituent concentration in the house at beginning of time step (mg/L)

Q_{gb} =   volumetric gas exchange rate between the bathroom and house (L/min)

$y_{b,t}$ =   gas phase constituent concentration in the bathroom at beginning of time step (mg/L)

Q_{gh} =   volumetric gas exchange rate between the house and atmosphere (L/min)

$y_{a,t}$ =   gas phase constituent concentration in the atmosphere (mg/L)

I_h =   house water use other than bathroom (L/min)

C_{in} =   constituent concentration in tap water (mg/L)

$f_{em,h}$ =   fraction of constituent emitted from household water use other than bathroom (unitless)

$(t_{t+1} - t_t)$ =   calculation time step (min)

V_h =   volume of house (L).

The average air concentration in the shower and bathroom are obtained by averaging the concentrations obtained for each time step over the duration of the shower and bathroom use. These concentrations and the durations of daily exposure are used to estimate risk from inhalation exposures to residential use of groundwater.

5.2.2.3.1 *Inhalation Exposure to Tap Water*. Where available, the exposure parameters used in this analysis are values cited in the *Exposure Factors Handbook* (U.S. EPA, 1997c). The remaining exposure factors required for this analysis were obtained from McKone (1987). The original articles have been obtained to verify the values used in the analysis. Exposure parameter values used for the dye and pigment analysis are provided and discussed in Section 6 of this document.

5.2.2.3.2 *Dermal Exposure to Tap Water*. Dermal exposure to tap water was also assessed for this analysis. The methodology used for assessing risk from this exposure route follows the guidelines set forth in *Dermal Exposure Assessment: Principles and Applications* (U.S. EPA, 1992) and *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual* (RAGS) (U.S. EPA, 1989).

The basic equation used to calculate the contaminant dose from showering was obtained from U.S. EPA (1992) and can be expressed as:

$$DA_{event} = C_{water} \times K_p^{\,w} \left[\frac{t_{event}}{1+B} + 2 \; (\frac{1+3B}{1+B}) \right] \times \frac{1}{10^3} \qquad (5\text{-}28)$$

where

DA_{event} =   dose absorbed per unit area per event (mg/cm^2)
C_{water} =   water concentration (mg/L)
$K_p^{\,w}$ =   skin permeability constant in water (cm/h)
t_{event} =   duration of event (h)
        =   lag time (h)
B =   bunge constant (unitless)
$1/10^3$ =   L/ cm^3.

From the dose absorbed, hazard quotients can be calculated for the constituents of concern by applying the following equation:

$$HQ = \frac{DA_{event} \text{ x } EF \text{ x } SA_{skin}}{RfD \text{ x } BW} \tag{5-29}$$

where

HQ	=	hazard quotient (unitless)
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2)
EF	=	event frequency (showers per day)
SA_{skin}	=	surface area of skin (cm^2)
RfD	=	oral reference dose (mg/kg/d)
BW	=	body weight (kg).

No RfDs or slope factors are available for the dermal route of exposure; however , Appendix A of RAGS gives some general guidance for calculating intakes via the dermal route and making appropriate comparisons with RfDs or slope factors. In sum, oral RfDs and slope factors may need to be adjusted (depending on the constituent) based on the oral absorption efficiency of the constituent. RfDs are multiplied by the oral absorption efficiency and slope factors are divided by the oral absorption efficiency. Generally, oral absorption efficiency data are not readily available and have to be researched independently. Alternatively, EPA Region 4 has adopted the following oral absorption efficiencies as interim default values: 80 percent for volatile organic chemicals, 50 percent for semivolatile organic chemicals, and 20 percent for metals (U.S. EPA, 1995). This analysis used the EPA Region 4 default values for dermal exposure.

6.0 Exposure and Toxicity Assessments

Exposure factor data used in the risk analysis are from the *Exposure Factors Handbook* (U.S. EPA, 1997c). The health benchmark values are based on the values presented in the online database Integrated Risk Information System (IRIS) or the Health Effects Assessment Summary Tables (HEAST).

6.1 Exposure Factors

The exposure assumptions used in this risk assessment are confined to the groundwater pathway for each receptor scenario. The exposure pathways evaluated for this analysis are presented in Table 6-1.

6.1.1 Exposure Parameters

The exposure parameters required for these pathways are drinking water intake rates, inhalation rates, body weights, and skin contact rate and body surface area; frequency of showers; and exposure duration. Data for these parameters are presented in this section. All household water is assumed contaminated, thus, the fraction contaminated is 1.

6.1.1.1 Drinking Water. For the deterministic analysis, drinking water intake was not a high-end parameter, and recommended central tendency intakes of 1.4 L/d for adults (mean value for adults) and 0.74 L/day for children (mean value for children ages 1 to 10 years) were used (Table 3-6 of the *Exposure Factors Handbook*, U.S. EPA, 1997c).

Table 6-1. Groundwater Exposures for Receptor Scenarios

Item	Farmer	Child of Farmer
Ingestion of drinking water	✓	✓
Inhalation during showering	✓	✓
Inhalation in bathroom (after showering)	✓	✓
Inhalation in the house (not including bathroom and shower)	✓	✓
Dermal exposure during bathing and showering	✓	a

[a] Dermal exposure factors are not available for children and there is a high degree of uncertainty associated with the estimated risk to children for this pathway.

Table 6-2. Exposure Parameters for Intake of Drinking Water (mL/d)

Age Group (years)	N	Data Mean	Data Standard Deviation	50th Percentile	90th Percentile	95th Percentile	Distribution	Population-Estimated Mean	Population-Estimated Standard Deviation
1-5	3,200	697.1	401.5	616.5	1,236	1,473	Gamma	698	406
20+	13,394	1,384	721.6	1,275	2,260	2,682	Gamma	1,383	703

Drinking water intake exposure factors used for the probabilistic analysis are presented in Table 6-2. Drinking water intake data were obtained from Table 3-6 of the EFH. Data (in mL/d) are presented by age groups. Weighted averages of percentiles, means, and standard deviations were calculated for children ages 1 to 5 and for adult age groups. Percentile data were used to fit parametric models (gamma, lognormal, Weibull) using maximum likelihood estimation. Based on a chi square goodness-of-fit test (at $p = 0.05$), a gamma distribution was selected for both age groups.

