5.0 Estimating Exposure Point Concentrations

Exposure point concentrations are constituent concentrations at the location in the environment where a receptor may be exposed. To determine constituent concentrations in environmental media with which a receptor comes in contact (e.g., groundwater, air, soil), several computer-based models and sets of equations are used. Generally, these include

- Source partition models
- Fate and transport models
- Farm food chain equations
- Aquatic food chain equations.

The three types of waste management units evaluated in this risk assessment are described in detail in Section 4.0. For this risk assessment, it was assumed that paint wastes are deposited in industrial landfills, treatment tanks, and surface impoundments. Chemical constituents found in paint waste are released from these WMUs into the surrounding environment. Releases to the atmosphere occur through volatilization of vapors from all three WMU types. Particulates are released from landfills by wind erosion from the surface of the landfill. Leachate is formed and migrates to groundwater from both landfills and surface impoundments. These materials are then transported by various mechanisms to the air, soil, surface water, and groundwater in the environment immediately surrounding the WMU. Once in the environment, constituents may move into the human food chain by contaminating fruits, vegetables, beef, milk, and fish consumed by humans. Constituents in the environment may also move into the ecological food web by contaminating plants and prey consumed by wildlife.

This risk analysis was performed in both a deterministic and probabilistic or Monte Carlo mode. Section 3.0 explains in detail the risk assessment framework, including the structure of the deterministic and Monte Carlo analyses. The models and algorithms described in this section were used for both types of analyses. Although the calculations were identical for these two types of analyses, the model input parameter values differed. In the deterministic analyses, the model input parameter values were fixed and a single set of results was generated. In the Monte Carlo analysis, certain model input parameter values were varied in each of 10,000 iterations to generate a distribution of media concentrations.

The following subsections describe these models and equations and their application in this risk assessment. Section 5.1 describes the source partition models used to predict environmental releases of constituents from WMUs. Section 5.2 discusses the air dispersion and deposition modeling and methodologies used to estimate constituent-specific soil and water concentrations used in the human health and ecological risk analyses. Groundwater modeling is also presented in Section 5.2, as are the calculations of indoor air concentrations associated with domestic use of contaminated groundwater. The methodology for calculating food chain concentrations based on air, soil, and water concentrations is discussed in Section 5.3.

Greater detail is provided in appendixes to this document:

- Appendix F, Variable Summary of Aboveground Fate and Transport Model. The input values or distributions used in the algorithms presented in Appendix M are presented and referenced in this appendix.

- Appendix K, Modifications to HWIR Source Partition Model Programs. This appendix explains the changes made to the source partition models for use in this risk assessment.

- Appendix L, Source Data. The source partition model input parameter values used in this risk assessment are presented in this appendix for landfills, treatment tanks, and surface impoundments.

- Appendix M, Indirect and Direct Exposure Equations. Algorithms used to calculate air pathway exposure point concentrations for soil, water, terrestrial food chain, and aquatic food chain are documented in this appendix.

- Appendix N, Air Dispersion and Deposition Modeling. This appendix provided details on the air dispersion and deposition modeling for this risk assessment.

- Appendix O, Groundwater Modeling Parameters. The input values or distributions used in the groundwater modeling are presented in this appendix.

- Appendix P, Shower Model. This appendix documents the algorithms used to calculate indoor exposure point concentrations due to showering with contaminated groundwater.

5.1 Source Partition Modeling of Constituent Releases

Source partition models were used to estimate environmental releases of constituents from landfills, treatment tanks, and surface impoundments. Each WMU has different release mechanisms that determine the environmental media impacted:

- **Landfills.** Wastes managed in off-site industrial landfills can release COCs as vapors or particles to the air via windblown erosion or as leachate to the groundwater.

- **Tanks.** Wastes managed in tanks can release COCs into the atmosphere via volatilization. Because tanks contain liquid waste, particulate emissions are not released or do not occur from this WMU. Waste in the tanks was assumed not to leak so that no direct releases to the groundwater or soil occur.

- **Surface Impoundments**. Release mechanisms from surface impoundments include volatilizing to the air and leaching to the groundwater. Because surface impoundments contain liquid waste, particulate emissions are not released or do not occur from this WMU.

Models developed for the Hazardous Waste Identification Rule (HWIR99) for landfills, tanks, and surface impoundments were used to estimate emissions to air and leachate to groundwater from these units. These models were selected because they are considered to be state-of-the-art models for modeling hazardous waste disposal. Additionally, these models are dynamic, they are based on a mass balance, and they include a rigorous hydrology model. The HWIR models consider a variety of removal pathways including aerobic and anaerobic biodegradation, hydrolysis, volatilization, and leaching.

The HWIR models were originally developed to operate within a larger modeling system, which is described in *Overview of the FRAMES - HWIR Technology Software System* (PNNL, 1999). A few basic changes were necessary to allow the model to run as a stand-alone model. These changes are documented in Appendix K.

The source partition models use information for a specific WMU (e.g., surface area), constituent, environmental setting (e.g., precipitation, temperature), and waste stream to estimate environmental release of COCs for each release mechanism. Because the purpose of this assessment was to calculate waste concentrations that are protective of human health and the environment, specific waste concentrations were not initially used in the source models. Instead, the models were executed using a unit waste concentration (e.g., 1 mg/kg). Additionally, the models were initially executed assuming the fraction of waste contaminated (f-wmu) is equal to 1. The results of these source model runs were then used to calculate target waste concentrations in the WMUs. Using waste volume data from the 3007 survey, these target waste concentrations in the WMUs were scaled to target waste concentrations in the paint waste streams (see Section 4.2) using procedures described in Section 8.0.

Table 5-1. WMU and Primary Release Mechanisms

WMU	Volatilization	Windblown Particulates	Leaching
Landfill	✓	✓	✓
Tanks	✓		
Surface impoundments	✓		✓

5.1.1 Landfill Partition Model

The landfill model was developed to approximate the dynamic effects of the gradual filling of active landfills. The landfill is divided into equal-volume vertical cells running from the site surface to the bottom of the landfill, each sized so that it requires 1 year to fill. Waste mass is added gradually, forming layers of waste. After 1 year, the cell is full and the waste is covered with a clean soil cover. Then the next cell begins to fill, and so on until the landfill reaches maximum capacity. Results for the landfill as a whole are then obtained by aggregating stored results for each single cell to account for the time that each cell in the landfill was filled. For example, the results for the landfill at the end of year 3 are a summation of results for the first cell filled at year 3 in the single-cell simulation, the second cell filled at year 2, and the third cell filled at year 1.

The active life of a landfill is assumed to be 30 years; therefore, 30 cells were modeled by the landfill model. The model estimates environmental releases from the landfill beginning in the first year of operation. The landfill fills for 30 years, after which time no more constituent mass is being added and the model accounts for continuous loss of constituent mass. This modeling process continues until 1 percent of the peak constituent mass remains or until 200 years, whichever comes first. The peak 9-year average was calculated from the series of annual model outputs. These 9-year average results were the inputs to the fate and transport models. Two hundred years was selected as a maximum modeling period because, for most constituents, the peak 9-year average is unchanged by modeling greater than 200 years and, in general, would occur during the last 9 years of its active life. Examination of the source partition modeling results in this analysis confirmed that this was true for all constituents evaluated.

Other assumptions made in modeling landfills include

- The landfill is assumed to be below grade so that all precipitation that falls onto the landfill either is evapotranspirated, percolates as infiltration, or increases the moisture content in the unit. That is, storm water run-on, runoff, and erosion are assumed not to occur.

- Each landfill cell can be approximated as a soil column consisting of two homogeneous zones, as shown in Figure 5-1: soil cover and landfill waste. Each zone can be approximated as having homogeneous porous media whose properties are uniform in space and time within the zone but may differ between zones.

- All fluxes (air emissions and leachate) are expressed as averages (g/m^2) over the total surface area of the landfill despite the fact that the footprint of the filled cells is smaller than the total area of the landfill until the last year of operation.

- No daily soil cover is assumed. However, once a landfill cell is filled after 1 year, that cell is assumed to have a permanent, permeable cover, and fate and transport processes through that cover are simulated. Soil cover is assumed to be native soil whose characteristics were developed for a national average soil type.

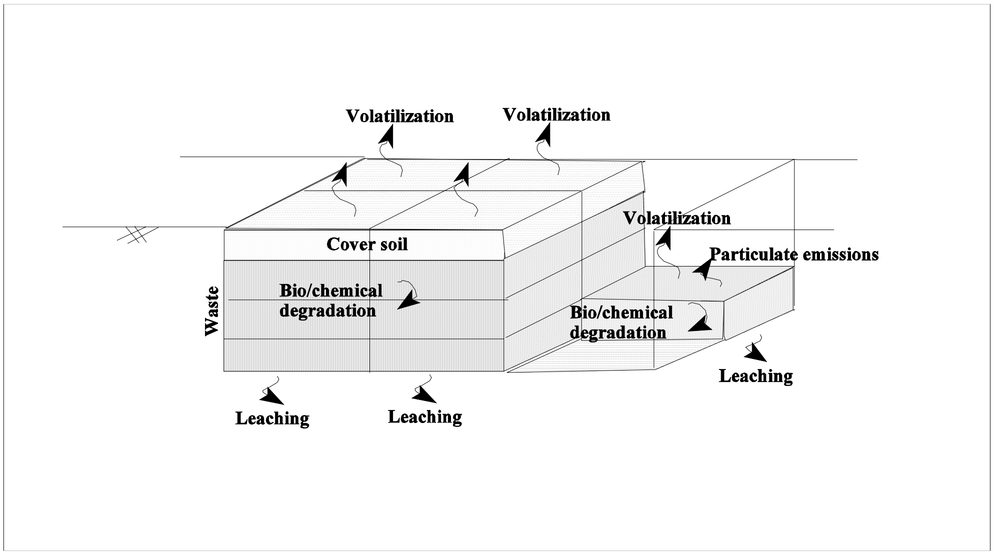

Figure 5-1. Illustration of landfill with six cells and three waste layers.

- The landfill is assumed to be unlined.
- Waste is added to the landfill cell in layers. A waste layer, for the purposes of the model, is simply a zone wherein initial concentrations are assumed to be uniform. Waste layers are conceptualized as being formed over time by the dumping of loads of waste (e.g., via dump truck) next to one another in the landfill cell until eventually a waste layer of uniform depth is formed. At this point, a new layer is started.
- The first-order chemical and biological loss processes that occur in the entire landfill, including cover soil and waste, include anaerobic biodegradation (i.e., decomposition processes that occur in an oxygen-free environment) and hydrolysis.
- First-order loss rates of particulate emissions from the landfill are calculated for losses due to wind erosion, vehicular activity, and spreading and compacting of waste for an active landfill cell. Additionally, losses due to wind erosion from any inactive landfill cell(s) are calculated.
- The annual average infiltration rate is assumed to be unaffected by the cover soil (i.e., cover soil is permeable).

As shown in Figure 5-1, waste is applied in layers within a single cell in the landfill. Because a cell takes 1 year to fill, the time required to lay down one layer is simply 365 days divided by the number of layers in a cell where the number of layers in a cell is a user-specified number. The depth of a waste layer is determined by dividing the depth of the waste zone by the number of layers in the cell. The surface area of a cell is calculated internal to the model and is equal to the load (Mg/yr) divided by the depth of the waste layer (m) times the bulk density of the waste (g/m^3) and a units conversion factor. The number of years the landfill operates then is the total surface area of the landfill divided by the surface area of a single waste cell.

The landfill model consists of several interacting submodels or algorithms, the most significant of which are

- Generic Soil Column Model (GSCM)
- Hydrology model
- Set of calculations that estimates airborne emissions of particulates (and sorbed constituent) due to wind erosion or other surface disturbances (e.g., vehicular traffic).

These three primary submodels are discussed in the following subsections.

5.1.1.1 Generic Soil Column Model. The General Soil Column Model is the fundamental "building block" of the landfill partition model; it describes the dynamic, vertical fate and transport of constituent within a single landfill cell. As stated previously, the GSCM simulation of the processes in the landfill cell begins in the first year of operation and continues through the filling of the cell (1 year), and thereafter until less than 1 percent of the cell's peak constituent mass remains or 200 years—whichever occurs first. The complete time series of the GSCM emissions simulations for the first cell (perhaps 200 years) are stored by the landfill model and are then used to represent new cells being filled in subsequent years. The assumptions and limitations of the GSCM are summarized here and are described in full in the background document *Source Modules for Nonwastewater Waste Management Units (Land Application Units, Waste Piles, and Landfills). Background Document and Implementation for the Multimedia, Multipathway, and Multireceptor Risk Assessment (3MRA) for HWIR 99* (U.S. EPA, 1999b).

- The medium modeled, whether soil, waste, or a soil/waste mixture, can be approximated for modeling purposes as an unconsolidated, homogeneous, porous medium.
- Internal loss processes, which include anaerobic biodegredation and hydrolysis can be considered to proceed in accordance with first-order reaction kinetics.
- The contaminant partitions to three phases: adsorbed (solid), dissolved (liquid), and gaseous. This partitioning of constituent mass is similar to the methods described in Jury et al. (1983, 1990).
- Contaminant partitioning between sorbed and aqueous phases is reversible and linear. Furthermore, the partitioning coefficient is unaffected by changes in concentrations or environmental conditions (e.g., pH, temperature) during the model execution.

- Contaminant partitioning between aqueous and gaseous phases can be described by Henry’s law. The gaseous phase constituent volatilizes from the surface of the landfill to the air.

- Constituent sorbed to surface materials can be lost to the atmosphere via wind erosion or other surface disturbance of particles. These losses are modeled as described in Section 5.1.1.3 and are linked to the GSCM as first-order loss mechanisms.

- The chemical is transported in one dimension through the soil column, a method similar to those methods described in Jury et al. (1983, 1990) and Shan and Stephens (1995).

- Formation of chemical species by chemical or biological processes in the landfill (e.g., daughter products) is not considered. (See Appendix T for results of the analysis conducted to determine whether to consider daughter products.)

- Leaching of aqueous-phase constituent mass occurs by advection or diffusion from the bottom of the WMU or vadose zone.

Greater detail on the GSCM algorithms is provided in the HWIR documentation (U.S. EPA, 1999b).

5.1.1.2 Hydrology Model. The hydrology model provides estimates of daily soil moisture, runoff, evapotranspiration, and infiltration. Runoff is assumed to be zero for the below-grade landfill. These daily estimates are then used by the GSCM algorithm in its daily time step to build up the annual average output variable values. Details for the hydrology algorithms incorporated into the landfill partition model are described in the HWIR documentation (U.S. EPA, 1999b).

5.1.1.3 Particulate Emissions. Particulate chemical fluxes to the atmosphere, representing chemicals sorbed to surficial soils that become airborne, are estimated using empirical relationships developed by EPA (1995) and Cowherd et al. (1985). Using these relationships, the landfill model estimates the total emission rate of contaminants sorbed onto particles that are 30 μm or smaller. In addition, the landfill model estimates the particle size distribution of these emitted solids.

The empirical relationships were developed for specific surface-disturbing activities, consisting of wind erosion, vehicular activity (trucks driving on the landfill surface), waste unloading activities, and waste spreading/compacting activities. For each of these activities, an activity-specific particle size distribution was estimated by EPA (1995), consisting of four particle size categories: 30 to 15 μm, 15 to 10 μm, 10 to 2.5 μm, and less than 2.5 μm. The landfill model estimates the total mass flux of 30 μm or smaller particles being emitted by these various surface disturbances and, based on the fraction of this total from each disturbance activity, calculates the resulting particle size distribution by mass-weighting the component distributions. It should be noted, however, that, when the ISC air model is run prior to the landfill model (as was necessary for this project due to scheduling constraints), this landfill model-generated, run-specific particle size distribution is not available for ISC. In addition, in order to use the landfill-model-generated particle size distributions, the ISC model would have had to be run for each iteration of the Monte Carlo simulation. Given these time constraints, a default distribution based on wind erosion only was assumed for the ISC model. The default wind-erosion-only distribution is: 30 to 15 μm (40 percent), 15 to 10 μm (10 percent), 10 to 2.5 μm (30 percent), and less than 2.5 μm (20 percent). While some paint waste such as emission control dust may be made up primarily of the smaller size particles, we assumed for this analysis that the waste material becomes well-mixed with other wastes and soils before being emitted as a particulate; therefore the default distribution (based on wind-eroded soil particles) is assumed to remain applicable.

5.1.1.4 Model Input Parameters for Landfill Model. The input parameters for the landfill model can be categorized into physical, operating, meteorological, soil, vadose, bulk waste, waste stream, and chemical input parameters. These parameters were developed for 68 landfills, 49 meteorological sites, 2 waste streams, and 43 constituents. Section 4.3 presents a detailed discussion of the location-dependent parameters, which are based on the location of the 49 meteorological sites used in this analysis. Table 5-2 presents the WMU model inputs required by the landfill model. Details pertaining to the parameter values used in the landfill partition model are presented in Appendix L.

The bulk waste parameters are defined in terms of industrial landfill waste and are not specific to paint waste. Additionally, it is assumed that soil values are a good representation of industrial landfill waste. These parameters were obtained from soils data and used the average value from four soil textures as surrogates. Because these parameters are correlated, empirical distributions were developed (see Section 4.3). There were two waste streams modeled within the landfill model: emission control dust and combined solids. Because combined solids had a significantly higher moisture content than dust, these two waste streams were modeled separately. Constituent-specific parameters are also listed. These parameter values are presented in Appendixes D and H.

5.1.1.5 Landfill Partition and Model Results. The emission rates (g/s) for particulate and volatile emissions to the air were calculated by the landfill partition model and then used in the multipathway fate and transport model (where emissions were converted from g/m^2-d using the surface area of the landfills). Results were generated for 10,000 runs for each of the 43 chemicals and for both the combined solids and emission control dust waste streams. Each run was identified by landfill WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations. Central tendency and high-end results were also calculated for the deterministic analyses. Section 3.2 discusses the deterministic analysis.