6.1.1.2 Inhalation Rate. For the deterministic analysis, inhalation rate was not a high-end parameter. For the adult age group, an inhalation rate of 15.2 m^3/d, the recommended value in the EFH for men was used. For children, the EFH-recommended value of 6.8 m^3/d for children ages 1 to 2 , was used as a protective assumption.

Probabilistic modeling inhalation parameters used in this analysis are presented in Table 6-3. No percentile data were available for inhalation rate and the default lognormal model was assumed. An analysis of inhalation data by Myers et al. (1998) found that, for ages under 3, CV was close to 70 percent and, for other age groups, it was close to 30 percent. The lognormal distribution was fitted by using CV=70 percent for child 1, CV=50 percent ((30+70)/2) for child 2, and CV=30 percent for child 3, child 4, and adult age groups. The exposure factors associated with showering and dermal exposures are presented in Table 6-4.

Table 6-3. Exposure Parameters for Inhalation

Age Group (years)	Distribution	Population-Estimated Mean (m^3/d)	Population-Estimated Standard Deviation (m^3/d)
1-5 yr	Lognormal	7.55	3.78
Adult	Lognormal	13.3	3.99

Table 6-4. Exposure Factors Used To Determine Risk (or HQ) from Inhalation Exposure to Contaminated Tap Water

Exposure Factor	Parameter Value	Reference
Shower duration	10 min	U.S. EPA, 1997d (Table 15-20) (represents time spent showering only)
Time in bathroom (includes shower duration, time spent in shower stall after showering, and time spent in bathroom after leaving shower stall)	40 min	U.S. EPA, 1997d (Tables 15-20, 15-23, and 15-114) (Table 15-114 represents 24-h cumulative time spent in bathroom, which is conservative to assume that all of this time occurs immediately following the shower)
Shower rate	5.5 L/min	Calculated (based on drop diameter and nozzle velocity)
Shower/bath water use	15 gallons per capita per day (gcd)	U.S. EPA, 1997d (Table 17-14) (median value across several studies)
Bathroom water use	35.5 gcd	U.S. EPA, 1997d (Table 17-14) (summation of median values for *shower*, *toilet*, and one-half of the *other* water use rates; RTI assumed *other* represents water use in sinks)
House water use	17.5 gcd	U.S. EPA, 1997d (Table 17-14) (Summation of median values for *Laundry, Dishwashing*, and one-half of the *other* water use rates; RTI assumed *other* represents water use in sinks)
Volume of shower stall	2 m^3	McKone, 1987
Volume of bathroom	10 m^3	McKone, 1987
Volume of house	369 m^3	U.S. EPA, 1997d (Table 17-31)
Volumetric gas exchange rate between shower and bathroom	100 L/min	RTI-derived value
Volumetric gas exchange rate between bathroom and house	300 L/min	RTI-derived value
Volumetric gas exchange rate between house and atmosphere	0.45 air changes per hour (2,768 L/min)	U.S. EPA, 1997d (Table 17-31) (median value; given a low overall confidence rating)
Fraction emitted, bathroom	0.50	Calculated
Fraction emitted, house water	0.66	Calculated
Time toilet emits	40 min/d	U.S. EPA, 1997d (Tables 15-20, 15-23, and 15-114) (Table 15-114 represents 24 h-cumulative time spent in bathroom, which is conservative to assume that all of this time occurs immediately following the shower)
Time house water emits	15.7 h/d	U.S. EPA, 1997d (based on cumulative time spent indoors at a residence, Table 15-131, minus time spent in bathroom [see above])

Table 6-5. Exposure Parameters for Body Weight (kg)

Age Group (years)	N	Data Mean	Data Standard Deviation	50th Percentile	90th Percentile	95th Percentile	Distribution	Population-Estimated Mean	Population-Estimated Standard Deviation
1-5	3,762	15.52	3.719	15.26	18.32	19.45	Lognormal	15.5	2.05
20+	12,504	71.41	15.45	69.26	89.75	97.64	Lognormal	71.2	13.3

6.1.1.3 <u>**Body Weight**</u>. For the deterministic analysis, body weight was not a high-end parameter. Therefore, the EFH-recommended value for adults of 70 kg was used. For children, the EFH-recommended median value of 15.2 kg for the 1- to 5-yr-old age group was used as a protective assumption.

Body weight data for the probabilistic analysis are provided in Table 6-5. These data were obtained from Tables 7-2, 7-3, 7-4, 7-5, 7-6, and 7-7 of the EFH. Data (in kilograms) are presented by age and gender. Weighted averages of percentiles, mean, and standard deviations were calculated for children ages 1 to 5. For adult age groups, male and female data were weighted and combined for each age group. Percentile data were used as the basis for fitting distributions. These data were analyzed to fit parametric models (gamma, lognormal, Weibull) using maximum likelihood estimation. Measures of goodness-of-fit were used to select the most appropriate model. In this case, lognormal distributions were determined to be the most appropriate for both age groups and were, therefore, used for this analysis.

6.1.1.4 <u>**Exposure Duration (Residence Time)**</u>. For the deterministic analysis, exposure duration was a high-end parameter for carcinogenic compounds. Therefore, a high-end value of 30 years for adults was used for carcinogens. Exposure duration was not a high-end parameter for noncarcinogenic constituents and 9 years was used for the deterministic analysis to calculate annual average exposure. This 9-year average corresponds to chronic exposure. These are the recommended values presented in the EFH for deterministic risk assessments. For children, a central tendency exposure duration of 7.3 years was used for noncarcinogens and a high-end exposure duration of 18 years (95th percentile for a 6, 9, and 12 year old) was used for carcinogens. Although these exposure durations for children result in children being older than 5 years of age at the end of the exposure period, a simplified modeling procedure was used so that no other exposure factors were adjusted to account for aging (e.g., changes in body weight or inhalation rate).

Exposure duration data used for the probabilistic modeling are presented in Table 6-6. Residence duration for farmers was used for the probabilistic analysis because this population segment is the least mobile and thus provides the distribution of data that is more conservative than distributions for other segments of the population. For residence duration (in years) for farmers and rural residents (Israeli and Nelson, 1992, as cited in U.S. EPA 1997e, Tables 15-163 and 15-164), the gamma distribution was used because it is the best fitted model in five age groups and the second best fitted model in two cases (based on data by Johnson and Capel, 1992,

Table 6-6. Exposure Parameters for Exposure Duration (Years)

Age Group (years)	N	Data Mean	Data Standard Deviation	50th Percentile	90th Percentile	95th Percentile	Distribution	Population-Estimated Mean	Population-Estimated Standard Deviation
1-5	1	6.5	9.75	5	13	17	Lognormal	6.53	5.6
Farmer		17.31	18.69				Gamma	17.31	18.69
Rural		7.8	11.28				Gamma	7.8	11.28

as cited in U.S. EPA 1997e, Tables 15-167 and 15-168). Goodness-of fit (GOF) was calculated for each of the four models using the chi-square test (Bickel and Doksum, 1977). For the two-parameter models, GOF was calculated against the generalized gamma model using the maximized log likelihood (likelihood ratio) test (Bickel and Doksum, 1977). Where percentile data were available but sample sizes were unknown, a regression F-test for GOF against the generalized gamma model was used. A population mean and a population standard deviation of 17.31 and 18.69 years, respectively, for farmers, were used in the analysis. For noncarcinogens a 9-year exposure duration was used to calculate annual average exposures. Again, for children, no other exposure factors were adjusted to account for aging, as discussed above.

6.2   Toxicity Assessment

The health benchmark values for metals used in this risk analysis were based on the values presented in IRIS or HEAST. The health benchmark values are documented in Appendix E, Constituent Specific Parameters. Summaries of the toxicity studies used as a basis for these values are presented in Appendix F, Toxicity Summaries.

6.2.1   Dose-Response Assessment

The purpose of the dose-response assessment was to determine the most sensitive health effects associated with a constituent and to attempt to express the relationship between dose and effect in quantitative terms. These quantitative terms are known as toxicity values or health benchmarks. Generally, health benchmarks are developed by EPA and are listed in IRIS (U.S. EPA, 1998b) or HEAST (U.S. EPA, 1997f). Four general types of health benchmarks are developed: reference doses (RfDs), reference concentrations (RfCs), cancer slope factors (CSFs), and unit risk factors (URFs).

RfDs and RfCs are used to evaluate noncancer effects for ingestion and inhalation exposures, respectively, and are defined as "an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime" (U.S. EPA, 1989). RfDs are expressed in milligrams of chemical intake per kilogram body weight per day (mg/kg-d), and RfCs are expressed as milligrams of chemical per cubic meter of air (mg/m^3) and both are applied to all receptors.

6.2.1.1 <u>Carcinogens</u>. CSFs and URFs may be derived from a number of statistically and/or biologically based models. Traditionally, the linearized multistage model has been the default model for extrapolating cancer risk estimates to low doses; however, other models have been used. Although several models may provide a good fit to the experimental data, the risk estimates at low doses may be different by several orders of magnitude. Under EPA's proposed cancer risk guidelines (61 FR 17960), significant changes to the default methodology are proposed. Although the new methodology has been used to develop some benchmarks listed in IRIS (e.g., PCBs), all of the cancer benchmarks used in this report are based on the linearized multistage model.

CSFs and URFs are used to evaluate cancer effects for ingestion and inhalation exposures, respectively. Unlike RfDs and RfCs, CSFs and URFs do not represent "safe" exposure levels, rather they are expressed as an upperbound slope factor that relates levels of exposure with a probability of effect or risk. The CSF is expressed in units of $(mg/kg/d)^{-1}$ and the URF for inhalation exposures is expressed in units of $(\mu g/m^3)^{-1}$.

6.2.1.2 <u>Noncarcinogens</u>. The RfD and RfC have been the primary benchmarks used to evaluate noncarcinogenic hazards posed by environmental exposures to chemicals and are based on a fundamental concept of toxicology known as the threshold approach. The threshold approach is based on the theory that there is a "safe" exposure level (the threshold) that must be exceeded before a toxic effect occurs.

RfDs and RfCs do not provide true dose-response information but are essentially estimates of an exposure level or concentration that is believed to be below the threshold level or no observed adverse effects level (NOAEL). It also is important to understand that all RfDs are not necessarily equivalent expressions of toxicity. The degree of uncertainty and confidence levels in the various RfDs varies a great deal and is based on different toxic effects. RfDs and RfCs that have been verified by an intra-Agency workgroup are listed in IRIS.

RfDs and RfCs are derived from the highest NOAEL identified in human epidemiological studies or from subchronic (generally a 90-day study in rats and mice) or chronic study in laboratory animals. If a NOAEL is not identified in any of the available studies, the lowest observed adverse effect level (LOAEL) is used. If the studies report dose levels as parts per million (ppm) in the diet or water, the dose levels are converted to mg/kg/d based on the consumption level and body weights of the test subjects. It is generally assumed that dose levels expressed on a mg/kg/d basis are equivalent in humans and animals; therefore, dose adjustments are not necessary unless chemical-specific pharmacokinetic data indicate that a dose adjustment is appropriate. NOAELs and LOAELs are adjusted ($NOAEL_{adj}$ or $LOAEL_{adj}$) for exposure protocols that are not continuous (i.e., less than 7 days per week or 24 hours per day). Differences in respiratory rates and respiratory physiology between humans and laboratory animals are well recognized; therefore, NOAELs and LOAELs identified from inhalation studies are converted to the human equivalent concentration ($NOAEL_{HEC}$ or $LOAEL_{HEC}$) before deriving the RfC. The RfC methodology is described in detail in U.S. EPA (1994b).