5.1.2 Treatment Tanks

The tank model simulates time-varying releases of constituents to the atmosphere. The tank unit has only volatile emissions (no particulates) and is assumed to have an impervious bottom so that there is no contaminant leaching. Therefore, the output from the tank model

Table 5-2. Landfill Partition Model Input Parameters

Parameter	Units	Parameter Type
Landfill Physical and Operating Parameters		
Active life of landfill	yr	Constant
Empty weight (trucks)	Mg	Constant
Fraction of paint waste in WMU	Mass fraction	Constant
Number of local watersheds	Unitless	Constant
Number of waste layers in a cell	Unitless	Constant
Payload volume	m^3	Constant
Size of truck	Unitless	Constant
Source type	Unitless	Constant
Spreading/compacting operations per day	1/d	Constant
Thickness of liner (or subsoil zone)	m	Constant
Waste zone thickness	m	Constant
Wet bulk density (waste)	g/cm^3 or Mg/m^3	Constant
Wheels per vehicle (mean)	Unitless	Constant
Average landfill capacity	Mg	Derived
Average surface area	m^2	Derived
Distance vehicle travels on active LF cell surface	m	Derived
Frequency of surface disturbance per month (active LF cell)	1/mo	Derived
Payload weight (carrying capacity of truck)	Mg	Derived
Source depth	m	Derived
Vehicle weight (mean)	Mg	Derived
Vehicles/day (mean annual)	1/d	Derived
Waste loading rate (dry)	Mg/y	Derived
Roughness ratio (LF waste zone surface)	Unitless	Lognormal
Dust suppression control efficiency	Unitless	Normal
Fraction vegetative cover (inactive LF cell)	Fraction	Normal

(continued)

Table 5-2. (continued)

Parameter	Units	Parameter Type
Landfill Physical and Operating Parameters (continued)		
Vehicle speed	km/h	Normal
Optional soil cover thickness	m	Triangular
Meteorological Parameters		
Site latitude	Degrees	Constant
Meteorological station ID	Unitless	Empirical
Soil Parameters		
Fraction organic carbon (cover soil)	Unitless	Constant
Saturated hydraulic conductivity (LF cover soil)	cm/h	Constant
Saturated water content (cover soil, total porosity)	L/L	Constant
Soil index (soil)	Integer	Constant
Soil moisture coefficient b (LF cover soil)	Unitless	Constant
Vadose Parameters		
Average vadose zone temperature	°C	Constant
Fraction organic carbon (subsoil)	Mass fraction	Constant
Saturated hydraulic conductivity (subsoil)	cm/h	Constant
Saturated water content (subsoil)	L/L	Constant
Soil index (vadose)	Integer	Constant
Soil moisture coefficient b (subsoil)	Unitless	Constant
Bulk Parameters		
Depth (root zone in LF waste zone)	cm	Constant
Porosity (total, waste)	Volume fraction	Constant
Dry bulk density (waste)	g/cm^3 or Mg/m^3	Derived
Mode of aggregate size distribution (LF waste zone surface)	mm	Empirical
Saturated hydraulic conductivity (waste)	cm/h	Empirical correlated
Soil moisture coefficient b (waste)	Unitless	Empirical correlated
Soil moisture field capacity (LF waste zone)	Fraction	Empirical correlated

(continued)

Table 5-2. (continued)

Parameter	Units	Parameter Type
Bulk Parameters (continued)		
Soil moisture wilting point (LF waste zone)	Fraction	Empirical correlated
Fraction organic carbon (waste)	Mass fraction	Triangular
Average waste/source pH	pH units	Uniform
Silt content (waste)	Mass percent	Uniform
Waste Stream Parameters		
Volumetric water content (waste on trucks) - dust	Volume percent	Uniform
Volumetric water content (waste on trucks) - sludge	Volume percent	Uniform
Constituent Parameters		
Aerobic biodegradation rate	1/d	Constant
Air diffusion coefficient	cm^2/s	Constant
Anaerobic biodegradation rate	1/d	Constant
CAS number	Unitless	Constant
Catalyzed hydrolysis	1/d	Constant
Chemical name	Unitless	Constant
Chemical type (O,M,Hg,S,D)	Unitless	Constant
Constituent concentration in waste (dry)	$\mu g/g$	Constant
Henry's law constant	$(atm\text{-}m^3)/mol$	Constant
Soil water partitioning coefficient	mL/g	Constant
Solubility for each media	mg/L	Constant
Temperature assumed for these properties	°C	Constant
Water diffusion coefficient	cm^2/s	Constant
Fraction of chemical concentration in the neutral species at a given pH and T	Fraction	Constant or derived
Partition coefficient medium	L/kg	Derived, empirical, log uniform, or pH-based isotherm

LF = Landfill.

calculates only release of vapor to the aboveground pathways. The tank model is a quasi-steady-state model, and the emissions occur only while the unit is operating. Volatile emissions were calculated for 50 years, which is the specified years of operation for treatment tanks in this analysis.

The tank model used in this assessment comprises the liquid and sediment components of the tank/surface impoundment model developed for HWIR99. A single set of equations and computer code has been developed for the liquid and sediment compartments of the model because of similarities in mass balance and transport between tanks and surface impoundments. The primary difference between tanks and surface impoundments is that a tank is assumed to have an impervious bottom so that there is no contaminant leaching. Both tanks and surface impoundments may be either aerated or quiescent, and the mass transport equations used to describe volatile contaminant losses from these units are the same. Both units may have some degree of solids settling. For aerated units, suspended solids in the influent waste primarily pass through the system with little solids settling (depending on the degree of agitation). For quiescent units, solids settling and accumulation may be significant. When this occurs, the unit has to be cleaned or dredged to remove the accumulated solids.

This section describes the liquid and sediment compartments that were used to model tanks. Surface impoundment partition modeling is discussed in Section 5.1.3.

5.1.2.1 <u>Model Overview</u>. The liquid compartment model

- Uses the mass balance approach, taking into consideration contaminant removal by volatilization, biodegradation, hydrolysis, leaching, and partitioning to solids

- Estimates volatilization rates for both aerated and quiescent surfaces

- Estimates suspended solids removal (settling) efficiency

- Estimates temperature effects.

Temporally, a quasi-steady-state monthly time step is used. Quasi-steady-state refers to the fact that the model employs time steps in its mathematical solution but, within these monthly time steps, steady-state assumptions are made. Each month, the model updates certain parameters based on average monthly environmental conditions (temperature, windspeed, precipitation, and evaporation). It is then assumed that the system equilibrates instantaneously to these new conditions and a steady-state solution is obtained for that month. The resulting 12 monthly values for all outputs are then averaged and reported as annual averages.

Key Assumptions. The general model construct can be useful for a wide variety of WMU applications. For this analysis, the following assumptions were used:

- Two-compartment model: "mostly" well-mixed liquid compartment and a well-mixed sediment compartment, which includes a temporary accumulating solids compartment

- First-order kinetics for volatilization in liquid compartment
- First-order kinetics (e.g., rates of change) for hydrolysis in both liquid and sediment compartment
- First-order kinetics for biodegradation with respect to both contaminant concentration and biomass concentration in liquid compartment
- First-order kinetics for biodegradation in sediment compartment
- Darcy's law for calculating the infiltration rate
- First-order kinetics for solids settling
- First-order biomass growth rate with respect to total biological oxygen demand (BOD) loading
- First-order biomass decay rate within the accumulating sediment compartment
- No contaminant in precipitation/rainfall
- Linear contaminant partitioning among adsorbed solids, dissolved phases, and vapor phases
- Daughter products are not included in the model; any constituents generated as a reaction intermediate or end product from either biodegradation or hydrolysis are not included in the model output.

Due to the simplicity of the biodegradation rate model employed and the use of Henry's law partitioning coefficients, the model is most applicable to dilute aqueous wastes. At higher contaminant concentrations, biodegradation of toxic constituents may be expected to exhibit zero-order or even inhibitory rate kinetics. For waste streams with high contaminant or high total organic concentrations, vapor phase contaminant partitioning may be better estimated using partial pressure (Raoult's law) rather than Henry's law.

Methodologies. The treatment tank is divided into two primary compartments: a "liquid" compartment and a "sediment" compartment. Mass balances are performed on these primary compartments at time intervals small enough that the hydraulic retention time in the liquid compartment is not significantly impacted by the solids settling and accumulation. Figure 5-2 provides a general schematic of a model construct.

In the liquid compartment, there is flow both in and out of the WMU. Solids generation occurs in the liquid compartment due to biological growth; solids destruction occurs in the sediment compartment due to sludge digestion. Using a well-mixed assumption, the suspended solids concentration within the WMU is assumed to be constant throughout the WMU. However, some stratification of sediment is expected across the length and depth of the WMU so

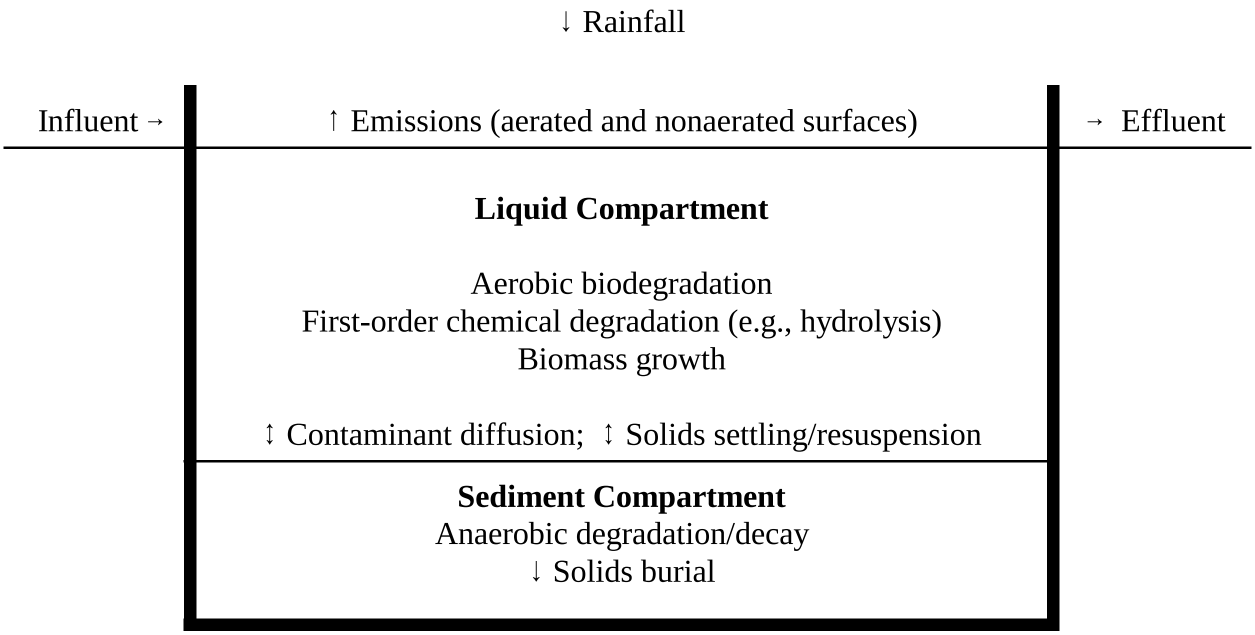

Figure 5-2. Schematic of general model construct for tanks.

that the effective total suspended solids (TSS) concentration within the tank is assumed to be a function of the WMU's TSS removal efficiency rather than equal to the effluent TSS concentration. The liquid (dissolved) phase contaminant concentration within the tank, however, is assumed to be equal to the effluent dissolved phase concentration (i.e., liquid is well mixed). Consequently, the term "mostly well mixed" is used to describe the liquid compartment.

The steady-state, mass balance assumptions on which the model is based are summarized as follows.

Constituent Mass Balance in Liquid Compartment. In the liquid compartment, there is flow both in and out of the WMU. There is also a leachate flow to the sediment compartment and out the bottom of the WMU for surface impoundments. Within the liquid compartment, there is contaminant loss through volatilization, hydrolysis, and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion.

Constituent Mass Balance in Sediment Compartment. Within the sediment compartment, there is contaminant loss through hydrolysis and biodegradation. Additionally, contaminant is transported across the liquid/sediment compartment interface by solids settling and resuspension and by contaminant diffusion.

Solids Mass Balance in Liquid Compartment. Sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. Sedimentation and resuspension are assumed to occur in the quiescent areas. For systems in which biodegradation occurs within the liquid compartment, there is also a production of biomass associated with the decomposition of organic constituents.

Solids Mass Balance in Sediment Compartment. In the sediment compartment, as in the liquid compartment, sedimentation and resuspension provide a means of sediment transfer between the liquid and sediment compartments. In the sediment compartment, however, there is some accumulation of sediment during the time step. This sediment accumulation is also referred to as sediment burial, and the rate of sediment accumulation is determined by the burial velocity. The primary output of the tank model is the annual average volatilization rate for each constituent.

5.1.2.2 Model Input Parameters for Tanks. The input parameters for the tank model can be categorized into tank, meteorological, bulk waste, and chemical input parameters. These parameters were developed for 200 tanks, 49 meteorological sites, and 43 chemicals. Section 4.3 presents a detailed discussion of the location-dependent parameters, which are based on the location of the 49 meteorological stations used in this analysis. Table 5-3 presents the WMU model inputs required by the tank model. Details pertaining to the parameter values used in the treatment tank partition model are presented in Appendix L.

Bulk waste parameters are defined in terms of industrial landfill waste and are not specific to paint waste. Constituent-specific parameters are presented in Appendixes D and H.

5.1.2.3 Treatment Tank Partition Model. The total emission rate (g/s) of volatile emissions to the air was calculated by the treatment tank partition model and then used in the multipathway fate and transport model (where emissions were converted from g/m^2-d using the surface area of the landfills). Results were generated for 10,000 runs for each of the 43 chemicals. Each run was identified by tank WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations. Central tendency and high-end results were also calculated for the deterministic analyses. Section 3.2 discusses the deterministic analysis.

5.1.3 Surface Impoundments

The surface impoundment model simulates time-varying releases of constituent to the atmosphere. The surface impoundment unit is the same as the tank model, but the bottom of the unit is assumed to be pervious so that contaminant leaching can occur. It is assumed that there are no direct liquid discharges to the surface due to overflows or structural failures. Therefore, the output from the surface impoundment model provides air emissions as input for calculations of fate and transport for aboveground pathways and leachate concentration as input for groundwater pathways. The model is a quasi-steady-state model, and the emissions occur only while the unit operates. The constituent releases are calculated for 50 years, which is the specified years of operation for surface impoundments in this analysis. After the operating period, the surface impoundment site is assumed to be closed and all constituents removed.

Table 5-3. Treatment Tank Partition Model Input Parameters

Parameter	Units	Parameter Type
Treatment Tank Partition Model Input Parameters		
Area agitated per aerator	m^2/hp	Distribution
Area of the surface impoundment	m^2	Dependent
Biologically active solids/total solids (ratio)	Unitless	Distribution
Biomass yield	g/g	Distribution
Depth of source	m	Dependent
Digestion (sediments)	1/s	Distribution
Economic life of surface impoundment	yr	Constant
Fraction of paint waste in WMU	Mass fraction	Constant
Fraction of SI occupied by sediments	Fraction	Constant
Fraction surface area turbulent	Fraction	Dependent
Impeller diameter	cm	Constant
Impeller speed	rad/s	Constant
Impellers/aerators (number)	Unitless	Dependent
Impellers/aerators (total power)	hp	Dependent
Maximum total power	hp	Constant
Number of economic lifetimes	Unitless	Constant
Oxygen transfer correction factor	Unitless	Constant
Oxygen transfer factor	lb O_2/h-hp	Constant
Peak output fraction for simulation termination	fraction	Constant
Volumetric influent flow rate	m^3/s	Dependent
WMU type	Unitless	Constant
Meteorological Parameters		
Site latitude	Degrees	Constant
Site longitude	Degrees	Constant
Meteorological station ID	Unitless	Empirical
Ambient temperature of site	°C	Constant
Bulk Waste Parameters		
BOD (influent)	g/cm^3	Distribution

(continued)

Table 5-3. (continued)

Parameter	Units	Parameter Type
Bulk Waste Parameters (continued)		
Bulk density for liquid waste	g/cm^3 or Mg/m^3	Constant
Density (liquid [water])	g/cm^3	Constant
Fraction organic carbon (waste solids)	Mass fraction	Distribution
Molecular weight (liquid [water])	g/mol	Constant
Particle diameter (mean, waste suspended solids)	cm	Distribution
Solids density	g/cm^3	Distribution
Temperature of waste	°C	Dependent
Total suspended solids (influent)	g/cm^3	Distribution
Waste pH	pH units	Distribution
Constituent Parameters		
Aerobic biodegradation rate	1/day	Constant
Air diffusion coefficient	cm^2/s	Constant
Anaerobic biodegradation rate	1/d	Constant
CAS number	Unitless	Constant
Catalyzed hydrolysis	1/d	Constant
Chemical name	Unitless	Constant
Chemical type (O,M,Hg,S,D)	Unitless	Constant
Constituent concentration in waste (dry)	$\mu g/g$	Constant
Henry's law constant	$(atm\text{-}m^3)/mol$	Constant
Soil water partitioning coefficient	mL/g	Constant
Solubility for each media	mg/L	Constant
Temperature assumed for these properties	°C	Constant
Water diffusion coefficient	cm^2/s	Constant
Fraction of chemical concentration in the neutral species at a given pH and T	Fraction	Constant or derived
Partition coefficient medium	L/kg	Derived, empirical, log uniform, or pH-based isotherm

Because the treatment tank and surface impoundment model are the same for the liquid compartment, that discussion is not repeated in this section. See Section 5.1.2 for a full description of the volatile emissions calculations.

5.1.3.1 Model Overview. In addition to volatile emissions, the surface impoundment model also outputs the average annual infiltration rate and the associated average annual leachate contaminant flux rate. Groundwater infiltration is assumed to be driven by the hydrostatic pressure head produced by the wastewater in the unit; when the unit ceases operation, it is assumed that no additional contaminant leaches from the source because the unit is closed and all liquid and sediment removed. Annual liquid infiltration rates and contaminant leachate flux rates are both calculated at the base of the unit and are output for use in the groundwater modeling.

SI Vadose Zone Infiltration. The leachate flux rate is calculated as the dissolved contaminant concentration in the sediment pore water times the infiltration rate of leachate flow. The infiltration algorithm was developed specifically for use in HWIR99 and estimates infiltration rates as a function of liquid depth in the WMU, the thickness and hydraulic conductivity of the sediment compartment (considered as two layers, unconsolidated and consolidated), a native soil layer immediately below the WMU that has been clogged by infiltrating sediment, and an unclogged native soil layer below the clogged layer. A schematic diagram of the conceptual model for the infiltration rate algorithm is shown in Figure 5-3. An iterative method is used to converge on the infiltration rate satisfying the physical requirement that the hydraulic pressure head at the groundwater table must equal atmospheric pressure.

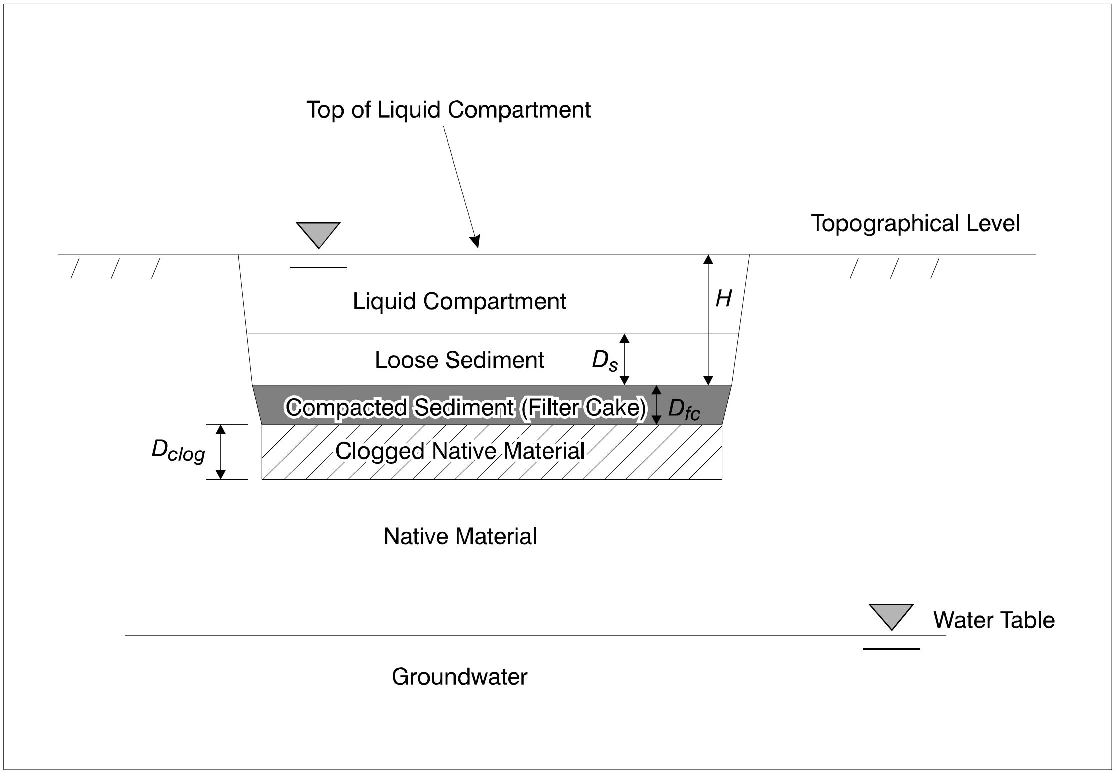

Figure 5-3. Conceptual model for SI infiltration algorithm.

5.1.3.2 Model Input Parameters for Surface Impoundment Source Model. The input parameters for the surface impoundment model can be categorized into physical and operating, meteorological, vadose, bulk waste, and chemical input parameters. These parameters were developed for 200 surface impoundments, 49 meteorological sites, and 43 chemicals. Section 4.3 presents a detailed discussion of the location-dependent parameters, which are based on the locations of the 49 meteorological sites used in this analysis. Table 5-4 presents the WMU model inputs required by the surface impoundment model. Details pertaining to the parameter values used in the surface impoundment partition model are presented in Appendix L.

The bulk waste parameters are defined in terms of industrial landfill waste and are not specific to paint waste. Constituent parameters are presented in Appendixes D and H.

5.1.3.3 Surface Impoundment Partition Model Results. The total emission rate (g/s) of particulate and volatile emissions to the air was calculated by the surface impoundment partition model and then used in the multipathway fate and transport model (where emissions were converted from g/m^2-d using the surface area of the surface impoundment). Results were generated for 10,000 runs for each of the 43 chemicals. Each run was identified by surface impoundment WMU, meteorological site, and a run ID that corresponded to one of the 10,000 Monte Carlo iterations. Central tendency and high-end results were also calculated for the deterministic analyses. Section 3.2 discusses the deterministic analysis.

5.2 Fate and Transport Modeling

This section describes the methodology and the models that were used to predict the fate and transport of chemical constituents in the environment. The methodology is based on three sources:

- *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions, Update* (U.S. EPA, 1998b)
- *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions* (U.S. EPA, 1990)
- *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities. Volume One* (U.S. EPA, 1998a)

Once a constituent is released from a waste management unit, it can move through the air, soil, surface water, groundwater, and food chain by natural processes. This transport into the environment may enable people and wildlife to be exposed to the released constituent. The purpose of the fate and transport modeling performed for this assessment is to estimate the concentration of a constituent in environmental media (i.e., air, water, soil, and food items) at certain locations around a waste management unit where individuals or wildlife may be located. To predict a contaminant's movement through these different media, several media-specific fate and transport models are employed. Fate and transport models typically used by EPA are either a series of computer-based algorithms or sets of equations that predict chemical movement due to natural forces. These fate and transport models integrate information on a site's geology,

Table 5-4. Surface Impoundment Partition Model Input Parameters

Parameter	Units	Parameter Type
Surface Impoundment Physical and Operating Parameters		
Area agitated per aerator	m^2/hp	Distribution
Area of the surface impoundment	m^2	Calculated
Biologically active solids/total solids (ratio)	Unitless	Distribution
Biomass yield	g/g	Distribution
Depth of source	m	Calculated
Digestion (sediments)	1/s	Distribution
Economic life of surface impoundment	yr	Constant
Fraction of paint waste in WMU	Mass fraction	Constant
Fraction of SI occupied by sediments	Fraction	Dependent
Fraction surface area turbulent	Fraction	Dependent
Impeller diameter	cm	Constant
Impeller speed	rad/s	Constant
Impellers/aerators (number)	Unitless	Dependent
Impellers/aerators (total power)	hp	Calculated
Maximum retention time	yr	Constant
Maximum total power	hp	Constant
Number of economic lifetimes	Unitless	Constant
Oxygen transfer correction factor	Unitless	Constant
Oxygen transfer factor	lb O_2/h-hp	Constant
Peak output fraction for simulation termination	Fraction	Constant
Retention time	yr	Calculated
Saturated hydraulic conductivity (sediment layer)	m/s	Distribution
Turbulent area	m^2	Calculated
Viscosity of water - place holder	g/cm-s	Constant
Volumetric influent flow rate	m^3/s	Calculated
WMU type	Unitless	Constant

(continued)

Table 5-4. (continued)

Parameter	Units	Parameter Type
Meteorological Parameters		
Site latitude	Degrees	Constant
Site longitude	Degrees	Constant
Meteorological station ID	Unitless	Empirical
Ambient temperature of site	°C	Constant
Vadose Parameters		
Saturated hydraulic conductivity (subsoil)	cm/h	Constant
Bulk Waste Parameters		
BOD (influent)	g/cm^3	Distribution
Bulk density for liquid waste	g/cm^3 or Mg/m^3	Constant
Density (liquid [water])	g/cm^3	Constant
Fraction organic carbon (waste solids)	Mass fraction	Distribution
Molecular weight (liquid [water])	g/mol	Constant
Particle diameter (mean, waste suspended solids)	cm	Distribution
Solids density	g/cm^3	Distribution
Temperature of waste	°C	Dependent
Total suspended solids (influent)	g/cm^3	Distribution
Waste pH	pH units	Distribution
Constituent Parameters		
Aerobic biodegradation rate	1/d	Constant
Air diffusion coefficient	cm^2/s	Constant
Anaerobic biodegradation rate	1/d	Constant
CAS number	Unitless	Constant
Catalyzed hydrolysis	1/d	Constant
Chemical name	Unitless	Constant
Chemical type (O,M,Hg,S,D)	Unitless	Constant
Constituent concentration in waste (dry)	$\mu g/g$	Constant
Henry's law constant	$(atm\text{-}m^3)/mol$	Constant
Soil water partitioning coefficient	mL/g	Constant

(continued)

Table 5-4. (continued)

Parameter	Units	Parameter Type
Constituent Parameters		
Solubility for each media	mg/L	Constant
Temperature assumed for these properties	°C	Constant
Water diffusion coefficient	cm^2/s	Constant
Fraction of chemical concentration in the neutral species at a given pH and T	Fraction	Constant or Derived
Partition coefficient medium	L/kg	Derived, empirical, log uniform, or pH-based isotherm

hydrology, and meteorology with chemical, physical, and biological processes that can take place in the environment. The result is a simulation of chemical movement in the environment and a prediction of the concentration of a constituent at a certain point called the "exposure point." The following fate and transport models were used for this analysis:

- Air dispersion and deposition model
- Watershed model
- Groundwater model
- Indoor air model (predicts the exposure to contaminants due to showering with contaminated groundwater).

These four models and the general framework for performing the fate and transport modeling are described in the following sections. The fate and transport framework is described in Section 5.2.1. Section 5.2.2 discusses the air dispersion and deposition modeling. Section 5.2.3 describes the watershed model used to determine soil and water constituent concentrations. Section 5.2.4 presents the model and methodology for predicting contaminant movement in groundwater, and Section 5.2.5 discusses the methodology for calculating indoor air concentrations due to domestic use of contaminated groundwater. Detailed descriptions of the models and a comprehensive list of the input values used in them can be found in Appendixes M and F, respectively.

5.2.1 Overview

Two separate fate and transport scenarios were established for the groundwater and air pathways. The decision to use separate scenarios for the two pathways was based on differences in exposure with respect to time frame and location. One scenario evaluates the movement of constituents through environmental media once they are released into the air by either volatilization or particulate release from the waste management unit. The second scenario considers the movement of the constituent through soils and the groundwater once it has leached from the bottom of a WMU. Environmental contamination from air releases can occur immediately while it may take hundreds of years for most contaminants to reach a groundwater well; thus, for most contaminants, the exposure point concentrations predicted did not include simultaneous contributions from both air and groundwater scenarios. In addition, the aboveground receptor locations may not necessarily overlap (i.e., the aboveground receptors are randomly located around the WMU and may not coincide with the location of the groundwater plume).

Both the groundwater and the air pathway modeling were designed to predict exposure point concentrations with which a resident living in the vicinity of the WMU may come into contact during normal daily activities. For example, the air pathway analysis evaluates constituent concentrations in aboveground environmental media (e.g., air, soil, surface water). Aboveground constituent concentrations enter the human food chain via farm food commodities and fish. These farm food chain and aquatic food chain calculations are discussed in Section 5.3. The groundwater pathway analysis estimates the concentration of a constituent that may be present in a resident's drinking water and indoor environment.

5.2.1.1 Air Pathway Fate and Transport Modeling Overview. Constituents released from a WMU either by volatilization of vapors or wind erosion of particulates will travel through the ambient air as a result of air movement. Wind direction and speed will determine how far the constituent travels from the original source of its release. As shown in Figure 5-4, constituents transported through the air can move through many different media: soil, biota, and surface waters. Initially, the wind will disperse the constituent in three dimensions. Due to the forces of gravity and scavenging (mechanism by which constituents are removed from the air by coming into contact with stationary objects, such as trees or by precipitation), the constituent may move out of the ambient air onto plant, soils, or surface waterbodies. The constituent will then continue to move once it has deposited on these areas. For example, precipitation may induce runoff and erosion onto other soils or into surface waters. All of these mechanisms will impact the final concentration at an exposure point. For the air pathway, exposure point concentrations were determined in

- Ambient air
- Soils
- Terrestrial food chain
- Aquatic food chain.

To obtain these exposure point concentrations, air dispersion modeling was used to predict ambient air concentrations and the deposition fluxes of the released constituents. The air concentrations and deposition values predicted were then input into a geographic information system. The GIS was used to integrate the air modeling data over the spatial extent of the geographic features in the study area. For example, the GIS was used to determine the average air concentration over geographic features such as an agricultural field or waterbody. This spatial integration produced normalized, spatially averaged air concentrations and deposition fluxes for areas adjacent to the WMU.

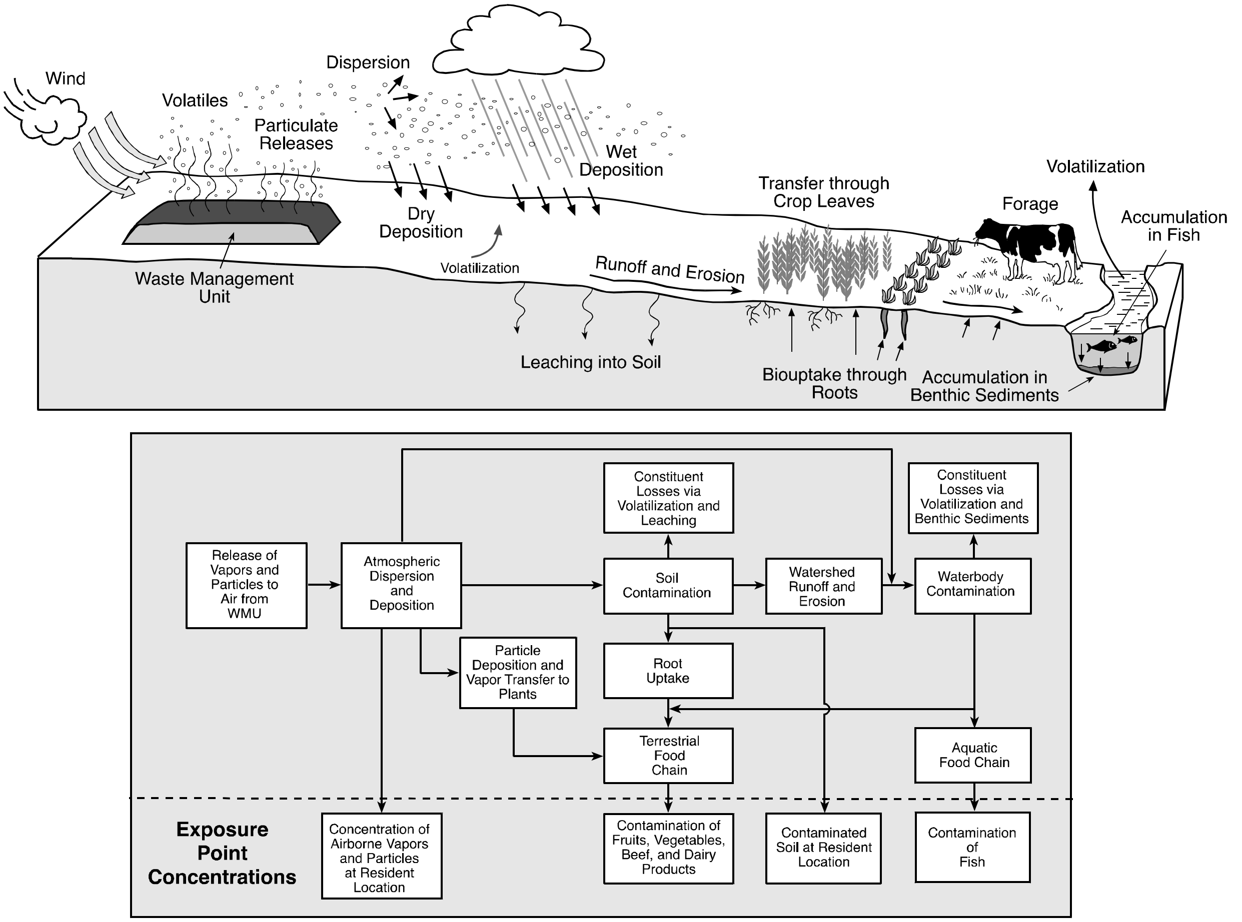

Figure 5-4. Air pathway fate and transport processes and exposure points.

The spatially averaged concentrations were then used as inputs into other models that calculated the fate and transport in other environmental media. For example, contaminants can be removed from the air and deposit on the earth's surface. Once they are on the surface they will continue to cycle through different media: soils, plants, animals, and surface waterbodies. For this analysis, once a contaminant was deposited on the surface or reached the surface media through vapor diffusion, the movement of the constituent in soil, plants, animals, and surface waterbodies was modeled. The contaminant was subject to fate processes such as infiltration into the soil column, uptake into plants, ingestion by animals, and runoff and erosion onto other soil areas or into surface waterbodies. These processes will decrease the contaminant's concentration in certain areas of the environment while they increase its concentration in others. Subsequent sections describe the fate and transport of a constituent in the air pathway.

5.2.1.2 <u>Groundwater Pathway Fate and Transport Modeling Overview</u>. Leachate generated by the landfill and the surface impoundment moves through the vadose zone into the groundwater aquifer. As shown in Figure 5-5, once in the groundwater aquifer, it is transported by groundwater flow to the location of a residential well where it contaminates the groundwater

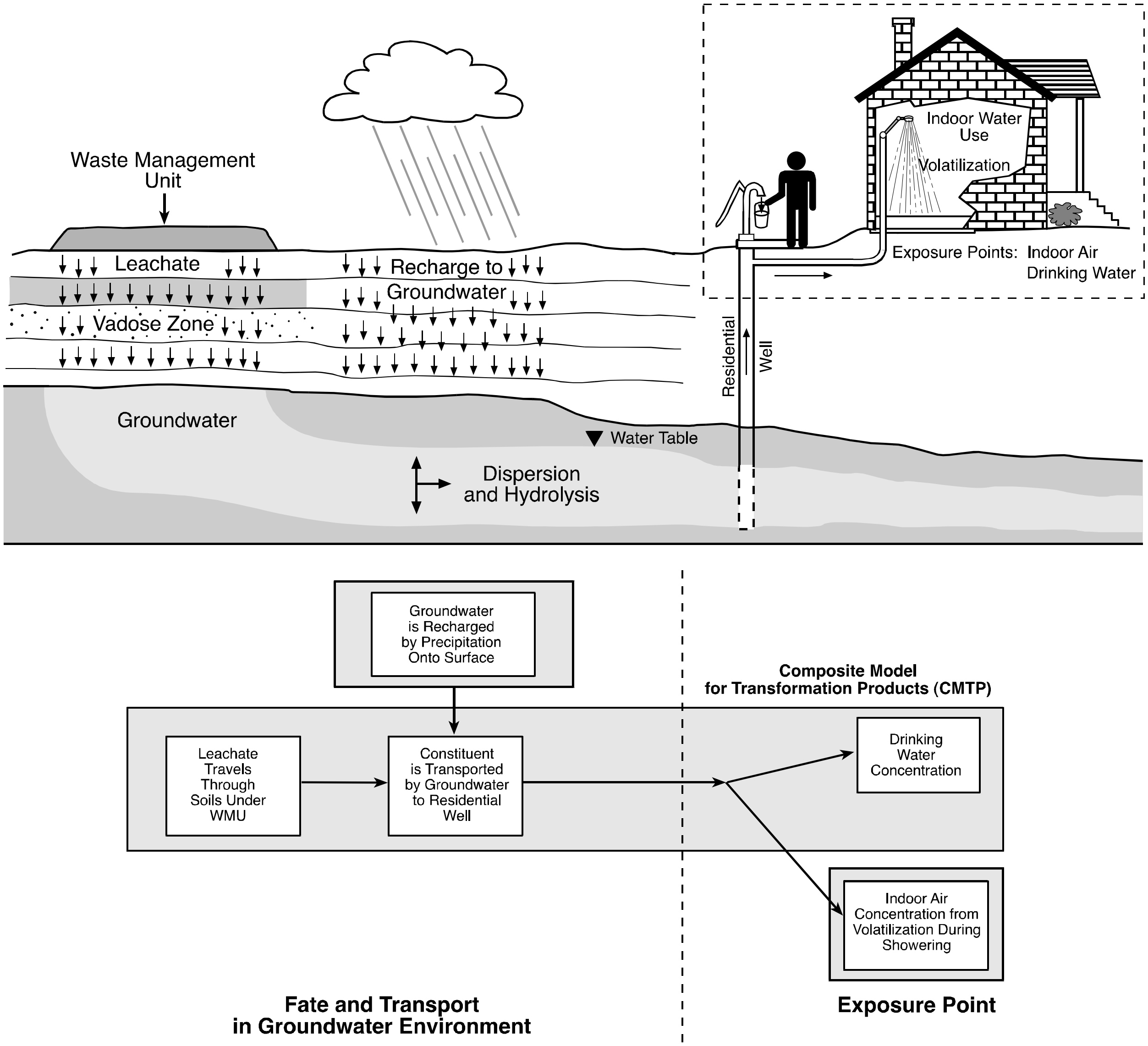

Figure 5-5. Groundwater fate and transport processes and exposure points.

drawn for domestic use. The vadose zone comprises the unsaturated soil layer between the bottom of the WMU and the saturated groundwater table. A groundwater model calculates two distinct phases of movement: first, through the vadose zone and, second, through the saturated water table. For the groundwater pathway, exposure point concentrations were determined in

- Domestic drinking water drawn from a residential well
- Indoor air resulting from domestic use of contaminated groundwater (e.g., showering).

The groundwater model estimates the movement of constituents through the vadose zone in which constituents move directly downward until they reach the groundwater table. Then, transport of constituents from the WMU to a residential well located downgradient of the WMU is modeled. Modeling constituent movement through the saturated groundwater takes into account transport and dispersion of constituent by the water moving through the saturated groundwater table. Time of transport through the groundwater aquifer may be hundreds of years, during which chemical processes (e.g., hydrolysis) also break down the constituent. The result of these calculations is a dilution attenuation factor (DAF) for chemical constituents. The DAF is the ratio of constituent concentration in the leachate released from the WMU to constituent concentration in the groundwater at the location of the residential well. These DAFs were used to relate constituent concentration at a residential well to leachate concentrations at the landfill and surface impoundment. The well water concentration was used to evaluate residential drinking water exposure and indoor air exposure due to indoor water use (e.g., showering). The groundwater pathway was evaluated separately from the air pathway in this assessment because the time frame and receptor locations for these two routes of exposure do not necessarily coincide.