Once a suitable NOAEL or LOAEL has been identified, the characteristics and the quality of the database are examined and the NOAEL or LOAEL is divided by uncertainty factors and

modifying factors to derive the RfD or RfC. Factors of 3 or 10 are commonly used as uncertainty factors. The default value for the modifying factor is 1. All uncertainty factors and modifying factors are multiplied together to derive the total uncertainty factor.

6.2.2   Health Benchmarks

Sources for existing toxicological benchmarks, listed in order of preference, are U.S. EPA's IRIS (U.S. EPA, 1998b) and HEAST (U.S. EPA, 1997f). All published benchmark values and their sources are listed in Appendix D. U.S. EPA values are the only values used in this analysis.

6.2.3   Adjustment of Oral Health Benchmarks for Use with Dermal Exposures

Most toxicity values are based on the administered dose (e.g., the amount given in the food) rather than the amount absorbed. Therefore, in some cases, it is necessary to adjust toxicity values to ensure that they match the exposure estimates. In other words, exposures may be based on an "administered" or "applied" dose, or they may be based on an absorbed dose. Typically, ingestion exposure estimates are based on an administered dose; however, dermal exposure estimates consider the amount absorbed through the skin. EPA has not developed toxicity values for dermal exposures. Instead, RfDs and CSFs are adjusted, if necessary, to represent an absorbed dose. This is accomplished by multiplying the RfD by the measured or predicted absorption efficiency of the chemical from the gastrointestinal tract or by dividing the CSF by the absorption efficiency (U.S. EPA, 1989). The following default oral absorption efficiencies were used: 80 percent for VOCs, 50 percent for SVOCs, and 20 percent for metals (U.S. EPA, 1995).

7.0 Risk Characterization

This section of the report summarizes the results of the deterministic and probabilistic risk analysis used to establish risk-based concentrations for spent filter aids and TAM sludge wastes. Risk-based concentrations are presented for each of the 53 constituents considered in this analysis for spent filter aids and TAM sludges. The concentration limits are derived based on estimates of risk via the groundwater pathway to the adult and child receptor from both ingestion and noningestion routes of exposure.

EPA's waste program generally defines risk levels of concern for carcinogens as risks of $1x10^{-5}$ (1 in 100,000) or greater at the upper end of the risk distribution (e.g., 90th or 95th percentile). The level of concern for noncancer effects is generally indicated by a hazard quotient (HQ) of 1 or greater at the upper end of the distribution. The results of this analysis are presented as concentrations for each constituent potentially present in the waste or the landfill leachate that would result in cancer risk $\geq 1x10^{-5}$ or HQ ≥ 1 for spent filter aids and TAM sludge wastes managed in unlined municipal landfills located near existing dye and pigment manufacturing facilities. More detailed results by exposure pathway, receptor, and waste are presented in Appendix F.

7.1 Risk Results for Triarylmethane Sludges

Table 7-1 shows the constituent concentration in the waste that corresponds to the level of concern at the 90th and 95th percentiles of the risk distribution based on probabilistic analysis and corresponding waste constituent concentration from the deterministic analysis. For the probabilistic analysis, the 90th and 95th percentiles are from a distribution of constituent concentrations generated from the Monte Carlo analysis where the risk level is held constant at 1 in 100,000 or the hazard quotient is held constant at a value of 1. Thus, the 90th percentile concentration means that 90 percent of the values for the waste concentration generated in the analysis would result in risk less than 1 in 100,000 or an HQ less than 1. Similarly for the 95th percentile, only 5 percent of the values generated in the analysis would result in higher risk.

Table 7-1 also identifies the pathway (i.e., inhalation or ingestion) and receptor (i.e., adult or child) from which the percentile data are derived. Thus, for the 90th percentile, an adult, oral pathway means that this pathway has the highest risk for all pathways evaluated at that percentile. As can be seen from the table, the 90th and 95th percentiles always have the same "driving pathway," which indicates that this is the highest risk pathway evaluated.

Table 7-2 presents 90th and 95th percentile probabilistic and deterministic results for leachate concentrations corresponding to risk levels of concern.

Table 7-1. Risk-Based Waste Concentrations for TAM Sludge Waste (mg/kg)*

Chemical	CAS Number	Group Number	90% Waste Concentration	90% Waste Concentration		95% Waste Concentration	95% Waste Concentration		Deterministic Waste Concentration
				Receptor	Pathway		Receptor	Pathway	
1,2-Diphenylhydrazine	122667	4	31	Adult	Oral	16	Adult	Oral	11
Acetone	67641	1	3333	Child	Oral	2000	Child	Oral	5500
Aniline	62533	2	17	Adult/Child	Inhalation	11	Adult/Child	Inhalation	18
Azobenzene	103333	4	716	Adult	Oral	360	Adult	Oral	257
Benzaldehyde	100527	2	5000	Child	Oral	2500	Child	Oral	3800
Benzene	71432	2	365	Child	Inhalation	167	Adult/Child	Inhalation	97
Benzidine	92875	2	0.03	Adult	Oral	0.01	Adult	Oral	0.01
Bis(2-ethylhexyl)phthalate	117817	5	NA	NA	NA	NA	NA	NA	120000
Bromodichloromethane	75274	6	156	Adult/Child	Inhalation	70	Adult	Inhalation	46
Chloroaniline, p-	106478	2	250	Child	Oral	143	Child	Oral	160
Chlorobenzene	108907	3	36	Adult/Child	Inhalation	17	Adult/Child	Inhalation	24
Chloroform	67663	2	102	Child	Inhalation	47	Adult/Child	Inhalation	27
Cresol, p-	106445	2	333	Child	Oral	200	Child	Oral	230
Dichlorobenzene, 1,2-	95501	4	1109	Adult/Child	Inhalation	665	Adult/Child	Inhalation	708