5.2.2 Dispersion and Deposition Modeling

Dispersion modeling is a computer-based set of calculations used to estimate ambient ground-level constituent concentrations associated with constituent releases from a WMU. The dispersion model uses information on meteorology (e.g., windspeed and direction, temperature) to estimate the movement of constituents through the atmosphere. Movement downwind is largely determined by windspeed and direction. Dispersion around the centerline of the contaminant plume is estimated by empirically derived dispersion coefficients that account for movement of constituents in the horizontal and vertical directions. In addition, constituent movement from the atmosphere to the ground is also modeled to account for deposition processes driven by gravitational settling and removal by precipitation.

The air dispersion and deposition modeling conducted for this analysis produced output files of data that were used to calculate environmental media concentrations and food chain concentrations (see Section 5.3). The dispersion model outputs included air concentration of vapors and particles, wet deposition of vapors and particles, and dry deposition of particles. Dry deposition of vapors was also calculated but outside the dispersion model.

5.2.2.1 Industrial Source Complex Short Term Dispersion Model. A number of dispersion models are available for estimating the transport of constituent through the atmosphere, several of which are available on EPA's Support Center for Regulatory Air Models (SCRAM) Bulletin Board (http://www.epa.gov/scram001/). These dispersion models were developed for a variety of applications and each has its own strengths and weaknesses. The Industrial Source Complex Short Term, Version 3 (ISCST3) model was selected for air dispersion modeling in this analysis. Because this assessment required a model with the capability to model area sources, ground-level and elevated sources, ambient air concentrations and deposition fluxes, vapors and particulates, and annual averaging times, ISCST3 was an appropriate model to use. In addition, ISCST3 has been supported by EPA's Office of Air Quality Planning and Standards and has been used extensively in regulatory applications.

ISCST3 (U.S. EPA, 1999a), a recommended dispersion model in EPA's *Guideline on Air Quality Models* (U.S. EPA, 1999c), is a steady-state, Gaussian plume dispersion model. A steady-state model is one in which the model inputs and outputs are constant with respect to time. That is, the system being modeled is assumed to be unchanging over time. The term Gaussian plume refers to the kind of mathematical solution used to solve the air dispersion equations. It essentially means that the constituent concentration is dispersed within the plume laterally and vertically according to a Gaussian distribution, which is similar to a normal distribution. These assumptions and solutions hold for each hour modeled. The results for each hour are then processed to provide values for different averaging times depending on the user's needs (e.g., annual average).

ISCST3 is capable of simulating dispersion of pollutants from a variety of sources, including point, area, volume, and line sources. ISCST3 can account for both long- and short-term air concentration of particles and vapor and wet and dry deposition of particles and vapor. In addition to deposition, wet and dry plume depletion can be selected to account for removal of matter by deposition processes and to maintain mass balance. Receptors can be specified in polar or cartesian arrays or can be set to discrete points as needed. Flat or rolling terrain may be modeled but only flat terrain may be used for area sources. ISCST3 considers effects on dispersion of environmental setting by allowing the user to set urban or rural dispersion parameters.

5.2.2.2 Configuration of ISCST3 for Air Dispersion and Deposition Modeling. Results of air dispersion and deposition modeling represent the initial step in the fate and transport of vapor and particle emissions in the environment. The ISCST3 model was used to estimate the

- Air concentration of vapors
- Air concentration of particles
- Wet deposition of vapors and particles
- Dry deposition of particles onto soils.

Dry deposition of vapors was calculated outside of ISCST3 as explained below.

All air concentrations and deposition values developed by ISCST3 were unit values based on modeling default unit emission rates. For treatment tank modeling, the default unit emission applied was 1 g/s-m^2. This unit emission rate resulted in unit air concentrations measured in (µg/m^3) per (unit emission rate of 1 g/s-m^2) and deposition rates measured in (g/m^2) per (unit emission rate of 1 g/s-m^2). Due to the large range of surface areas being considered for landfills and surface impoundments, modeling of these units required the use of two different unit emission rates. That is, a unit emission rate of 1 µg/s-m^2 was applied for sources larger than 5,000 m^2, and 1 mg/s-m^2 was applied for sources 5,000 m^2 and smaller. External to ISCST3, the resulting unit concentrations and deposition rates were adjusted by applying the appropriate multipliers (1E+06 g/µg and 1E+03 g/mg). Later in the exposure modeling process, the unit air concentrations [(µg/m^3) per (unit emission rate of 1 g/s-m^2)] and deposition rates [(g/m^2) per (unit emission rate of 1 g/ s-m^2)] were multiplied by chemical-specific emission rates to produce values used to calculate environmental media concentrations.

Modeling was conducted using 5 years of data obtained from 49 representative meteorological stations throughout the country (see Section 4.3 for a discussion of meteorological site selection). Modeling was conducted for many separate scenarios designed to cover a broad range of WMU characteristics, including

- Ground-level and elevated sources
- 21 surface area sizes for landfills, 20 area sizes for surface impoundments, and 31 area-height combinations for tanks
- Distances ranging from 0 to 2,000 m from the WMU placed in 16 directions spaced every 22.5 degrees around the circumference of the WMU.

Air Concentrations of Vapor and Particles. ISCST3 estimates air concentrations of particles and vapors based on a number of variables including wet and dry deposition and plume depletion. The model accounts for downwind movement of the plume containing airborne vapors and particles. It also accounts for dispersion of vapors and particles around the center line of the plume as the plume travels in a downwind direction. Removal of constituent mass from the plume occurs due to wet and dry deposition. Wet and dry deposition are important processes in indirect exposure modeling because they account for the movement of constituent mass from the atmosphere to soil, water, and vegetation. Deposition is discussed below. There is, however, a closely related process, known as depletion, that affects the calculation of air concentrations.

Assumptions Made for ISCST3 Modeling

- Wet and dry depletion were activated in the dispersion modeling for particles. Wet depletion was considered for vapors.
- Area source was modeled for all WMUs.
- To minimize error due to site orientation, circular area sources centered on the origin were modeled.
- Modeling was conducted using unit emission rates.
- Receptors were placed on receptor rings ranging from 0 to 2,000 m starting from the edge of the source with 16 receptor points on each ring.
- The rural option was used in the ISCST3 modeling since the types of WMUs being assessed are typically in nonurban areas.
- Flat terrain was assumed.

Depletion is essentially the mirror of deposition. That is, while deposition accounts for the amount of constituent that moves to the ground, depletion accounts for the amount of mass removed from the atmosphere by deposition. The ISCST3 model allows the user to model depletion and deposition separately (i.e., the user may select depletion, deposition, or both depletion and deposition). When depletion is included, the mass deposited on the ground due to wet or dry deposition is removed from the plume, thereby conserving mass (U.S. EPA, 1995d). In this analysis, air concentration of particles was modeled with both wet and dry depletion activated. For vapors, ISCST3 was used to model only wet deposition and depletion, and dry deposition of vapors was calculated outside the model. As a result of calculating dry deposition/depletion of vapors outside ISCST3, the mass balance for vapors is not maintained and there is uncertainty introduced into the air modeling calculation, which would tend to overpredict vapor air concentrations. To consider dry deposition and depletion of vapors using ISCST3, it would have been necessary to provide chemical-specific gas deposition information, which is not readily available. Furthermore, it would have been necessary to conduct chemical-specific modeling, significantly increasing the number of air dispersion model runs needed.

Wet Deposition of Particles and Vapor. Wet deposition is the deposition of material on a surface from a plume as a result of precipitation. The amount of material removed by wet deposition from the plume is a function of the scavenging rate coefficient, which is based on particle size (U.S. EPA, 1995d). To perform these calculations, wet deposition, wet depletion, and dry depletion were all selected in the input run-stream file. Precipitation data from the Solar and Meteorological Surface Observation Network (SAMSON) CD-ROM, (U.S. DOC and U.S. DOE, 1993) were required to process the meteorological inputs for this analysis.

Dry Deposition of Particles. Dry deposition refers to the deposition of material on a surface (e.g., ground, vegetation) from a plume of material as a result of processes such as gravitational settling, turbulent diffusion, and molecular diffusion. Dry deposition is calculated as the product of air concentration and dry deposition velocity. To calculate dry deposition, ISCST3 requires mass mean diameter, particle density, and mass fraction to be input into the source pathway for deposition calculations (U.S. EPA, 1995b). Dry deposition calculations also require the meteorological input file to contain surface friction velocity, hourly Monin-Obukhov length, and surface roughness length. Surface friction velocity and hourly Monin-Obukhov length were calculated in the PCRAMMET preprocessor (U.S. EPA, 1995c). More detail on the PCRAMMET preprocessor is provided in Appendix N.

Dry Deposition of Vapors. Dry deposition of vapors was calculated using a step external to the ISCST3 model because chemical-specific dry deposition modeling within ISCST3 was precluded by time and resource considerations. Using a dry deposition algorithm for particles (from the ISCST user's manual), dry deposition of vapor was calculated by multiplying the vapor air concentration by a default deposition velocity of 0.2 cm/s (Koester and Hites, 1992). This approach assumes that vapors behave as fine aerosols and, therefore, are amenable to modeling using the dry deposition algorithm for particles.[1]

To calculate the weighted dry deposition velocity, land use was obtained from 1:250,000-scale quadrangles of land use and GIRAS spatial data obtained from the EPA website and placed in an ARC-INFO format (U.S. EPA, 1994). Land use was based on data from the mid-1970s to the early 1980s. The fraction of time in each stability class was based on 5-year hourly meteorological files used in ISCST3 modeling.

Averaging Time. For the paints listing risk assessment, all human health and ecological risks were evaluated based on benchmarks for chronic, long-term exposure. Therefore, the air concentrations and deposition values required for the human health and ecological risk assessment were long-term averages. Long-term averages calculated by the ISCST3 model were annual averages. However, since the ISCST3 model was run using 5 years of meteorological data, it actually averages the hourly concentrations over the entire 5-year period.

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[1] Koester and Hites (1992) suggest that the dry deposition of vapor may be negligible.

Rural vs. Urban. The rural vs. urban setting in ISCST3 allows the user to account for differences between rural and urban environments. In urban environments, the built environment (e.g., buildings, roads, and parking lots) alters the dispersion character of the atmosphere, particularly at night due to building-induced turbulence and reduced nighttime cooling. Thus there is greater nighttime mixing of constituents in urban areas compared to rural areas. For purposes of ISCST3 modeling, the urban classification applies mainly to large cities; even small cities and suburban areas are classified as rural for ISCST3 purposes. For this analysis, the rural setting was used because, although the specific location of the WMUs was not known, they are not likely to be located within large cities.

Placements of Points Where Air Concentrations Were Calculated. A grid of points where air concentration and deposition values were calculated was established using a polar grid. Air concentration and deposition values were produced for each point on the grid (i.e., x, y coordinate) in a polar array consisting of 16 radials spaced every 22.50 degrees and rings at distances ranging from 0 to 2,000 m from the edge of the WMU.

Flat vs. Elevated Terrain. The ISCST3 model allows the user to model to account for elevated terrain by specifying an elevation for each point on the grid where air concentrations and deposition values are calculated. This feature, however, is not available for use with area sources. Because all sources modeled in this analysis were area sources, elevated terrain was not considered.

TOXIC vs. Regulatory Mode. The most recent version of ISCST3 (99155, U.S. EPA, 1999a) allows the user to select a regulatory default option or to select a TOXICS option. The regulatory default option uses Romberg numeric integration solution to estimate air concentration from an area source. Based on the results of validation tests performed by EPA, EPA has concluded that the Romberg algorithm performs very well in terms of efficiency and reasonableness (U.S. EPA, 1992d). However, this algorithm takes a significant amount of time to execute for large area sources. To improve model run times, the TOXICS option was added by EPA to the area source model. The TOXICS option also uses a Romberg numeric integration solution to estimate air concentrations and deposition rates near the WMU. Farther from the WMU, however, the TOXICS option uses a two-point Gaussian Quadrature routine instead of the Romberg solution to estimate air concentration and deposition. The two-point Gaussian Quadrature solution is computationally more efficient, which accounts for the shorter model run time. For this study, a sensitivity analysis was conducted to compare the estimated air concentrations calculated using the regulatory option to those calculated using the TOXICS mode. This analysis showed small differences between results obtained using either option (see Appendix N). Given the benefit of reduced run times, the TOXICS option was selected in this analysis.

Source Shape. All WMU types modeled in this analysis were modeled as area sources. Landfills and surface impoundments were modeled as ground-level sources, and treatment tanks were modeled as elevated sources. The ISCST3 model allows the user to model area sources as polygonal sources with from 3 to 20 sides (U.S. EPA, 1999a). The ISCST3 was set up in this analysis to model an area source as a 20-sided polygon shaped to approximate a circle. This option was chosen because there are no actual data on the shape of sources and a circular source minimizes air modeling errors caused by source shapes and orientation.

5.2.2.3 <u>Preparing ISCST3 Input Files</u>. Two types of input files are required to run ISCST3, the run-stream file and the meteorological file. The run-stream file is an ASCII file that contains the model option settings, source parameters, and receptor locations. The meteorological file contains hourly values of windspeed, wind direction, stability class, mixing height, ambient air temperature, and precipitation type and amount.

ISCST Run-Stream Files. The ISCST3 run-stream file is composed of six pathways, which drive different model functions. They are the Control Pathway, Source Pathway, Receptor Pathway, Meteorology Pathway, Terrain Grid Pathway, and Output Pathway. Each of these pathways is described in Appendix N. The Terrain Grid Pathway is not presented since it is used only with point sources (i.e., for facilities with stacks), which were not modeled in this analysis.

Meteorological Files. The meteorological file was generated using the meteorological preprocessor PCRAMMET (U.S. EPA, 1995c). The preprocessor pairs hourly surface observations with upper air measurements. For each of the 49 meteorological stations modeled, 5 years of surface and upper air data were used. The preprocessor creates a file in binary format that contains hourly windspeed, wind direction, atmospheric stability class, temperature, and mixing height. Land use data also were required by PCRAMMET in the vicinity of each meteorological station to derive air model inputs such as Bowen ratio, surface roughness height, minimum Monin-Obukhov length, noontime albedo, and the fraction of net radiation absorbed by the ground. Appendix N discusses the derivation of each of these model inputs.

Key Meteorological Data for the ISCST3 Model

Wind Direction: Determines the direction of the greatest impacts.

Windspeed: Ground-level air concentration is inversely proportional to windspeed, so the lower the windspeed the higher the concentration.

Stability Class: Impacts rate of lateral and vertical diffusion. The more unstable the air, the greater the diffusion.

Mixing Height: Determines the height to which chemical constituents can be diffused vertically.

Surface and Upper Air Data. Hourly surface meteorological data were obtained from the SAMSON CD-ROM (U.S. DOC and U.S. DOE, 1993). The variables include temperature, pressure, wind direction, windspeed, opaque cloud cover, ceiling height (a measure of the height of the cloud cover), current weather, and hourly precipitation. Verification of the meteorological data sources showed that SAMSON precipitation data were not adequate. That is, long-term average precipitation amounts calculated from SAMSON fell significantly short of the long-term values provided in the international station meteorological climate summaries (ISMCS; USN et al., 1992). Because SAMSON precipitation data were found to be inadequate, precipitation data were obtained from cooperative station daily summaries (NCDC et al., 1995). A program, which uses the available SAMSON data as a template, was developed to allocate these daily data to hourly time series required for this analysis.

Mixing heights were calculated using surface and upper air station data. The upper air data were compiled from the Radiosonde Data of North America CD-ROM (NCDC, 1997). Mixing height is the height to which vertical dispersion of constituent can occur.

Filling Missing Data. Missing surface data were identified using a computer program to search for incidents of missing data on the observation indicator, opaque cloud cover, temperature, station pressure, wind direction and speed, and ceiling height. Missing surface data were filled in by another computer program. This program fills in up to 5 consecutive hours of data for cloud cover, ceiling height, temperature, pressure, wind direction, and windspeed. For single missing values, the program follows the objective procedures developed by Atkinson and Lee (1992). For two to five consecutive missing values, other rules were developed because the subjective methods provided by Atkinson and Lee (1992) rely on professional judgment and could not be programmed. The program flagged files where missing data exceeded five consecutive values. In the few cases where this occurred and the missing data did not constitute 10 percent of the file, they were provided manually according to procedures set forth in Atkinson and Lee (1992). Years that were missing 10 percent or more of the data were discarded (Atkinson and Lee, 1992). If a meteorological station did not have 5 years with sufficient data completeness, the station was discarded and another station was selected.

5.2.2.4 <u>Source Areas Modeled</u>. In the modeling analysis, three types of WMUs were considered (i.e., landfill, treatment tanks, and surface impoundments). Because the ISCST3 model is sensitive to the size of the area source, the relationship between air concentrations and size of the area source was analyzed. As illustrated in Figure 5-6, the results show that, for relatively small area sources, air concentrations increase significantly as the size of the area source increases. For large area sources, this increase in air concentrations is not as significant.

To address this model sensitivity yet avoid modeling over 2,800 separate WMUs, area-based strata that represent the distribution of surface areas for each of the WMU types were developed. Landfills and surface impoundment units were modeled as ground-level area sources, and tanks were modeled as elevated area sources. Separate strata were developed for each WMU type. The sampling of WMUs and the creation of strata to represent the distribution of dimensions found in the underlying databases are discussed in Section 4.4. In that section, the source areas and heights used in the modeling analysis are presented.

For each WMU type, the median area for each strata were input into ISCST and modeled for each of the 49 meteorological stations. For tanks, each area-height combination was modeled for each of the 49 meteorological locations. This provided a set of unit air concentrations for use in the analysis. For any specific WMU, the median unit air concentration and deposition values for the strata were used to represent all WMUs from that strata. The strata were designed to be narrower for the smaller areas because this is where air concentrations have the greatest rate of change with respect to change in area. This minimizes the error associated with not modeling each individual WMU.

5.2.2.5 <u>Spatial Averaging of Air Concentrations and Deposition</u>. A GIS model was used to calculate air concentration and deposition rates for the resident location, buffer area, agricultural field, and waterbodies. This crucial step combines the spatial characterization of the

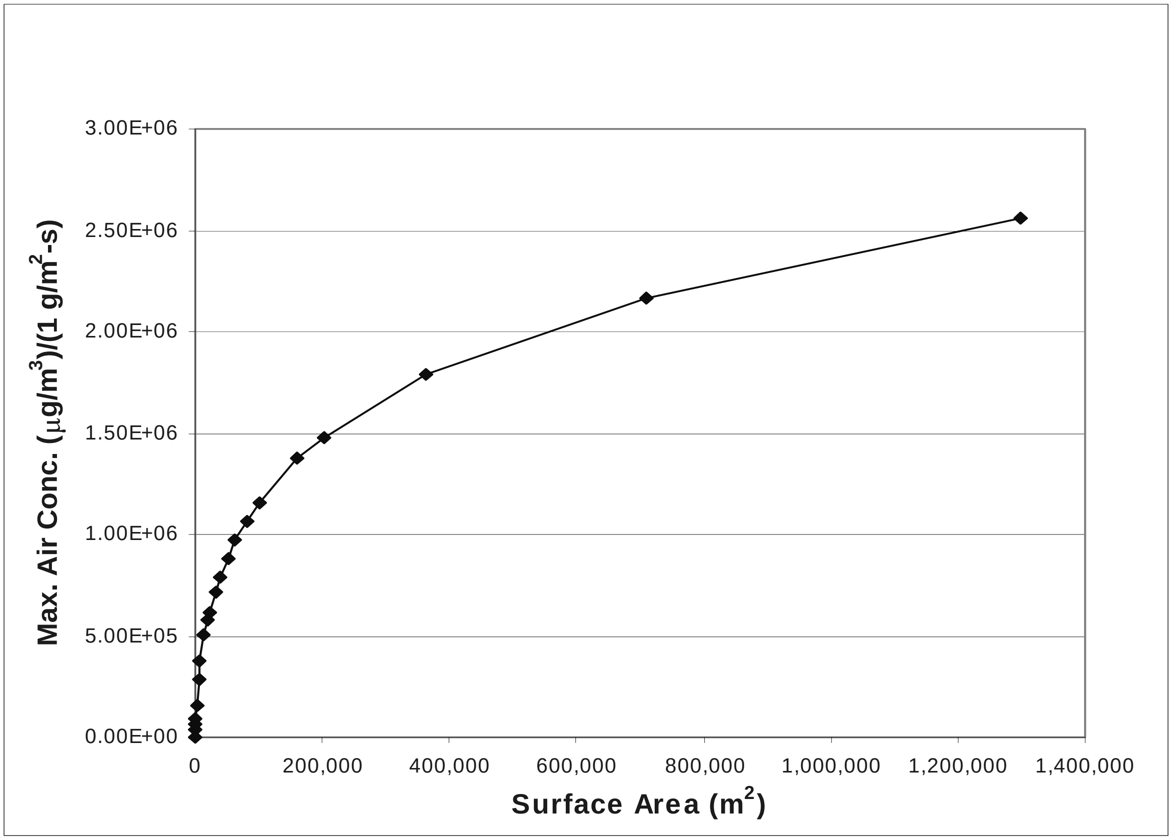


Figure 5-6. Air concentration vs. size of area source.

resident location, buffer area, agricultural field, and waterbody in the site layout with air modeling outputs for each WMU/meteorological location/WMU surface area combination.