Chemical	CAS Number	Group Number	90% Waste Concentration	90% Waste Concentration		95% Waste Concentration	95% Waste Concentration		Deterministic Waste Concentration
				Receptor	Pathway		Receptor	Pathway	
Dichlorobenzene, 1,4-	106467	4	2607	Adult	Oral	1276	Adult	Oral	935

* Relevant data for some constituents are not included due to business confidentiality concerns   NA - not applicable because the concentration was > 1,000,000

Table 7-1. (continued)

Chemical	CAS Number	Group Number	90% Waste Concentration	90% Waste Concentration		95% Waste Concentration	95% Waste Concentration		Deterministic Waste Concentration
				Receptor	Pathway		Receptor	Pathway	
Dimethoxybenzidine, 3,3'-	119904	2	518	Adult	Oral	238	Adult	Oral	140
Diphenylamine	122394	4	26524	Adult	Oral	12352	Adult	Oral	18000
Ethylbenzene	100414	3	3429	Adult/Child	Inhalation	1664	Adult/Child	Inhalation	2200
Formaldehyde	50000	1	7000	Child	Oral	2500	Child	Oral	7143
Methyl isobutyl ketone	108101	2	42	Adult/Child	Inhalation	27	Adult/Child	Inhalation	40
Methylene chloride	75092	6	927	Adult	Oral	414	Adult	Oral	325
Naphthalene	91203	4	17	Adult/Child	Inhalation	9	Adult/Child	Inhalation	40
N-N-Dimethylaniline	121697	2	300	Child	Oral	143	Child	Oral	167
N-nitrosodiphenylamine	86306	4	7393	Adult	Oral	3686	Adult	Oral	2657
Phenol	108952	2	833	Adult/Child	Inhalation	416	Adult/Child	Inhalation	20000
Phenylenediamine, o-	95545	1	61	Adult	Oral	30	Adult	Oral	36
Phenylenediamine, p-	106503	1	5000	Child	Oral	2500	Child	Oral	6200
Pyridine	110861	2	29	Adult/Child	Inhalation	17	Adult/Child	Inhalation	31
Toluene	108883	3	665	Adult/Child	Inhalation	333	Adult/Child	Inhalation	400
Toluidine, o-	95534	2	13	Adult	Oral	10	Adult	Oral	7

Chemical	CAS Number	Group Number	90% Waste Concentration	90% Waste Concentration		95% Waste Concentration	95% Waste Concentration		Deterministic Waste Concentration
				Receptor	Pathway		Receptor	Pathway	
Toluidine, p-	106490	2	23	Adult	Oral	15	Adult	Oral	10
Trichlorobenzene, 1,2,4-	120821	4	3840	Adult/Child	Inhalation	1997	Adult/Child	Inhalation	2628
Xylenes (total)	1330207	3	1247	Adult/Child	Inhalation	713	Adult/Child	Inhalation	907

Table 7-2. Risk-Based Leachate Concentrations for TAM Sludge Waste (mg/L)*

Chemical	CAS Number	Group Number	90% Leachate Concentration	90% Leachate Concentration		95% Leachate Concentration	90% Leachate Concentration		Deterministic Leachate Concentration
				Receptor	Pathway		Receptor	Pathway	
1,2-Diphenylhydrazine	122667	4	0.0042	Adult	Oral	0.003	Adult	Oral	.0038
Acetone	67641	1	5.56	Child	Oral	3.67	Child	Oral	8.6
Aniline	62533	2	0.029	Adult/Child	Inhalation	0.02	Adult/Child	Inhalation	0.045
Azobenzene	103333	4	0.013	Adult	Oral	0.0086	Adult	Oral	0.012
Benzaldehyde	100527	2	5.56	Child	Oral	3.67	Child	Oral	8.3
Benzene	71432	2	0.11	Adult/Child	Inhalation	0.69	Adult/Child	Inhalation	0.083
Benzidine	92875	2	0.000023	Adult	Oral	0.000013	Adult	Oral	0.00002
Bis(2-ethylhexyl)phthalate	117817	5	0.17	Adult	Oral	0.048	Adult	Oral	.0019
Bromodichloromethane	75274	6	0.0595	Adult	Inhalation	0.037	Adult	Inhalation	.048

Table 7-2. (continued)

Chemical	CAS Number	Group Number	90% Leachate Concentration	90% Leachate Concentration		95% Leachate Concentration	90% Leachate Concentration		Deterministic Leachate Concentration
				Receptor	Pathway		Receptor	Pathway	
Chloroaniline, p-	106478	2	0.25	Child	Oral	0.15	Child	Oral	0.010
Chlorobenzene	108907	3	0.0036	Adult/Child	Inhalation	0.0025	Adult/Child	Inhalation	0.32
Chloroform	67663	2	0.042	Adult/Child	Inhalation	0.026	Adult/Child	Inhalation	0.0065
Cresol, p-	106445	2	0.33	Child	Oral	0.2	Child	Oral	0.41
Dichlorobenzene, 1,2-	95501	4	0.043	Adult/Child	Inhalation	0.03	Adult/Child	Inhalation	0.075
Dichlorobenzene, 1,4-	106467	4	0.11	Adult	Oral	0.066	Adult	Oral	0.098
Dimethoxybenzidine, 3,3'-	119904	2	0.38	Adult	Oral	0.22	Adult	Oral	0.0055

* Relevant data for some constituents are not included due to business confidentiality concerns.

Chemical	CAS Number	Group Number	90% Leachate Concentration	Receptor	Pathway	95% Leachate Concentration	Receptor	Pathway	Deterministic Leachate Concentration
Diphenylamine	122394	4	1.091	Adult	Oral	0.78	Adult	Oral	1.85
Ethylbenzene	100414	3	0.17	Adult/Child	Inhalation	0.12	Adult/Child	Inhalation	0.32
Formaldehyde	50000	1	11.11	Child	Oral	10	Child	Oral	18
Methyl isobutyl ketone	108101	2	0.042	Adult/Child	Inhalation	0.029	Adult/Child	Inhalation	0.068
Methylene chloride	75092	6	0.067	Adult	Oral	0.41	Adult	Oral	0.53
Naphthalene	91203	4	0.0028	Adult/Child	Inhalation	0.0021	Adult/Child	Inhalation	0.0053
N-N-Dimethylaniline	121697	2	0.11	Child	Oral	0.1	Child	Oral	0.16
N-nitrosodiphenylamine	86306	4	0.62	Adult	Oral	0.37	Adult	Oral	0.55
Phenol	108952	2	0.8	Adult/Child	Inhalation	0.57	Adult/Child	Inhalation	49

Chemical	CAS Number	Group Number	90% Leachate Concentration	90% Leachate Concentration		95% Leachate Concentration	90% Leachate Concentration		Deterministic Leachate Concentration
				Receptor	Pathway		Receptor	Pathway	
				d	n		d	n	
Phenylenediamine, o-	95545	1	0.11	Adult	Oral	0.07	Adult	Oral	0.09
Phenylenediamine, p-	106503	1	10	Child	Oral	7	Child	Oral	16
Pyridine	110861	2	0.042	Adult/Child	Inhalation	0.03	Adult/Child	Inhalation	0.066
Toluene	108883	3	0.071	Adult/Child	Inhalation	0.053	Adult/Child	Inhalation	0.12
Toluidine, o-	95534	2	0.022	Adult	Oral	0.013	Adult	Oral	0.017
Toluidine, p-	106490	2	0.029	Adult	Oral	0.016	Adult	Oral	0.021
Trichlorobenzene, 1,2,4-	120821	4	0.046	Adult/Child	Inhalation	0.034	Adult/Child	Inhalation	0.081
Xylene, m-	108383	3	100	Child	Oral	63	Child	Oral	
Xylene, o-	95476	3	100	Child	Oral	63	Child	Oral	
Xylenes (total)	1330207	3	0.071	Adult/Child	Inhalation	0.051	Adult/Child	Inhalation	0.013

7.2   Risk Results for Filter Aid Waste

Relevant data are not included at present because of business confidentiality concerns.

8.0 Uncertainty

Previous sections of this document present the data, assumptions, and models used to develop risk estimates for spent filter aids and TAM sludge from the manufacture of dyes and pigments disposed of in municipal landfills. This section qualitatively addresses the primary sources of uncertainty within the risk assessment and the effects that this uncertainty have on interpreting the results.

Uncertainty is inherent in the risk assessment process. It occurs because the risk assessment process is complex, and variability is inherent in the environment. The sources of uncertainty may be classified as parameter uncertainty and variability, exposure scenario uncertainty, and model uncertainty. Parameter uncertainty occurs when parameters appearing in equations cannot be measured precisely and/or accurately. Variability refers to the normal variations in physical and biological processes that cannot be reduced with additional data. Variability in this risk assessment has been addressed by using a probabilistic analysis. Exposure scenario uncertainty occurs because of the complexity involved in measuring receptor exposure to constituents of concern. Model uncertainty is associated with all models used in risk assessment and occurs because computer models require simplifications of reality and thus exclude some variables and interactions that influence fate and transport but cannot be included in models due to complexity or lack of data. Each of these issues is described below.

8.1 Parameter Uncertainty

Parameter uncertainty occurs when variables appearing in equations cannot be measured precisely and or accurately. In the dye and pigment risk assessment there are many sources of parameter uncertainty.

8.1.1 Waste Characterization

The factors considered in waste characterization are:

- \# Constituent concentration
- \# Waste quantity
- \# Fraction organic carbon (f_{oc})
- \# Bulk density.

Waste characterization data available for use in this risk assessment are limited. The filter aid waste constituent composition is assumed based on constituents measured in any dye and pigment waste stream during the sampling and analysis phase of this waste listing determination. For TAM sludge, waste constituents are assumed based upon reasonable professional judgement

that the constituents might be expected to be used as starting materials or generated from the manufacturing processes for TAM dyes and pigments. Because of the limited data on constituent concentrations in the dye and pigment waste streams assessed, the Agency used a risk assessment approach designed to develop risk-based concentrations rather than to estimate risk from specific existing concentrations.

Waste quantity data have been claimed as Confidential Business Information (CBI) for spent filter aid waste streams. Therefore a distribution of data were developed based upon the reported quantity data and the 50th and 90th percentile values from this distribution were used in the deterministic analysis and the distribution of all values used in the Monte Carlo analysis. Data relevant to managment practices for TAM sludges are not included due to business confidentiality concerns. A single volume was used in the analysis for TAM sludges. Although this data point is known and there is no variability in a single point, uncertainty surrounds other TAM sludge waste steams that may be generated and managed in this manner in the future.

Other waste characterization data are also limited. For example, waste fraction organic carbon (f_{oc}) values for each waste steam were estimated from limited numbers of data points. The f_{oc} for spent filter aid waste was estimated based on all filter aid samples. However, the foc for TAM sludge was estimated based on data for f_{oc} for all wastewater treatment sludge from the dye and pigment industry, because no TAM sludge was sampled in the sampling and analysis phase. Similar assumptions were made concerning waste bulk density. Assumptions for these values are based on general knowledge of similar waste streams from other sources, because bulk density of these wastes was not measured during sampling. This is also a source of waste characterization uncertainty.

8.1.2   Waste Management Unit

The factors considered in waste characterization are:

*   #   Landfill area
*   #   Landfill depth
*   #   Landfill design and operating criteria.

Risk estimates were based on reported disposal in a municipal landfill. To address the uncertainty associated with the description of the management unit, a national distribution of municipal landfill areas was used in this analysis. This distribution of municipal landfill areas was developed in the 1980s and may no longer reflect the national distribution or, more specifically, the distribution of landfills near dye and pigment manufacturing facilities. This lack of specific data is another source of parameter uncertainty. No national data have been collected on the depth of municipal landfills. For this analysis, landfill depths from permitted landfills in the State of Texas were used (Appendix G). There is much uncertainty concerning these data and especially the use of these data with a national distribution of landfill areas.