In an automated batch program, the ASCII files produced by ISCST3 were converted from a polar array of values into an evenly spaced grid of concentration values distributed around the center of the site layout in the form of a GIS point coverage. To calculate the point estimate for the resident location, the program estimates the air concentration of vapors and particles from this GIS point coverage by selecting the grid point nearest the resident location. These values are used directly to determine human inhalation exposures.

To calculate the spatial averages for the buffer area, agricultural field, and waterbody polygons, the program individually overlaid these areas with this point coverage and averaged the overlapping points. These mean concentration values and their associated identifiers are the output of the program and represent the average air concentrations and deposition values used in subsequent modeling steps to predict soils, water, and food chain concentrations.

5.2.3 Estimation of Soil and Water Concentrations

This section describes the components that make up the watershed and waterbody model. The model consists of a set of equations that predict the movement of constituents in the soil and a set of equations that determine the fate of constituents in a surface waterbody. The output of these models provides the concentration of the constituent for the

- First 20 cm of soil
- First 1 cm of soil
- Dissolved water portion of the waterbody
- Suspended solids portion of the waterbody
- Benthic sediment portion of the waterbody
- Benthic sediment pore water portion of the waterbody.

Discussions in this section are general in nature. Two appendixes support the discussions with more detailed information: Appendix M contains the full set of equations used to calculate media concentrations, and Appendix F lists the physical/chemical properties used; the parameter values selected for fate, transport, and exposure modeling; and citations for the parameter values selected.

5.2.3.1 **Predicting Soil Concentrations.** Soil concentrations in a watershed were calculated for both the buffer area and agricultural field. For this analysis, it is assumed that constituents deposited on buffer area soils adjacent to the WMU via air dispersion and deposition are subject to transport by erosion and runoff to the agricultural field. The concentration of a constituent in soil is a function of what has been deposited onto the soil from the air and what has been transported overland by erosion and runoff from adjacent land areas minus soil losses. Soil losses accounted for in this analysis include biodegradation, volatilization, leaching, surface runoff, and erosion. The following sections describe how each of these addition and removal mechanisms are estimated to obtain an overall soil concentration.

Characterization of location-specific meteorological factors (e.g., precipitation, irrigation, runoff, and evapotranspiration) and soils is important in predicting soil concentrations. Location-dependent meteorological factors and soil characteristics were taken from several sources and are described in Section 4.3.

Addition of Constituents to Soils. Addition of constituents to soils, also called constituent loading, can result from atmospheric deposition and overland movement of constituents. Constituent loadings to the soil were calculated using equations presented in Tables M.2-2 and M.2-3 in Appendix M.

The primary loading mechanism is by wet and dry deposition of constituents from the air to the ground. Constituent deposition was predicted based on the average air concentration and deposition flux for both the buffer area and the agricultural field. These mechanisms include wet deposition of vapors and particles and dry deposition of vapors and particles.

There is no direct overland movement of constituents from the WMU to the buffer or agricultural field. However, loading due to overland transport from the buffer to the agricultural field can occur given that the buffer is contaminated due to direct deposition. The same erosion and runoff terms used to estimate loading from soils into the waterbodies are used to estimate loading from the buffer onto the agricultural field.

Soil concentration due to loading from both overland transport and direct deposition is dependent on the depth of the soil column into which the constituent is mixed. The mixing depth in the soil (z) depends on whether the soil is disturbed by agricultural tilling. In this analysis, 20 cm was used as the mixing depth for tilled soils and 1 cm was used for untilled soils (U.S. EPA, 1993). The soil bulk density converts a deposition rate from grams of contaminant per unit area to grams of contaminant per kilogram of soil.

Losses from Soils. Constituent losses from the soil were calculated using equations presented in Tables M.2-4 through M.2-8 in Appendix M. Soil concentrations are calculated using a total loss term that represents total losses due to five separate processes:

- Leaching of the chemical into the groundwater due to precipitation
- Erosion of the chemical laterally along with the soil due to water
- Runoff of the dissolved chemical with the lateral flow of water
- Biodegradation of the chemical in situ
- Volatilization losses of the chemical.

The total loss term provides one overall loss constant that is applied in the calculation of soil concentrations. The loss constant is applied over the same time period of constituent loading (i.e., the time period of deposition) during the operating life of the WMU. Each component of the total loss term is discussed below.

The leaching loss constant representing the rate at which constituents leach from soil is a chemical-specific value that is a function of location-specific precipitation, irrigation, runoff, and evapotranspiration. The equation for calculating the leaching loss constant, ksl, is as shown in Table M.2-5 in Appendix M.

The Universal Soil Loss Equation was used to estimate soil erosion losses (X_e) as shown in Table M.2-8 in Appendix M. USLE is an empirically derived equation originally developed by the Soil Conservation Service of the U.S. Department of Agriculture to estimate soil erosion losses from agricultural fields during soil conservation planning. The USLE is applied in the context of the Gross Erosion Sediment-Delivery Ratio method outlined in USDA (1978) and described in greater detail in the SCS *National Engineering Handbook* (USDA, 1971). Gross erosion is defined as the summation of erosion from all sources within a watershed, as estimated for sheet and rill erosion by USLE. The sediment delivery ratio adjusts gross erosion rates to account for terrain and cover features, which effectively reduce sediment erosion.

The loss constant due to surface runoff was calculated as a function of chemical-specific values and location-specific average annual runoff, as shown in Table M.2-10 in Appendix M.

In this analysis, biodegradation losses from soil were considered for all organic chemicals. The vast majority of soil biodegradation rates were taken from *Handbook of Environmental Degradation Rates* (Howard et al., 1991). The Handbook provides chemical-specific values for the half-life in soil. The highest half-life was used to derive degradation rate since a higher half-life results in a lower degradation rate. A single biodegradation rate was used for each constituent. This biodegradation rate was applied to all WMUs in all locations. Site-specific conditions may cause the biodegradation rate to be greater or less than the default value used, but this variability was not captured by this analysis.

The volatilization loss constant calculates the contaminant loss due to volatilization from soil. The calculation of the volatilization loss factor, ksv, is shown in Table M.2-11 in Appendix M. Volatilization kinetics are driven by chemical-specific properties and environmental conditions, including location-specific properties. Constituent properties that influence volatilization include Henry's law constant, the soil-water partitioning coefficient, and the diffusivity in air. Volatilization is further influenced by environmental conditions including soil mixing depth, soil bulk density, ambient air temperature, average annual windspeed, and impacted surface area. In this assessment, location-specific values were used for the ambient air temperature and the average annual windspeed.

Calculation of Soil Concentrations. Following calculation of the constituent loading rates and the rate loss constants, soil concentrations were estimated. Equations presented in Table M.2-1 in Appendix M were used to estimate soil concentrations.

Soil constituent concentration changes with each year of operation of a facility. During the 30-year period for landfills or 50-year period for tanks and surface impoundments in which a WMU is assumed to operate, the constituent concentration in soil will steadily increase. To account for this phenomenon, the equation to calculate average soil concentration over the time period of deposition (facility operation) explicitly considers the time period of exposure for the receptor of interest. This is accomplished by accounting for total loading and loss for each year and then determining an average soil concentration.

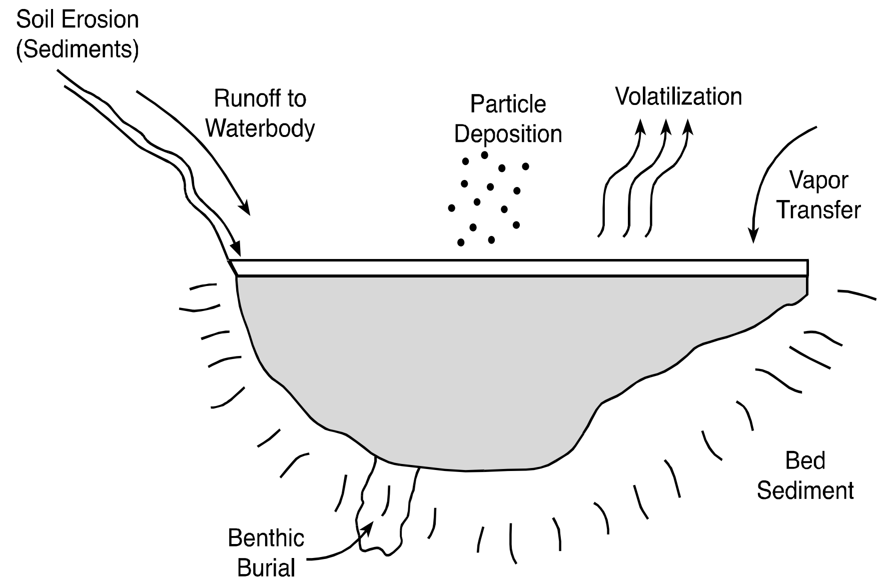
For evaluating cancer risks, soil concentrations were explicitly averaged over the exposure period for the specific human receptor. This was accomplished using two time measures: the time period over which deposition occurs (i.e., the operating period of the WMU) and exposure duration. For exposure duration less than the operating period of the WMU, the average soil concentration is determined by the average of the last years of operation where the number of years integrated is equal to the exposure duration. During this period, soil concentration is steadily increasing. For exposure durations greater than the operating period of the WMU, the soil concentrations are integrated over two periods. First, for the period spanning the entire operating period of the WMU, soil concentrations are steadily increasing. Second, for the period of time after the operating period of the WMU, soil concentration will be steadily declining. Since exposure duration is limited to 50 years, the scenarios in which the exposure duration is greater than the operating period of the WMU can only occur for landfills.

To evaluate noncancer effects from chemicals, soil concentrations were used for the time corresponding to the end of the WMU operating period (i.e., 30 or 50 years). This is because less-than-lifetime average exposures are of interest for evaluating the potential for noncancer effects, such as developmental toxicity. Many chemicals will have reached steady-state well before year 30 (or 50 depending on the WMU type). However, other chemicals will tend to build up in soils for much longer periods. Limiting the soil concentrations to the operating period of the WMU represents a balance between the period of facility operating (which is uncertain) and a shorter time period (which could underestimate exposure from soil). Ecotoxicological risks were evaluated based on the same soil concentrations used to evaluate human noncancer risks (see Section 9.0).

5.2.3.2 <u>Predicting Surface Water Concentrations</u>. The waterbody in this analysis is a stream located downgradient of the WMU. Depending on the scenario that is described in Section 4.2, the stream is either adjacent to the buffer area (the area that separates the WMU from the human receptor locations) or is located adjacent to the agricultural field on the side farthest from the WMU. For modeling purposes, the stream is shaped as a rectangle 5.5 m wide and as long as the width of the agricultural field. The assumption that the stream is 5.5 m was used because this width is typical of a third-order fishable stream (van der Leeden et al., 1990). A third-order stream refers to a type of stream segment classification. In this classification scheme, a first-order stream segment is one with no tributaries. That is, a first-order stream segment receives all of its flow from runoff from the surrounding watershed soils. A second-order stream segment is produced when two first-order stream segments come together. A third-order stream segment occurs when two second-order segments come together. (A third-order stream segment is not produced when a second-order and a first-order stream segment combine.) The third-order steam segment, therefore, has the combined flow of at least two second-order stream segments. The third-order stream was selected because it reasonably represents the smallest waterbody that would routinely support recreational fishing of consumable fish. The stream segment modeled in this assessment is assumed to be homogeneously mixed with a depth of 0.21 m (including water column and benthic sediment) and a flow rate/velocity of 0.5 m/s (van der Leeden et al., 1990).

Figure 5-7 illustrates the method used to calculate average constituent concentrations for the waterbodies used for fishing and for agriculture stock water. Constituents can enter the waterbody by one of four pathways. Constituents in the air above the waterbody can be deposited directly onto the waterbody's surface. This occurs for airborne particles via wet and dry deposition due to gravitational settling and scavenging by precipitation (i.e., wet and dry deposition). Vapors can also deposit directly onto the waterbody's surface via scavenging by precipitation (i.e., wet deposition). Constituents on the soils surrounding the waterbody can enter the waterbody through runoff and erosion. Constituents that dissolve into water in the soil can be transported via overland water runoff to the waterbody. Constituents that sorb onto soil particles can be transported via overland erosion of soil to the waterbody.

Once in the waterbody, constituents are assumed to be uniformly mixed in a single stream segment. There is water flow in and out of the steam segment. Water flowing into the waterbody is assumed to have zero constituent concentration and water flowing out contains constituent concentrations resulting from WMU contamination. Within the waterbody, the model accounts for loss of constituent from the water column via volatilization to the air above the waterbody surface. It also accounts for loss of constituent to the benthic sediment, that is, constituents that are buried deep enough to no longer be subject to resuspension into the waterbody.



Source: U.S. EPA, 1988.

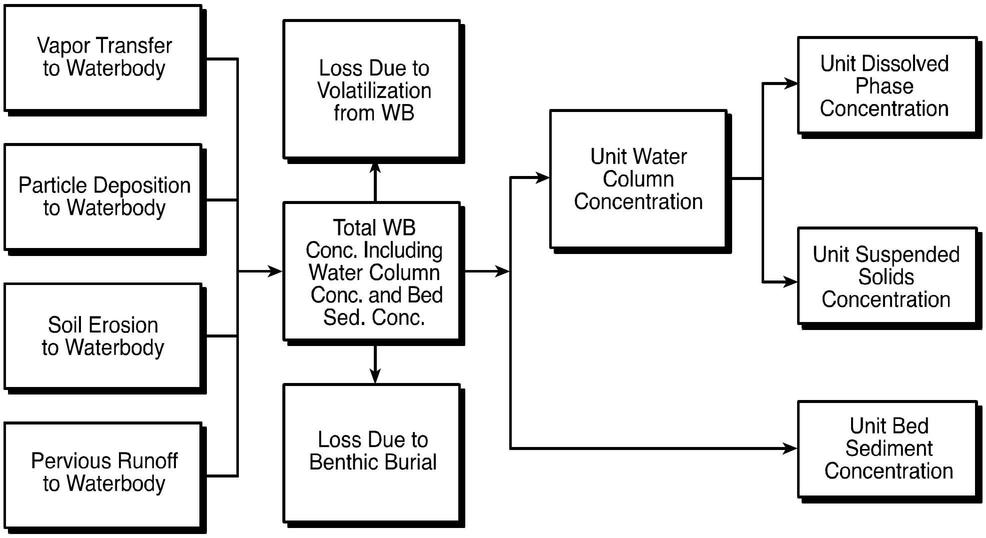


Figure 5-7. Waterbody constituent loading and loss processes.

The total mass of constituent in the waterbody is divided into four compartments:

- Constituents in dissolved water
- Constituents sorbed onto suspended solids
- Constituents sorbed onto sediment particles
- Constituents dissolved in pore water in the sediment layer.

Dissolved water concentration of constituents accounts for constituent mass that dissolved in water and is contained in the flowing water in the waterbody. Suspended solids concentration of constituents accounts for constituent mass that is sorbed onto particles that are suspended in the flowing water in the waterbody. Sediment constituent concentrations account for constituent mass that is sorbed onto particles that settle to the sediment layer at the bottom of the waterbody. Pore water constituent concentrations account for constituent mass that is dissolved in the water contained in the pore spaces in the sediment layer.

Humans and wildlife are exposed to constituents in these various waterbody compartments depending on the route of exposure. For example, one waterbody is used for drinking water for beef cattle and dairy cows. Because these animals drink water directly from the waterbody (i.e., the water is not filtered), their exposure to constituents from drinking stock water comes from both dissolved and suspended constituent concentrations. Section 5.3 describes the calculation of farm food chain and aquatic food chain concentrations in greater detail. Section 9.0 describes ecological exposures that, in part, are based on waterbody concentrations.

Loading to a Waterbody. There are four ways a constituent may enter the waterbody:

- Constituents may be bound to airborne particles that are deposited on the waterbody surface (e.g., wet and dry deposition of particles)
- Vapor phase constituents in the air may transfer to the surface waterbody (e.g., wet and dry deposition of vapors)
- Constituents may be dissolved in runoff that enters the surface waterbody
- Constituents may be sorbed to eroded soil particles that enter the surface waterbody.

These concentrations were predicted using the results of air modeling and soil calculations described in Sections 5.2.2 and 5.3.1, respectively. Equations describing how the model calculates these loadings are provided in Tables M.3-4 through M.3-8 in Appendix M.

Constituent Losses from Surface Waterbody. Constituent losses due to volatilization and burial in benthic sediments were considered. After accounting for the losses, the total waterbody concentration was apportioned to the water column and bed sediments irrespective of the loss mechanism. Therefore, at equilibrium, both loss mechanisms affect both compartments to the same extent. The two divisions of the total waterbody concentration (into the water column concentration and the bed sediment concentration) were further divided chemically between the fraction that is sorbed to sediments and suspended solids and the fraction that is dissolved. Equations describing how the model calculates these losses are provided in Tables M.3-9 through M.3-11 in Appendix M.

Calculating Surface Water and Sediment Concentrations. Following calculation of the constituent loading and loss rates, waterbody constituent concentrations were estimated using equations presented in Table M.3-3 in Appendix M.

As discussed in Section 5.2.3.1, watershed soil constituent concentration changes with each year of operation of a facility. To account for this phenomenon, soil concentrations were calculated over the time period of deposition (facility operation), and the time period of exposure for the receptor of interest was explicitly considered. This time-averaged soil concentration is reflected in the calculation of waterbody concentrations. The waterbody model is a steady-state model; that is, all inputs and outputs from the waterbody system are assumed to be unchanging with time. Therefore, the soil concentrations were calculated outside the waterbody model and input to the model as constant rates reflecting different exposure scenarios and receptors.

For evaluating cancer risks, waterbody loadings were calculated using soil concentrations that were explicitly averaged over the exposure period for the specific human receptor as discussed in Section 5.2.3.1. To evaluate noncancer effects from chemicals, waterbody concentrations were calculated using soil concentrations calculated for the time period corresponding to the end of the WMU operating period (i.e., 30 years for landfills or 50 years for treatment tanks and surface impoundments) because less-than-lifetime average exposures are of interest for evaluating the potential for noncancer effects such as developmental toxicity.

Ecotoxicological risks were evaluated based on the same waterbody concentrations used to evaluate human noncancer risks (see Section 9.0).

5.2.3.3 <u>Predicting Media Concentrations for Mercury</u>. Mercury may be present in the environment in multiple oxidation states. Elemental mercury (Hg^0) and divalent mercury (Hg^{2+}) are the predominant species found in natural environments. Data specifying the fraction of elemental mercury to divalent mercury in the source and environmental compartments were not available; hence, two separate analyses were completed for elemental and divalent mercury. To simplify the modeling, no transformations to other species in the WMUs were considered for either of the two mercury species.

For elemental mercury, 100 percent of the emissions were considered to be vapor, and inhalation was the only exposure pathway considered in the analysis. Inhalation exposure to receptors from landfills was assumed to occur during the 30 years of operating life, whereas inhalation exposure from surface impoundments and tanks was assumed to occur during the 50 years of operating life.

Divalent mercury emissions from the landfill were partitioned between vapor and particulate phases based on source model results. Once deposited to the soil, mercury may respeciate based on soil chemistry. According to EPA's *Mercury Report to Congress* (U.S. EPA, 1997b), it can be assumed that 98 percent of the mercury is present in the inorganic phase (either as divalent or elemental mercury) and 2 percent of the mercury is present in the organic phase (as methylmercury). In this analysis, the modeling was simplified so that all mercury present in soil was assumed to be divalent.

Divalent mercury that reached the waterbody was speciated between methylmercury and divalent mercury. In the *Mercury Report to Congress* (U.S. EPA, 1997b), EPA recommends that 15 percent of mercury in a waterbody be assumed to be methylmercury and 85 percent be assumed to be divalent mercury. For the fish consumption pathway, this assumption was used.

Bioaccumulation factors derived for methylmercury were applied to the methylmercury dissolved water concentrations and used to calculate concentrations of methylmercury in fish.

5.2.4 Groundwater Concentrations

The groundwater pathway was modeled for this analysis to determine the residential drinking water well concentrations resulting from the release of waste constituents from the WMU. The release of a constituent occurs when liquid that percolates through the WMU generates leachate, which can infiltrate from the bottom of the WMU into the subsurface. For landfills, the liquid percolating through the landfill is from precipitation and water in the waste. For surface impoundments, the liquid is primarily the wastewater managed in the impoundments.

The waste constituents dissolved in the leachate are transported via aqueous phase migration through the unsaturated zone (soil layer under the WMU) to the underlying saturated zone (i.e., groundwater) and then downgradient to a residential drinking water well. The exposure concentration is evaluated at the intake point of a hypothetical residential drinking water well located at a specified distance from the downgradient edge of the waste management unit. Hereafter, the residential drinking water well is referred to as the “receptor well.” This conceptual model of the groundwater fate and transport of contaminant releases from WMUs is illustrated in Figure 5-8.

The transport of leachate from the WMU through the unsaturated and saturated zone is quantitatively evaluated with the EPACMTP (U.S. EPA, 1996, 1997a). EPACMTP is used by EPA to make regulatory decisions for wastes managed in land disposal units (landfills, surface impoundments, wastepiles, or land application units) for a number of EPA hazardous waste regulatory efforts. EPACMTP simulates flow and transport of contaminants in the unsaturated zone and aquifer beneath a waste disposal unit to yield the concentration arriving at a specified receptor well location.

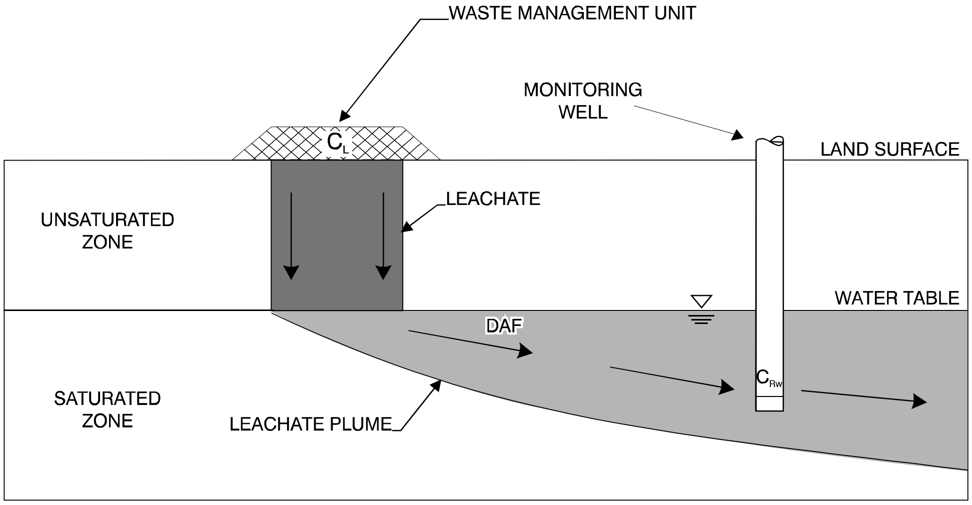


Figure 5-8. Schematic diagram of groundwater modeling scenario.

Fate and transport processes accounted for in the model are advection, hydrodynamic dispersion, equilibrium linear or nonlinear sorption, and transformation processes via chemical hydrolysis. In this analysis, partition coefficients developed from data compiled from the scientific literature were used. More information on constituent-specific partition coefficients is presented in Appendix H. 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. It is assumed that the soil and aquifer are uniform porous media and that flow and transport are described by Darcy's law and the advection-dispersion equation, respectively. EPACMTP does not account for preferential pathways such as fractures and macropores or facilitated transport, which may affect migration of strongly sorbing constituents such as metals.

The primary transport mechanisms in the subsurface are downward movement along with infiltrating water flow in the unsaturated zone and movement along with ambient groundwater flow in the saturated zone. The advective movement in the unsaturated zone is one-dimensional, while the saturated zone module accounts for three-dimensional flow and transport. The model also considers mixing due to hydrodynamic dispersion in both the unsaturated and saturated zones. In the unsaturated zone, flow is gravity-driven and prevails in the vertically downward direction. Therefore, the flow is modeled 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 negligible in the unsaturated zone. This assumption is based on the fact that lateral migration due to transverse dispersion is negligible compared with the horizontal dimensions of the WMUs. In addition, this assumption is environmentally protective because it allows the leading front of the constituent plume to arrive at the water table with greater peak concentration in the case of a finite source.

In the saturated zone, the movement of constituents is primarily driven by ambient groundwater flow, which in turn is controlled by a regional hydraulic gradient and hydraulic conductivity in the aquifer formation. The model does take into account the effects of infiltration from the waste source as well as regional recharge into the aquifer. The effect of infiltration from the waste source is to increase the groundwater flow in the horizontal transverse and vertical directions underneath and in the immediate vicinity of the waste source as may result from groundwater mounding. This three-dimensional flow pattern will enhance the horizontal and vertical spreading of the plume. The effect of regional recharge outside of the waste source is to cause a downward dip in the movement of the plume as it moves in the downgradient groundwater flow direction.

In addition to advective movement along with groundwater flow, the model simulates mixing of contaminants with groundwater due to hydrodynamic dispersion, which acts in the longitudinal, i.e, along the groundwater flow direction, as well as in horizontal and vertical transverse directions. The rate of movement of contaminants may be strongly affected by sorption reactions in both the unsaturated and saturated zone. The effect of sorption is expressed in a retardation factor, which is directly related to the magnitude of the constituent-specific K_d value (K_{oc} in the case of organics). Constituents with a zero or low K_d (or K_{oc}) value will have a retardation factor of 1, or close to it, which indicates that they will move at the same velocity as the groundwater, or close to it. Constituents with high K_d values, such as certain semivolatile organic constituents and many metals, will have high retardation factors and may move many times slower than groundwater. The relationship between K_d and retardation factor is expressed as

$$R = 1 + \frac{\rho_b \times K_d}{\varphi} \tag{5-1}$$

where

R = retardation factor
ρ_b = soil or aquifer bulk density (mg)
K_d = solid-water partition coefficient (g/cm^3)
φ = water content (in unsaturated zone) or porosity (in saturated zone).

The concentration of constituents in both the unsaturated and saturated zone may be further attenuated by biochemical transformation reactions. Although EPACMTP can, in principle, model a wide range of transformation processes as long as they can be represented as a first-order process, in practice a more conservative approach is adopted in which only degradation of organic constituents via hydrolysis transformation processes is simulated. For those constituents that may be transformed via hydrolysis into toxic daughter products, EPACMTP can account for the formation and subsequent fate and transport of these daughter products. The COCs considered in this analysis are those identified in paint wastes and do not include degradation (or daughter) products. The effect of degradation products on chemical concentrations was evaluated in a screening analysis (Appendix T). Two sets of deterministic runs were conducted for surrogate chemicals consisting of parent-daughter products. The concentration of daughter product was evaluated at receptor wells and found to be negligible.

Hydrolysis rates were calculated from chemical-specific hydrolysis rate constants using soil and aquifer temperature and pH values. Whereas chemical hydrolysis can be accurately modeled as a function of these factors, many other types of transformation processes, such as biodegradation, are much more site-specific and can be highly variable and therefore much more difficult to simulate using a generic model such as EPACMTP. EPA is therefore using a conservative assumption that these processes do not occur; that is, biodegradation is not accounted for. In the case of metals that may exist in multiple valence states, such as chromium (III) and chromium (VI), these species are modeled as separate constituents; no redox transformations are assumed to occur.

Whereas EPACMTP simulates steady-state flow in both the unsaturated zone and the saturated zone; contaminant transport can be either steady-state or transient. Modeling flow as steady-state assumes that the infiltration and recharge, as well as ambient groundwater flow conditions, are representative of long-term average conditions that do not change over the duration of the simulation process. The model does not account for transient flow conditions that may result from temporal (e.g., seasonal) variations in precipitation and regional groundwater flow. The EPACMTP model can accommodate either steady-state or transient contaminant transport. Steady-state transport modeling corresponds to a very conservative risk scenario in which a waste unit continues to release contaminants indefinitely (continuous source), so that eventually the receptor well concentration reaches a constant, never changing value. In this analysis, transient transport simulations were performed. This analysis is referred to as a "finite source" scenario because it explicitly takes into account the finite time period over which a waste unit operates and releases contaminants, as well as the actual time it may take contaminants to reach the receptor well. The leachate concentrations calculated by the landfill partition model were used to calculate a total waste concentration to leachate concentration ratio, which was then used to determine the length of time that the landfill will leach.

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 maximum time-averaged concentration. In this analysis, a maximum 9-year average was used for all constituents. The 9-year exposure time period corresponds to the average residence time used in the subsequent risk calculations. The maximum 9-year average receptor well concentration represents the highest average concentration at the well over any 9-year time period.

The metals-modeling methodology in EPACMTP incorporates three options to specify the K_d for a given metal:

- As a statistical distribution of values
- As an empirical distribution of values
- Chosen from adsorption isotherms coded into the EPACMTP program. An isotherm is an expression of the equilibrium relationship between the aqueous concentration and the sorbed concentration of a metal (or other constituent) at constant temperature.

For this analysis, the K_d for most metals was defined as an empirical distribution when sufficient data were available (i.e., nine or more data points based on a search of literature values). The K_d for metals with fewer data available from the scientific literature was defined as a log uniform distribution of values. The log uniform distribution was developed by taking the mean of the logs of the available data and extending the distribution by 1.5 log units in both directions. The third option for determining the K_d involves the automated use of adsorption isotherms. Because technical issues have been raised regarding the data supporting the adsorption isotherms, EPA decided not to use this component of EPACMTP; thus, this third option was not used for any of the constituents in this analysis. More details about the development of statistical distributions of K_d values used in this analysis are presented in Appendix H.

Modeling of organic constituents was based on modeling four organic constituent groups instead of each individual organic constituent to reduce the number of model runs required. Four organic chemical groups were identified based on chemical properties of the constituents (i.e., the organic carbon partition coefficient, K_{oc}, and hydrolysis constant, K_h). The specific constituents modeled as surrogates were ethylene glycol, chloroform, acrylonitrile, and tetrachloroethylene. Pentachlorophenol represented a special case in that it is an ionizing organic compound and its K_{oc} value changes as a function of pH. Therefore, pentachlorophenol was modeled individually along with the four surrogate groups. Two other groups representing dibutylphthalate, di(2-ethylhexylphthalate), ethylbenzene, and xylenes were screened from the groundwater analysis because the leachate concentration predicted by the source partition modeling was zero at the 90th percentile level. These constituents had high k_{oc} values. The specific constituents that are associated with these surrogate groups are identified in Appendix O.

The exposure concentration is evaluated at the intake point of a hypothetical residential drinking water well located at a specified distance from the downgradient edge of the waste management unit and at a specified depth below the water table. The location of the receptor well intake point is defined in terms of its x, y, and z coordinates. (See Figure 5-9, in which x refers to the downgradient distance from the edge of the waste unit, y refers to the horizontal transverse distance from the plume centerline, and z refers to the vertical distance below the water table.) The shaded areas in Figure 5-9 conceptually represent the horizontal and vertical extent of the contaminant plume. In this analysis, the x, y, and z distances are defined as

x = 0 to 1 mile
y = 0 to edge of plume
z = 0 to 10 m (or base of aquifer if less than 10 m).

In other words, the well is placed at a downgradient distance up to 1 mile, based on a nationwide distribution of nearest downgradient residential wells from Subtitle D municipal landfills. This distribution is discussed and presented in Section 4.3. It is assumed that the same distribution holds for other types of waste units as well. The limits on the y-location of the well place the receptor well within the lateral extent of the plume. Because of symmetry around the plume centerline, only one-half of the lateral extent of the plume is actually modeled. A uniform distribution is used to model the y-location of the well. The constraints on the vertical depth of the well intake point reflect the fact that residential wells are expected to be in the shallower portion of the aquifer. The z location of the well is modeled using a uniform distribution with limits of 0 (i.e., well at the water table) to the lesser of 10 m or the total saturated aquifer thickness if the latter is less than 10 m.

Assuming a square waste unit, so that the square root of the waste unit area equals the length and width of the unit, the lateral extent of the plume, which is used to determine the y location of the well, is approximated as

$$Plume\ \ edge \ = \ 0.5\sqrt{(AREA)} \ + \ 3\sqrt{\alpha_T \times 2(\sqrt{(AREA)} + x_{rw})} \tag{5-2}$$

where

AREA = area of the waste unit
α_T = horizontal transverse aquifer dispersivity
x_{rw} = downgradient distance of receptor well from edge of waste unit.

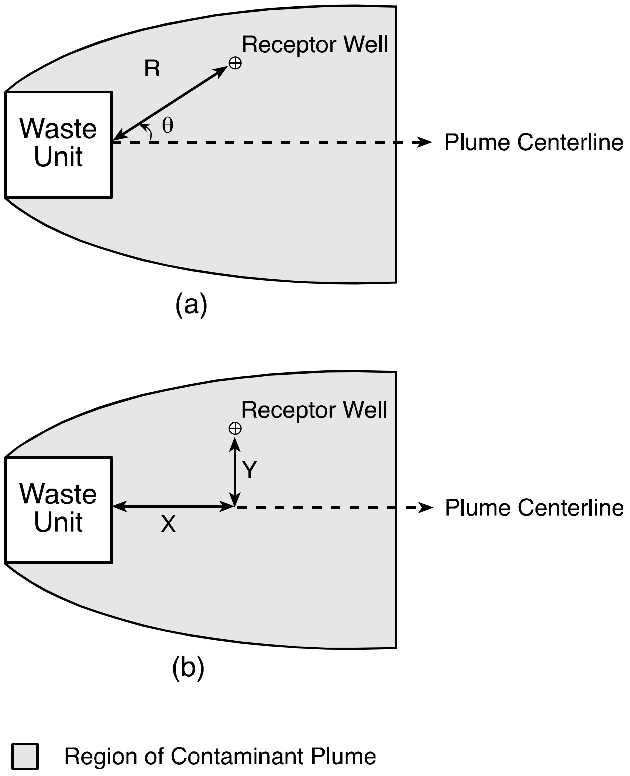


Figure 5-9. **Schematic plan view showing procedure for determining the downstream location of the receptor well: (a) well location determined using radial distance, R, and angle off center θ;and (b) well location generated uniformly within plume limit.**

The first term on the right side of this equation represents the contribution from the waste unit area itself to the extent of the plume. The factor 0.5 represents the fact that the plume edge is calculated from the plume centerline; i.e., only one-half of the actual plume width is considered.

The second term on the right-side represents the lateral spreading of the plume due to transverse dispersion, with increasing downgradient distance from the waste unit. The magnitude of dispersive effects is expressed in the α_T parameter. The AREA and x_{rw} terms represent the total downgradient plume travel distance, starting from the upgradient edge of the waste unit.

The above approximation for the lateral extent of the contaminant plume is based on the assumption that plume spreading in the horizontal transverse direction is caused by dispersive mixing, which results in a Gaussian profile of the plume cross section. Under this assumption, the approximation for the plume boundary will capture 99.7 percent of the contaminant mass. The above approximation, however, does not take into account advective spreading of the plume due to lateral groundwater flow in the vicinity of the waste unit and may, therefore, underestimate the actual lateral extent of the plume. However, in this case, the approximation will be conservative, as it will prevent placement of the well at the outer fringes of the plume, which will have the lowest concentrations compared to wells closer in.

Because the subsurface migration of some highly sorbing constituents (e.g., Cr III, Pb) may be very slow, it may take a long time for the plume to reach the receptor well, and the maximum 9-year exposure (see above) may not occur until a very long time after the waste unit ceases operations. This time delay may be on the order of thousands of years. Appendix U presents median and maximum time of arrival of peak concentration for constituents modeled in this analysis. In this analysis, a maximum exposure time period of 10,000 years is used. In other words, if the model predicts that the maximum 9-year exposure has not yet occurred after 10,000 years, the actual receptor well concentration at 10,000 years will be used in the risk calculations. Imposing this time limit discounts potential risks that may not occur until very far into the future relative to exposures and risks that are predicted to occur in the nearer term.

5.2.5 Predicting Indoor Air Concentrations

Constituents that have reached a residential groundwater well may be extracted and used for domestic purposes (e.g., showering). In this analysis, it is assumed that constituents that reach a residential groundwater well will be used not only as drinking water, but also for showering. This type of exposure is called indirect exposure because a constituent is passed from contaminated media to a receptor indirectly. The indirect exposure from showering occurs when vapors are generated as a result of volatilization of the constituent in the domestic water drawn from the groundwater well.

Volatilization of constituents may occur with other indoor water uses as well. In previous analyses, however, showering activity has been the dominant pathway for indoor air exposures for receptors using groundwater. Therefore, only showering activity is modeled in this analysis.

Exposure to constituents released into the indoor air from showering activity was evaluated for the adult resident only. Young children are assumed to take baths primarily and thus not to encounter significant exposure to constituents released during showering activity. The adult resident is assumed to take one shower per day. The duration of each shower event is assumed to be a distribution of values ranging from 1 to 60 minutes. Exposure due to showering with contaminated residential groundwater is assumed to occur 350 days of the year, which is consistent with exposure frequency assumptions made for other exposure pathways.

Appendix P presents the algorithms that are the basis of the shower model and the exposure factors used in the shower model.

5.3 Calculation of Food Chain Concentrations

Constituents can pass from contaminated air, soil, and surface water and reach people and wildlife through the food chain. For example, constituents that are transported in air may be deposited on plants or onto soil in which the plants are growing. Constituents from the air and soil may be accumulated in fruits and vegetables that are consumed by people. In addition, beef and dairy cattle may feed on forage, grain, and silage that is grown in contaminated areas. The beef and dairy products may be subsequently consumed by people. Similarly, constituents that contaminate surface water may be accumulated by fish, which are subsequently consumed by a recreational fisher.