In addition, these units were modeled as unlined and uncapped landfills because of the uncertainty associated with liner and cap failure. As protective assumptions, the landfill was modeled as unlined and uncapped. This may tend to result in an overestimation of risk. It does,

however, account for the distinct possibility of liner and/or cap failure at some point during the time frame modeled for this risk assessment (10,000 years). It is impossible to predict when or how that may occur.

8.1.3   Location

The landfills accepting dye and pigment wastes were assumed to be located near the facilities where these wastes are generated. For this analysis, the locations were limited to the sites of the facilities that report generating and disposing of spent filter aids or TAM sludges in a municipal landfill. The parameters associated with location are climate-specific parameters such as precipitation, evapotranspiration, and runoff and hydrogeologic parameters associated with the unsaturated zone beneath the landfill and the aquifer characteristics. The climate uncertainties were addressed by using annual average parameter values associated with each location. For the hydrogeologic parameters, uncertainty was addressed in the probabilistic analysis by using a distribution of parameter values for each variable required by the EPACMTP model for each location-specific hydrogeologic code.

8.1.4   Toxicologic Benchmarks

Sources of uncertainty in toxicological benchmarks include extrapolations of responses at high experimental doses of a contaminant under controlled conditions to low doses under highly variable environmental conditions, and adequacy of the database (e.g., number of studies available, toxic endpoints evaluated, exposure routes evaluated, sample sizes, and length of study).

Cancer slope factors can vary by several orders of magnitude depending on the extrapolation model used. A limited understanding of cancer biology in laboratory animals and humans adds to the uncertainty of identifying true human carcinogens.

Much uncertainty is also associated with the noncancer health benchmarks or reference doses. RfDs may include the following uncertainty factors:

> #   Tenfold factor when extrapolating from valid experimental results in studies using prolonged exposure to average healthy humans. This factor is intended to account for the variation in sensitivity among the members of the human population and is referenced as "10H".

> #   Tenfold factor when extrapolating from valid results of long-term studies on experimental animals when results of studies of human exposure are not available or are inadequate. This factor is intended to account for the uncertainty involved in extrapolating from animal data to humans and is referenced as "10A".

> #   Tenfold factor when extrapolating from less than chronic results on experimental animals when there are no useful long-term human data. This factor is intended to account for the uncertainty involved in extrapolating from less than chronic NOAELs to chronic NOAELs and is referenced as "10S".

\#   Tenfold factor when deriving an RfD from a LOAEL instead of a NOAEL. This
         factor is intended to account for the uncertainty involved in extrapolating from
         LOAELs to NOAELs and is referenced as "10L".

Professional judgment is used to determine the modifying factor (MF), which is an additional uncertainty factor that is greater than zero and less than or equal to 10. The magnitude of the MF depends upon the professional assessment of scientific uncertainties of the study and database not explicitly treated above; e.g., the completeness of the overall database and the number of species tested. The default value for the MF is 1.

In general, toxicologic benchmarks are designed to be conservative (i.e., overestimate risk) because of the uncertainties and challenges associated with condensing all available toxicity data into a single quantitative expression. Therefore, use of the current toxicological benchmarks most likely overestimates risk for the pathways evaluated.

8.2   Exposure Scenario Uncertainties

Exposure modeling relies heavily on default assumptions regarding population activity patterns, mobility, dietary habits, body weights, and other factors. Risk estimates presented in this document address hypothetical chronic exposures for various receptors designed to provide a realistic range of potential scenarios; therefore, predictions for long-term average exposures are required for each receptor. Although it is possible to study various populations to determine various exposure parameters (e.g., age-specific soil ingestion rates or intake rates for food) or to assess past exposures (epidemiological studies) or current exposures, risk assessment is about prediction. Therefore, long-term exposure monitoring in this context is infeasible. The *Exposure Factor Handbook* (U.S. EPA 1997c,d,e) provides the current state-of-the-science regarding exposure modeling and assumptions and is used throughout this document. To the extent that actual exposure scenarios vary from the assumptions used in this risk assessment, risks could be underestimated or overestimated.

The Monte Carlo analysis for the adult exposure scenario addresses the variability in the exposure modeling by using distributions of exposure factors. These distributions are presented in Section 6.0 and are assumed to be adequate to address the variability in individual adult exposures required for this risk assessment.

Greater uncertainty is associated with the child exposure scenario than the adult exposure scenario. The child is not a static receptor, and intakes and body weights change simultaneously over time. For the Monte Carlo analysis conducted for this listing determination, a simplifying assumption was made that is known to bias the results in a conservative direction. The drinking water intakes, inhalation rates, and body weights for the single age range of children (1 to 5 years of age) were used, but the exposure duration at these higher doses was assumed to continue for the entire childhood exposure duration (ranging from 1 year to a maximum of 27 years) for children. This produces a somewhat higher risk estimate for children than would be expected if modeled in way that considers the aging of children. The principle consideration is the intake

(water or air) divided by body weight. This ratio can be up to three times higher for the 1- to 5-year-old child than for the 12- to 19-year-old child.

8.3   Model Uncertainty

Model uncertainty is associated with all models used in all phases of a risk assessment. These include the models used to assess toxicity as well as the computer models used to predict the fate and transport of chemicals in the environment. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in the models due either to increased complexity or to a lack of data. Fate and transport models are based on numerous assumptions, simplifications, and an incomplete understanding of factors involved.

The models used in this risk assessment were selected based on science policy and professional judgment. The partitioning model and the groundwater transport models were used because they provide the information needed for this analysis and are generally considered by EPA to be state-of-the-science for this type of analysis. Both have undergone some level of peer review. Even though the models used in this report have been widely used and accepted, they remain a significant source of uncertainty that could result in underestimating or overestimating risk.

9.0 References

Bickel, P. J., and K. A. Doksum. 1977. *Mathematical Statistics*. Holden-Day, San Francisco.