This section presents the methodology used to calculate contaminant concentrations for each of the food chain pathways considered. An approach was developed for a terrestrial food chain to calculate concentrations of produce, beef, and milk that are consumed by the adult and child farmer evaluated in this assessment. In addition, an approach was developed for an aquatic food chain to calculate concentrations in fish that may be consumed by a recreational fisher.

5.3.1 Terrestrial Food Chain

The terrestrial food chain is designed to predict the accumulation of a contaminant in the edible parts of a plant from uptake of contaminants in soil and through translocation and direct deposition of contaminants in air. Concentrations are predicted for three main categories of food crops presumed to be eaten by humans: exposed fruits and vegetables, protected fruits, and root vegetables. The terms “exposed” and “protected” refer to whether or not the edible portion of the produce is exposed to the atmosphere. Examples include tomatoes (exposed vegetable), bananas (protected fruit), and potatoes (root vegetables).

In addition, the terrestrial food chain estimates the contaminant concentration in farm crops for cattle. Vegetation consumed by cattle includes grain, forage, and silage. Grain is considered to be a protected vegetation and forage an exposed vegetation. Silage refers to any plants harvested for animal consumption, whether protected or exposed. Silage is calculated as exposed vegetation; however, an empirical correction factor for silage takes into account that silage is partly protected and partly exposed.

Table 5-5 summarizes the mechanisms by which vegetation can be exposed to contaminants. The three mechanisms are: deposition of particle-bound contaminants to exposed plant tissues, vapor-phase deposition of contaminants to exposed plant tissues, and root uptake. Exposed vegetation is subject to contamination via particulate deposition, vapor-phase deposition, and root uptake, while protected vegetation is contaminated only through root uptake because the edible portion of the vegetation is not in direct contact with air. Accumulation by root uptake can occur through the uptake of soil water for aboveground protected vegetables and fruit or by absorption into the outer parts of the root vegetables.

5.3.1.1 <u>Aboveground Vegetation</u>. Aboveground vegetation is subject to contamination via deposition of particle-bound contaminants, vapor transfer of contaminants, and root uptake as shown in Figure 5-10.

Deposition of Particle-Bound Contaminants. Particle-bound contaminants from the air are deposited by wet and dry deposition; thus they affect only exposed vegetation. As described earlier, an air dispersion model was used to calculate the wet and dry deposition rates for the particle-bound contaminants. The contaminants from dry deposition were assumed to remain on the plant’s surface until weathering occurs. However, only a fraction of contaminant from wet deposition remains on the plant’s surface; the rest washes off immediately.

Table 5-5. Terrestrial Food Chain Vegetation

Type of Vegetation	Particulate Deposition	Vapor-phase Deposition	Root Uptake
Human ingestion			
Exposed vegetables	✓	✓	✓
Exposed fruit	✓	✓	✓
Protected fruit			✓
Root vegetables			✓
Beef and dairy cow ingestion			
Forage	✓	✓	✓
Silage	✓	✓	✓
Grain			✓

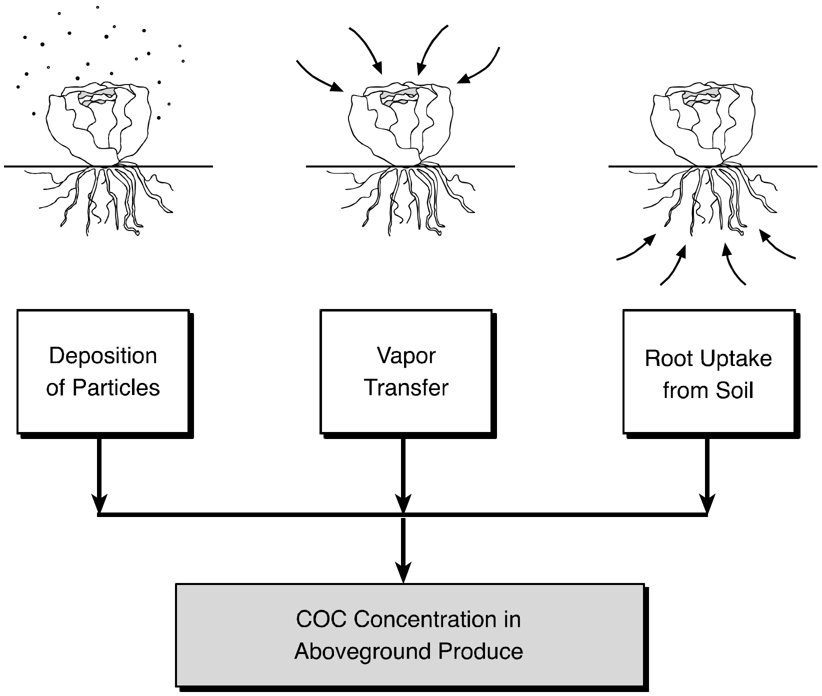


Figure 5-10. Constituent transfer mechanisms for aboveground produce.

Not all airborne particles will settle on a plant's edible surface; some will fall to the ground, others will fall on other surfaces that will undergo weathering processes such as wind removal, water removal, and growth dilution, and most will end up in the soil or runoff. Thus, only a fraction of the total deposition rate per area is used to estimate the amount of airborne particles that contacts the edible portion of the plant.

The calculation of vegetative concentration due to deposition also takes into account the length of time plants are exposed to contaminants. One determination of the length of exposure is the growing season. For instance, the time from when a tomato begins to grow until it is harvested equals its length of exposure to deposition. The productivity level of the plant or biomass is also a factor. The biomass is determined by the amount of standing crop for the average farm. The biomass is needed to take into account the dilution of constituent by biomass growth.

Deposition of Vapor-Phase Contaminants. The concentration of contaminants due to vapor-phase deposition depends on the constituent being considered. Most metals do not exist in vapor form and thus are not assessed for this pathway. Elemental mercury is the exception because it can exist as a vapor. However, in this analysis, elemental mercury was used only to calculate inhalation exposures; therefore, deposition of elemental mercury was not considered. The divalent form of mercury was modeled for food chain exposures because it exists in both vapor and particulate phases and is more readily deposited on crops and soils.

For constituents that have low octanol-water partition coefficient, K_{ow} (i.e., where the log $K_{ow} < 5$), the calculation is based on deposition of contaminants in the vapor phase from the air to the plant tissue. This calculation is essentially the same as the calculation described for particulates and takes into account both wet and dry deposition of vapors. For constituents that have a high log K_{ow} (>5), evidence shows that wet deposition is negligible and deposition of vapor phase constituent by contact with the plant surface is the primary mechanism; therefore, a different equation is used that is based on the vapor-phase air concentration of the constituent.

Vapor-phase deposition for high log K_{ow} constituents uses a constituent-specific air-to-plant biotransfer factor to estimate the concentration of contaminants in vegetation. The air-to-plant biotransfer factor is defined as the ratio of contaminant concentration in exposed plant parts to the vapor-phase concentration of contaminant in air. The biotransfer factors are derived for each type of vegetation and consider leaf density, density of air, and percent of wet leaf that is dry matter. An empirical correction factor recommended by EPA (U.S. EPA, 1997c) is applied to the calculation for each type of vegetation. The factor is used to adjust the air-to-plant bioconcentration factors that are developed using the different types of vegetation considered in this analysis. This factor also is applied to take into account the difference between outer-surface and whole-plant concentrations. This is important for lipophilic organic chemicals that tend to remain on the outer portion of the plant surface since washing and peeling fruits and vegetables reduces the outer surface residues. Because silage is assumed to be partly protected and partly exposed, the correction factor for silage takes into account that some of the vegetation is not contaminated due to vapor deposition onto plant surfaces.

Root Uptake. Plants take up contaminants in the soil through their roots by translocation. Water, which contains contaminants from the soil, moves up the roots, moving contaminants to the aboveground shoots. Root uptake is a function of a constituent's plant-soil bioconcentration factor. The plant-soil bioconcentration factor measures the constituent's ability to accumulate in plant tissue and is the ratio of constituent concentration in plants to the concentration in soil.

For organics, an equation has been developed to calculate the plant-soil bioconcentration factor; for other constituents, this factor was developed based on measured data collected from various sources. The plant-soil bioconcentration factor derived for organic constituents is dependent on the solubility of a constituent in water, which is inversely proportional to the octanol-water partition coefficient, K_{ow}. In constituents that are more lipophilic and do not move readily through the roots, the bioconcentration factor is assumed to account for possible resuspension and redeposition, which may add to constituent concentration.

5.3.1.2 **<u>Belowground Vegetation</u>.** In belowground plants, roots can take in contaminants from the soil that may accumulate in the edible portion of the plant. The calculation of concentrations in root vegetables varies depending on the constituent type. For metals and mercury, the calculation is essentially the same as discussed for root uptake in aboveground plants. For organics, the calculation is a function of the root concentration factor, which is used to estimate the amount of constituent moving from the soil into the root vegetable.

Similar to the calculation for vapor-phase deposition, an empirical correction factor is applied to the concentrations in belowground vegetables. The correction factor is applied to adjust the root concentration factor so that it is appropriate for bulky belowground root crops. The factor also accounts for constituent losses due to cleaning and cooking and the tendency of lipophilic contaminants to remain in the outer portions of the root (U.S. EPA, 1997c).

5.3.1.3 **<u>Animal Tissue Concentration</u>.** The animal products considered in this risk analysis are beef and milk from beef and dairy cattle, respectively. The contaminant concentrations in beef tissue and milk were estimated based on the amount of contaminant the cattle were assumed to have consumed through ingestion. Specifically, the ingestion rates for cattle were calculated for intake of plant material, incidental soil ingestion, and water ingestion. Plant ingestion comes from forage, grain, and silage. The animals are also assumed to ingest soil, with which they come in contact during grazing or other activities on untilled soils. A stream located at the end of the agricultural field is used as the drinking water source for cattle. It is assumed that water is not treated prior to consumption by cattle. Therefore, calculations for the ingestion of contaminants via drinking water for cattle are based on the total contaminant concentration in the waterbody (i.e., dissolved and suspended sediment).

Different ingestion rates were used for beef and dairy cattle, depending on the amount of feed they consume and the activity patterns of the animals. For example, beef cattle are assumed to spend more time grazing and, therefore, have a higher incidental ingestion rate of soil. The animal concentrations also depend on biotransfer factors, which are the ratio of the contaminant concentration in animal tissue to the daily intake of contaminant by the animal. Constituent-specific biotransfer factors were derived for both milk and beef. Chemical concentrations in feed, soil, and water are multiplied by their respective ingestion rates and by constituent-specific BTFs and then summed to obtain the concentration of individual constituents in tissue.

5.3.2 Aquatic Food Chain

An aquatic food chain model was used to estimate the concentration of constituent that may accumulate in fish. It is assumed for this analysis that fish are a food source for a recreational fisher. Trophic level three (T3) and four (T4) fish were considered in this analysis. Trophic level three fish are those that consume invertebrates and plankton. Trophic level four fish are those that consume other fish. Most of the fish that humans eat are T4 fish (e.g., salmon, trout, walleye, bass) and medium to large T3 fish (e.g., carp, smelt, perch, catfish, sucker, bullhead, sauger).

The concentration of constituent that accumulates in fish is calculated using the concentration calculated for the waterbody adjacent to the buffer. The contaminants in the water column consist of dissolved constituents and constituents associated with suspended solids. For metals, the calculation of contaminants in fish is based on the total concentration of contaminants in the waterbody (i.e., dissolved and suspended solids). For organics and mercury, the calculation of contaminants in fish is based on the dissolved fraction because they are more bioavailable in that form.

Fish tissue concentrations are dependent on a bioconcentration factor (BCF) or a bioaccumulation factor (BAF), depending on the constituent. These factors are needed to estimate the amount of constituent being transferred from the waterbody into the fish tissue. Specifically, they reflect the ratio between the tissue concentration in fish and the appropriate waterbody concentration. BCFs only take into account partitioning from the waterbody to the fish and do not consider accumulation through the food chain. BCFs are used for organics with a low octanol-water partition coefficient, K_{ow} (i.e., where the log $K_{ow} < 5$), which is an indicator that the constituent does not readily bioaccumulate up the food chain. BAFs consider not only uptake from the waterbody, but also uptake through the food chain. BAFs are either modeled or based on empirical data and are used for metals and organics with high log K_{ow}, which is an indicator of a constituent’s tendency to bioaccumulate. It should be noted that constituents that are known to metabolize in fish (e.g., phthalates) can be overestimated if the BAF is based on the K_{ow}. For such constituents, empirical data were used to ensure that fish tissue concentrations were not greatly overestimated. If empirical data were not available, a default value of 1 was used.

BCFs and BAFs were developed for each constituent to reflect accumulation in each trophic level considered. They were also developed to estimate the concentration in the fish filet versus the total fish. Essentially, the BCFs and BAFs were modified for each calculation to account for varying lipid content in each type of fish and portion of the fish being evaluated. The fish concentrations calculated for human receptors are generally lower than whole fish concentrations. Human receptors consume only the filet portion of the fish, which has a lower lipid content. Because constituents tend to accumulate in the fatty tissue, the concentration in the filet portion of the fish is lower than the concentration in the whole fish.

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6.0 Human Exposure Assessment

This section describes the human exposure assessment that was conducted for this risk assessment. An exposure assessment is the determination or estimation of the magnitude, frequency, duration, and route of exposure to contaminants that an individual may experience. The term "exposure," as defined by the EPA exposure guidelines (U.S. EPA, 1992), is the condition that occurs when a contaminant comes into contact with the outer boundary of the body. The exposure of an individual to a contaminant is what completes an exposure pathway (i.e., the course a constituent takes from the waste management unit to an exposed individual). Once the body is exposed, the constituent can cross the outer boundary and enter the body. The amount of contaminant that crosses and is available for adsorption at internal exchange boundaries is referred to as the "dose" (U.S. EPA, 1992). Each exposure pathway, as illustrated in Figure 6-1, includes an exposure point and exposure route.

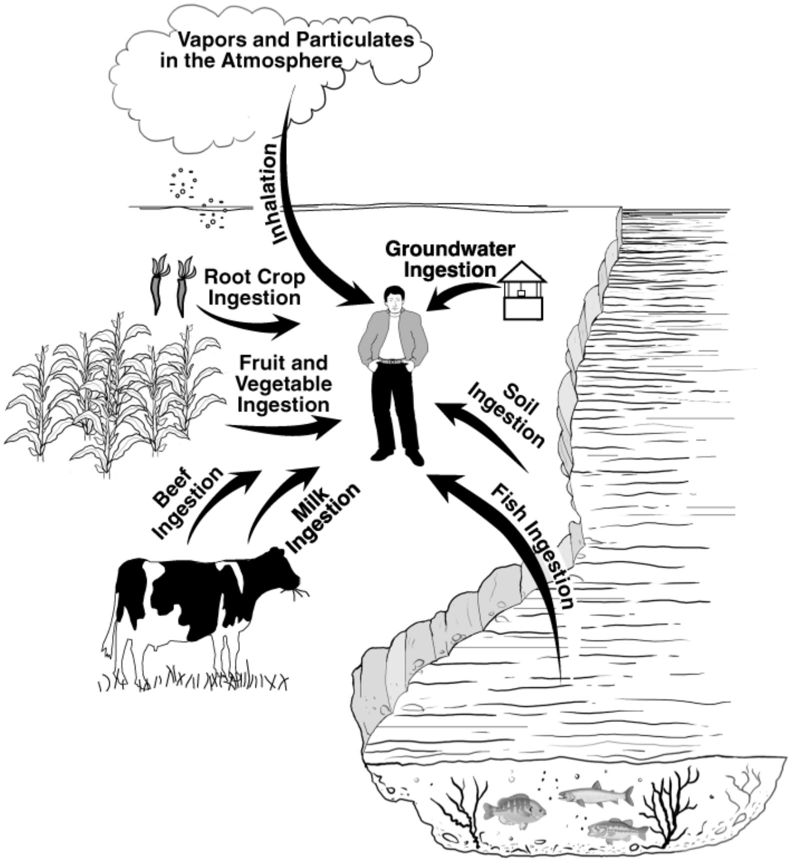

Figure 6-1. Human exposure pathways.

The paint listings risk assessment evaluated the risk to receptors in the vicinity of a WMU. The individuals evaluated were those residents closest to the WMU. The distances from the WMU to the residents were taken from empirical data and are described in detail in Section 4.3.

Although all receptors are placed in the vicinity of the WMU, not all individuals experience the same exposure. Different individuals will have a different magnitude, frequency, duration, and route of exposure. Two steps were taken in this analysis to capture the variability in individual exposures by taking into account differences in physiological characteristics and daily activity patterns. The first step was to define major categories of lifestyle that individuals living near a WMU might have. This step created five distinct types of receptors. The second step was to vary the values (i.e., exposure factors) used to calculate exposure/intake within each of the receptor types. These two steps are described in the following sections. Section 6.1 presents an overview of the receptors, selected exposure pathways, and exposure scenarios considered for this assessment. Section 6.2 presents particular exposure factors (i.e., values needed to calculate human exposure) used in the analysis. Section 6.3 describes the methods used to estimate dose including average daily dose (ADD) and lifetime average daily dose (LADD).

6.1 Receptors and Exposure Pathways

Five types of human receptors were assumed to be representative of the individuals who might live around a WMU: a child resident, an adult resident, a child of a farmer, an adult farmer, and an adult recreational fisher. These receptors reflect the range of possible individual exposures for direct and indirect exposure pathways. The routes of exposure differ for the resident, farmer, and fisher. For example, for this assessment, it is assumed that a resident is likely to purchase produce from a market and that a farmer consumes produce grown on the farm.

Table 6-1 lists each receptor along with the specific exposure pathways that apply to that receptor. The groundwater pathways were evaluated separately because they may occur on a different time frame from that of other pathways. Air emissions occur immediately after waste disposal, while it may take hundreds of years for most contaminants to leach from a WMU, travel in the groundwater, and reach a residential dwelling. As the table illustrates, not all receptors are exposed through the same pathways. Adult and child residents are exposed via inhalation of air, ingestion of soil, and ingestion of groundwater. Although the time frame of exposure to contamination via air, soil, and groundwater could overlap for some organic constituents, the exposures are considered separately because receptors are randomly placed throughout the area of exposure and may not coincide with the location of the groundwater plume. Only adult residents are exposed to the groundwater pathway via the inhalation of vapors generated from constituents volatilizing from well water during showering. In addition to the inhalation of air and ingestion of soil, the adult and child farmer are exposed via ingestion of homegrown above- and belowground produce, beef, and dairy products. The fisher is a recreational angler who also is a resident and is exposed to air and soil in addition to catching fish in the nearby waterbody.

Table 6-1. Receptors and Exposure Pathways

Receptor	Inhalation of Ambient Air	Ingestion of Soil	Ingestion of Above- and Belowground Produce	Ingestion of Beef and Dairy Products	Ingestion of Fish	Inhalation of Indoor Air (Shower)	Ingestion of Drinking Water
Adult resident	✓	✓					
Child resident	✓	✓					
Farmer	✓	✓	✓	✓			
Child farmer	✓	✓	✓	✓			
Fisher	✓	✓			✓		
Adult resident[a]						✓	✓
Child resident[a]							✓

[a] Groundwater pathways were considered separately for the adult resident and the child resident because the time frame for groundwater exposure often is not consistent with that of other exposure pathways. Furthermore, aboveground receptors are randomly located and do not necessarily coincide with the location of the groundwater plume.

6.1.1 Childhood Exposure

Children are an important subpopulation to consider in a risk assessment because they are likely to be more susceptible to exposures than adults; compared to adults, children may eat more food and drink more fluids per unit of body weight. This higher intake-rate-to-body-weight ratio can result in a higher average daily dose than adults experience.

As children mature, however, their physical characteristics and behavior patterns change. To capture these changes in the analysis, the life of a child was divided into several cohorts: cohort 1 (ages 1 to 5), cohort 2 (ages 6 to 11), cohort 3 (ages 12 to 19), and cohort 4 (ages 20 to 70). Each cohort has distributions of the values, called “exposure parameters,” that are required to calculate exposure to an individual. The exposure parameter distributions for each cohort reflect the physical characteristics and behavior patterns of that age range. Data from the *Exposure Factors Handbook* were used to derive distributions appropriate for each cohort (U.S. EPA, 1997a,b,c). It should be noted that the distributions for the 20- to 70-yr-old cohort are the same as those used for adult receptors.

The development of the child exposure parameters consisted of three steps:

- Define the start age and cohort of the child
- Select the exposure duration of the child
- Calculate time-weighted exposure parameters.

To capture the higher intake-rate-to-body-weight ratio of children, a start age between the ages of 1 and 6 was selected. For the probabilistic analysis, a start age between these ages was selected randomly for each iteration. For the deterministic analysis, the start age was set at 3 years.

To select the exposure duration for each of the 10,000 iterations in the probabilistic analysis, a distribution was chosen to define the exposure duration based on the start age. For example, if the start age was 2, the distribution for cohort 1, a 1- to 5-yr-old, was used to define exposure duration. However, if the start age was 6, the distribution developed for cohort 2, children between ages 6 and 11, was used to define exposure duration. In the deterministic approach, a central tendency or high-end exposure duration was used for each cohort rather than a distribution.

After the start age and the exposure duration were defined for a given iteration, all the other exposure parameters needed to calculate exposure to a child were randomly selected from the distributions associated with each of the cohorts through which the child would age. Once this selection was finished, an exposure parameter selected from each cohort was time-weighted and combined with its corresponding value from the other cohorts to create a single time-weighted parameter. For example, the beef ingestion rates selected from each age cohort were time-weighted according to the number of years the child remained in the cohort and combined to generate a single time-adjusted beef ingestion rate for the child. The same was done for all the parameters (e.g., body weight, inhalation rate, fruit ingestion rate) required to perform an assessment of exposure. The equation used to combine each cohort's exposure parameters into one time-weighted exposure parameter is as follows:

$$EP_{TW} = \frac{(EP_1 \times ED_1) + (EP_2 \times ED_2) + (EP_3 \times ED_3) + (EP_4 \times ED_4)}{ED} \qquad (6\text{-}1)$$

where

EP_{TW} = time-weighted exposure parameter (e.g., ingestion rate of milk, body weight)
EP_1 = exposure parameter cohort 1 (ages 1 to 5)
ED_1 = time spent in cohort 1
EP_2 = exposure parameter cohort 2 (ages 6 to 11)
ED_2 = time spent in cohort 2
EP_3 = exposure parameter cohort 3 (ages 12 to 19)
ED_3 = time spent in cohort 3
EP_4 = exposure parameter cohort 4 (ages 20 to 70)
ED_4 = time spent in cohort 4
ED = total exposure duration of the receptor (sum of $ED_1 + ED_2 + ED_3 + ED_4$).

The time-weighted exposure parameters were used to assess the carcinogenic risk to the child receptors. Because noncarcinogenic risk is generally higher for lower age groups due to higher intake rates relative to body weight, the assessment of noncancer hazards to children used the exposure parameters from the cohort corresponding to the child's start age.

6.1.2 Exposure Pathways

Human receptors may come into contact with COCs present in environmental media by a variety of pathways. In general, exposure pathways are either direct, such as inhalation of ambient air, or indirect, such as the farm food chain pathways. The exposure pathways considered in this assessment were inhalation of ambient air, ingestion of soil, ingestion of aboveground produce, ingestion of belowground produce (i.e., root crops), ingestion of beef and dairy products, ingestion of fish, inhalation of indoor air contaminated from groundwater via showering, and ingestion of drinking water from contaminated groundwater sources. As noted previously, exposure to groundwater was considered separately in this analysis because the time scales differ and the groundwater plume occurs in a limited area relative to potential receptor locations.

6.1.2.1 Inhalation of Ambient Air. Both vapors and particles can be inhaled in ambient air by a receptor. All receptors were affected via direct inhalation.

6.1.2.2 Ingestion of Soil. All receptors were exposed to soil based on incidental ingestion, mostly due to hand-to-mouth behavior. Soil ingested was modeled as the top 1 cm of soil and as untilled.

6.1.2.3 Ingestion of Above- and Belowground Produce. Ingestion of the following categories of produce were used in this risk assessment: exposed fruit, protected fruit, exposed vegetables, and root vegetables. For aboveground produce, the terms “exposed” and “protected” refer to whether the edible portion of the plant is exposed to the atmosphere. It was assumed that farmers grow a portion of their fruits and vegetables on land near the WMU and that these fruits and vegetables become contaminated via soil and air. Belowground produce refers to root crops grown by the farmer. The soil root crops were grown in was assumed to be tilled for these pathways, so COCs were mixed throughout the root zone.

6.1.2.4 Ingestion of Beef and Dairy Products. Beef and dairy cattle were assumed to be exposed to COCs via differing intake rates of contaminated forage, silage, grain, and water. Adult and child farmer receptors were assumed to consume beef and drink milk from the dairy cattle.

6.1.2.5 Ingestion of Fish. Fish are exposed to COCs via uptake of contaminants from surface waters. Adult fishers were assumed to consume fish caught in local waterbodies.

6.1.2.6 Inhalation of Indoor Air. Groundwater is contaminated from COCs leaching from the WMU into the vadose zone and migrating into the groundwater. Groundwater was assumed to be extracted via an offsite well into the residence and used for showering by the adult residential receptor. COCs can volatilize from shower water, thus resulting in inhalation exposures.

6.1.2.7 Ingestion of Drinking Water. Ingestion of drinking water is also an indirect groundwater pathway similar to inhalation of indoor air. Groundwater from the offsite well was assumed to be used as a source of drinking water for residents (adult and child). Both groundwater pathways (i.e., inhalation of indoor air and ingestion of drinking water) were considered separately from other pathways since they typically occur on a different time frame and receptor locations do not necessarily coincide.

6.2 Exposure Factors

The exposure factors used in this risk assessment are listed in Table 6-2, along with the data sources and whether they were represented by a distribution or a fixed value in the Monte Carlo analysis. Exposure factors are used to calculate the dose of a chemical based on contact with contaminated media or food, the duration of that contact, and the body weight of the exposed individuals. The primary data source of human exposure model inputs used in this risk assessment was EPA's *Exposure Factors Handbook* (U.S. EPA, 1997a,b,c). The *Exposure Factors Handbook* summarizes data on human behaviors and characteristics related to human exposure from relevant key studies and provides recommendations and associated confidence estimates on the values of exposure factors. EPA carefully reviewed and evaluated the quality of the data before their inclusion in the *Exposure Factors Handbook.* EPA's evaluation criteria included peer review, reproducibility, pertinence to the United States, currency, adequacy of the data collection period, validity of the approach, representativeness of the population, characterization of the variability, lack of bias in study design, and measurement error (U.S. EPA, 1997a,b,c).

For probabilistic risk analyses, probability distribution functions (PDFs) were developed from the values in the *Exposure Factors Handbook*. For deterministic analyses, high-end and central tendency values were selected based on data and recommendations in the *Exposure Factors Handbook*. Because exposure duration was evaluated as a potentially sensitive parameter in the high-end deterministic analysis, both central tendency and high-end values were evaluated.

Appendix G presents the exposure factors used in the probabilistic and deterministic risk analyses, respectively. Appendix G also describes the rationale and data used to select the parametric models (gamma, lognormal, and Weibull) for those exposure factors that were varied in the probabilistic analysis, the maximum and minimum exposure parameter values used in the probabilistic analyses, and the central tendency and high-end values used for the deterministic analysis.

6.2.1 Intake Factors

This section presents the basis for the intake rates used for soil, food items, and drinking water in the probabilistic and deterministic analyses. Adult and child receptor intake rates for water, soil, and food items were derived from data in the *Exposure Factors Handbook* (U.S. EPA, 1997a,b).

6.2.1.1 Soil Ingestion. Ingestion of contaminated soil is a pathway common to all receptors. Because most available data are from studies measuring soil ingestion in children under the ages of 5 or 6, the adult soil ingestion rate was used for children older than age 5.

Table 6-2. Human Exposure Factor Input Parameters and Data Sources

Parameter	Variable Type	Data Source
Body weight (adult, child)	Distribution	U.S. EPA (1997a)
Inhalation rate (adult, child)	Distribution	U.S. EPA (1997a)
Ingestion rate: soil (adult, child)	Distribution	U.S. EPA (1997a)
Ingestion rate: drinking water (adult, child)	Distribution	U.S. EPA (1997a)
Consumption rate for farmer: exposed vegetables (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for farmer: root vegetables (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for farmer: exposed fruit (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for farmer: protected fruit (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for recreational fisher: fish (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for farmer: beef (adult, child)	Distribution	U.S. EPA (1997b)
Consumption rate for farmer: milk (adult, child)	Distribution	U.S. EPA (1997b)
Shower contact time (adult)	Distribution	U.S. EPA (1997c)
Exposure duration (adult, child)	Distribution	U.S. EPA (1997c)
Exposure frequency (adult, child)	Fixed (constant)	U.S. EPA policy
Fraction contaminated: soil	Fixed (constant)	U.S. EPA policy
Fraction contaminated: drinking water	Fixed (constant)	U.S EPA policy
Fraction contaminated for recreational fisher (fish)	Fixed (constant)	U.S. EPA (1997b)
Fraction homegrown for farmer: exposed vegetables	Fixed (constant)	U.S. EPA (1997b)
Fraction homegrown for farmer: root vegetables	Fixed (constant)	U.S. EPA (1997b)
Fraction homegrown for farmer: exposed fruit	Fixed (constant)	U.S. EPA (1997b)
Fraction homegrown for farmer: protected fruit	Fixed (constant)	U.S. EPA (1997b)
Fraction contaminated (home-raised) for farmer: beef	Fixed (constant)	U.S. EPA (1997b)
Fraction contaminated (home-raised) for farmer: dairy	Fixed (constant)	U.S. EPA (1997b)
Event frequency–showering	Fixed (constant)	U.S. EPA (1997c)
Fraction of trophic level 3 (T3) fish consumed	Fixed (constant)	U.S. EPA (1997b)
Fraction of trophic level 4 (T4) fish consumed	Fixed (constant)	U.S. EPA (1997b)
Food preparation and cooking losses: beef	Fixed (constant)	U.S. EPA (1997b)
Human lifetime (used in carcinogenic risk calculation)	Fixed (constant)	U.S. EPA policy

Thus, soil ingestion rates used in the probabilistic analysis were varied for children under the age of 5 only; soil ingestion rates for children over the age of 5 and adults were not varied.

6.2.1.2 Fruits and Vegetables Ingestion. Ingestion of contaminated homegrown fruits and vegetables is a potential pathway of exposure for adult farmers and their children. Consumption rate data of homegrown exposed fruit, protected fruit, exposed vegetables, and root vegetables by these receptors were obtained from the *Exposure Factors Handbook*. The terms "exposed" and "protected" refer to whether the edible portion of the plant is exposed to the atmosphere. Examples of exposed fruits are apples, peaches, pears, and berries. Protected fruits include melons, bananas, oranges, and pineapples. Aboveground exposed vegetables include tomatoes, green leafy vegetables (e.g., lettuce, cabbage, kale), cucumber, summer squash, peppers, broccoli, okra, and snap beans. Common root vegetables include carrots, onions, potatoes, and beets (U.S. EPA, 1997b).

Although farmers grow much of their food, they generally do not grow all of their food. Therefore, the fraction of their diet that may be contaminated was considered. Specifically, the *Exposure Factors Handbook* provides recommendations on the percent of the total diet of farmers that is homegrown. Adult and child residents were assumed to purchase all of their food from supermarkets.

6.2.1.3 Beef and Dairy Ingestion. The farmer (adult and child) is the only receptor who is assumed to ingest beef and dairy products from cattle raised in the immediate vicinity of a WMU. As with fruits and vegetables, it was necessary to consider the fraction of the total beef and dairy in the farmer's diet that consists of products raised in the immediate vicinity of a WMU. In addition, beef consumption rate data were adjusted to account for food preparation and cooking losses.

6.2.1.4 Fish Ingestion. Fish ingestion rates were based on an adult recreational angler who catches and eats some fish from a stream impacted by contaminants released from WMUs. The fraction of fish intake that is home-caught is 0.325 for households that fish (Table 13-71, U.S. EPA, 1997b).

6.2.1.5 Drinking Water Ingestion. Use of groundwater from a contaminated well downgradient from a WMU is a pathway common to adult and child residents. Groundwater pathways were considered separately for the adult resident and the child resident because the time frame for groundwater exposure often is not consistent with that of other exposure pathways. Furthermore, aboveground receptors are randomly located and do not necessarily coincide with the location of the groundwater plume.

6.2.1.6 Inhalation Rates. The *Exposure Factors Handbook* reports inhalation values by age, gender, activity pattern, and outdoor workers; however, it does not provide high-end values in most cases. The inhalation rate is the same for all adults, whether resident, farmer, or fisher, while both child receptors (resident and farmer) use the same child inhalation rate.

6.2.2 Other Exposure Factors

6.2.2.1 Body Weights. Distributions of body weight were developed for adult (resident, farmer, and fisher) and child (resident and farmer) receptors based on data from the *Exposure Factors Handbook.*

6.2.2.2 Exposure Duration. Exposure duration refers to the amount of time that a receptor is exposed to a contaminant source. For this risk analysis, exposure duration was assumed to correspond with the receptor's residence time in the same house. Exposure durations were determined using data on residential occupancy from the *Exposure Factors Handbook* (U.S. EPA, 1997c). The data used to develop parameter information for resident receptors were age-specific. Thus, separate distributions were developed for both adult and child residents. Because the only data available for farmers were not age-specific, exposure duration did not differ between adult and child farmers. Exposure durations used in this risk assessment are provided in Appendix G.

Exposure duration for all adult and child receptors was capped at 50 years based on exposures from WMUs. The operating lifespan of surface impoundments and tanks is 50 years. For landfills, air emission exposures were further limited to 30 years, the assumed operating life of landfills in this analysis. Because of residual contamination in soil, receptors were allowed to be exposed to COCs from landfills for up to 50 years via noninhalation pathways. For adults, 50 years is the maximum exposure duration, starting at age 20, that will not exceed the 70-year lifetime assumption implicit in the averaging time used. To be consistent with adult lifetime assumptions as well as the operating lifespan of WMUs, child receptor exposures were also limited to 50 years.

6.2.2.3 Exposure Frequency. Exposure frequency is the frequency at which the receptor is exposed to the contaminated source during the exposure duration. Exposure frequency is not expected to vary much, so distributions were not developed. All receptors were assumed to be exposed to the contaminant source 350 d/yr. This value is based on an assumption that individuals are away from their homes (e.g., on vacation) approximately 2 weeks out of the year.

6.2.2.4 Lifetime and Averaging Time. Averaging time is the period of time over which a receptor's dose is averaged. When evaluating carcinogens, total dose is averaged over the lifetime of the individual, assumed to be 70 years. When evaluating noncarcinogens, dose is averaged over the last year of exposure because noncancer effects may become evident during less-than-lifetime exposure durations if toxic thresholds are exceeded. Essentially, this amounts to setting exposure duration and averaging time equal so that they cancel each other out in the equation for average daily dose. Thus, neither exposure duration nor averaging time is included in the ADD equation.

6.3 Dose Estimates

The purpose of the exposure assessment is to estimate the dose to each receptor by combining intake values with media concentrations. Estimates of exposure are based on the potential dose (e.g., the dose ingested or inhaled) rather than the applied dose (e.g., the dose delivered to the gastrointestinal tract) or the internal dose (e.g., the dose delivered to the target organ). This is generally consistent with the exposure metric used in most epidemiologic and toxicologic studies that serve as the basis for establishing the toxicological benchmarks used for risk assessment (see Section 7.0).

Doses from individual pathways (e.g., soil, exposed vegetables) were calculated by multiplying the contaminant concentration with the respective intake rate on a per kilogram body weight basis. Doses received from the various ingestion pathways (e.g., soil, food) were then summed over the period of time in which exposure occurred, resulting in an average daily dose received from ingestion exposure. The ADD was used to assess noncancer risk from ingestion exposures. Inhalation noncancer risks were evaluated using ambient air concentrations. For cancer effects, where the biological response is described in terms of lifetime probabilities even though exposure may not occur over the entire lifetime, dose is presented as a lifetime average daily dose. The LADD was used to assess cancer risks from each exposure route (i.e., inhalation and ingestion).

6.3.1 Average Daily Dose

For the purposes of this risk analysis, the ADD was defined as

$$\mathrm{ADD} = \mathrm{C} \times \mathrm{IR} \tag{6-2}$$

where

C = concentration, mass/volume or mass/mass
IR = intake rate mass/body weight mass/time or volume/body weight mass/time.

Contaminant concentration represents the concentration of a chemical in a medium that contacts the body. Intake rate for the respective ingestion pathway was applied. For several food parameters, intake rates were provided in milligram/kilogram body weight/day. However, intake rates for fish, soil, and water were adjusted by body weight in order to be on a milligram/kilogram body weight basis. The ADD was calculated for a 1-year period corresponding to the last year of operation for the WMU (i.e., year 30 for landfills or 50 for surface impoundments and tanks), and the fate and transport modeling was based on long-term values reflecting multiyear averaging of data for certain model inputs.

Pathway-specific ADDs, designated as ADD_is, were calculated for individual ingestion pathways (e.g., soil, exposed vegetables). Each ingestion pathway was also evaluated individually to assess its contribution to the total noncancer risk. The summation of the ADD_is results in an ADD for the ingestion pathway (ADD_{ingest}), which was used to assess noncancer risks from total ingestion exposure. However, risks incurred from exposure to contaminated drinking water were not included in the ADD_{ingest} and were evaluated separately due to potential differences in the respective time frames and receptor locations for groundwater and air pathways. An inhalation pathway ADD was not calculated for inhalation exposures because inhalation noncancer risks were evaluated using ambient air concentrations.

6.3.2 Lifetime Average Daily Dose

The lifetime average daily dose, used for assessing risks for carcinogenic effects, was defined as

$$\mathrm{LADD} = \frac{C \times IR \times ED \times EF}{AT \times 365} \qquad (6\text{-}3)$$

where

C = average concentration, mass/mass or mass/volume
IR = intake rate, mass/body weight mass/time or volume/body weight mass/time
ED = exposure duration, yr
EF = exposure frequency, d/yr
AT = averaging time, yr.

As with the ADD, contaminant concentration represents the concentration of a chemical in a medium that contacts the body. Intake rate depends on the route of exposure; for example, it might be an inhalation rate or an ingestion rate. Exposure frequency is the number of days per year the receptor is exposed to the contaminated source during the exposure duration; 365 is the number of days in a year.

For cancer effects, biological responses are described in terms of lifetime probabilities, even though exposure may not be lifelong. Here, the exposure duration (the length of time of contact with a contaminant) was used to average the ADD over a lifetime (70 years). Note that the media concentrations used in the analysis for assessing the LADD (e.g., soil concentration) were generally averaged explicitly over the duration of exposure. This provides a more exact estimate of the lifetime average daily dose. An $\mathrm{LADD}_{\mathrm{ingest}}$ was calculated for ingestion exposures and an $\mathrm{LADD}_{\mathrm{inh}}$ was calculated for inhalation exposures.

6.4 References

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