Carsel, R.F., and R.S. Parrish. 1988. Developing joint probability distributions of soil water retention characteristics. *Water Resources Research* 25(5):755-769.

Freeze, R.A., and J.A. Cherry. 1979. *Ground Water*. Prentice-Hall, Inc., Englewood Cliffs, NJ.

Gelhar, L.W., K.R. Welty, and Rehfeldt. 1992. A critical review on field-scale dispersion in aquifers. *Water Res. Research* 28(7):1955-1974.

Geraghty, J.J., D.W. Miller, F. Van Der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. (3rd ed.). Water Information Center, Inc., New York ISBN: 0-912394-03-X.

Israeli and Nelson, 1992, as cited in U.S. EPA, 1997a, *Analysis of Subtitle D Waste Management Units Databases Used in the Groundwater Pathway Analysis for HWIR*, Office of Solid Waste, U.S. EPA, Washington, DC.

Johnson and Capel, as cited in U.S. EPA, 1997a, *Analysis of Subtitle D Waste Management Units Databases Used in the Groundwater Pathway Analysis for HWIR*, Office of Solid Waste, U.S. EPA, Washington, DC.

Jury, W.A., W.F. Spencer, and W.J. Farmer. 1983. Behavior assessment model for trace organics in soil: I. Model description. *J. Environ. Qual.* 12(4):558-564.

Jury, W.A., D. Russo, G. Streile, and H.E. Abd. 1990. Evaluation of volatilization by organic chemicals residing below the soil surface. *Water Resources Research* 26(1):13-20.

Jury, W.A., W.J. Farmer, and W.F. Spencer. 1984. Behavior assessment model for trace organics in soil: II. chemical classification and parameter sensitivity. *J. Environ. Qual.* 13(4):567-572.

Kool, J.B., P.S. Huyakorn, E.A. Sudicky, and Z.A. Saleem. 1994. A composite modeling approach for subsurface transport of degrading contaminants from land-disposal sites. *Journal of Contaminant Hydrology* 17:69-90.

Little, J. C. 1992. Applying the two-resistence theory to contaminant volatilization in showers. *Environ. Sci. Technol.* 26(7):1194-1201.

McKone, T. E. 1987. Human exposure to volatile organics in household tap water: the indoor inhalation pathway. *Environ. Sci. Technol.* 21(12):1194-1201.

McWorter, D.B., and D.K. Sunada. 1977. *Groundwater Hydrology and Hydraulics*. Water Resources Publications, Fort Collins, CO.

Myers, L., J. Lashley, and R. Whitmore. 1998. *Development of Statistical Distributions for Exposure Factors*. Research Triangle Institute, Research Triangle Park, NC.

NOC (Naval Oceanography Command). 1992. International Station Meteorological Climate Summary (CD-ROM), Version 2.0. Jointly produced by: Naval Oceanography Command Detachment Asheville, USAFETAC OL-A, and National Climatic Data Center.

Research Triangle Institute. 1997. *Background Document: Nongroundwater Pathway Risk Assessment; Petroleum Process Waste Listing Determination*. Prepared for EPA Office of Solid Waste. Research Triangle Park, NC.

Shea, J. H. 1974. Deficiencies of elastic particles of certain sizes. *Journal of Sedimentary Petrology*. 44:985-1003.

U.S. EPA (Environmental Protection Agency) 1999. *Background Document for Identification and Listing of the Deferred Dye and Pigment Wastes*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1988. *National Survey of Solid Waste (Municipal) Landfill Facilities*. EPA/530-SW-88-034. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1989. *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual.* EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1991. *National Survey of Hazardous Waste Generators and Treatment Storage, Disposal and Recycling Facilities in 1986: Hazardous Waste Management in RCRA TSDR Units*. EPA 530-SW-91-060. Office of Solid Waste and Emergency Response, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1992. *Dermal Exposure Assessment: Principles and Applications*. Office of Health and Environmental Assessment, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1994a. *The Hydrologic Evaluation of Landfill Performance (HELP) Model Engineering Documentation of Version 3*. EPA/600/R-94/168B. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1994b. *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry*. EPA/600/8-90/066F. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1995. *Supplemental Guidance to RAGs: Region 4 Bulletins Human Health Risk Assessment*. U.S. EPA Region 4, Waste Management Division, Atlanta, GA.

U.S. EPA (Environmental Protection Agency). 1996. *An SAB Report: Review of a Methodology for Establishing Human Health and Ecologically Based Exit Criteria for the Hazardous Waste Identification Rule (HWIR)*. EPA-SAB-EC-96-002. Prepared by the HWIR Subcommittee of the Executive Committee, Science Advisory Board, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996a. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) Background Document*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996b. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) Background Document for the Finite Source Methodology*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1996c. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) Background Document for Metals*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997a. *Analysis of Subtitle D Waste Management Units Databases Used in the Groundwater Pathway Analysis for HWIR*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997b. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) User's Guide*. Office of Solid Waste, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997c. *Exposure Factors Handbook. Volume I*, General Factors. EPA/600/P-95/002Fa. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997d. *Exposure Factors Handbook. Volume II, Food Ingestion Factors*. EPA/600/P-95/002Fb. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997e. *Exposure Factors Handbook. Volume III, Activity Factors*. EPA/600/P-95/002Fc. Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1997f. *Health Effects Assessment Summary Tables. FY 1997 Update*. Office of Emergency and Remedial Response and Office of Research and Development, Washington, DC.

U.S. EPA (Environmental Protection Agency). 1998a. *Addendum to Users Guide for the Industrial Source Complex (ISC3) Dispersion Models. Volume I. User Instructions* (Revised). EPA-454-B-98-003A. Emissions Monitoring and Analysis Division, Office of Quality Planning and Standards, Research Triangle Park, NC.

U.S. EPA (Environmental Protection Agency). 1998b. Integrated Risk Information System (IRIS) Database. Cincinnati, OH. August.