Chapter 2
Estimation of Chemical Releases and Media Concentrations

What's covered in Chapter 2:

- Source Release Systems
- Releases to Groundwater and Estimation of Groundwater Waste Concentrations
- Releases to Surface Pathways and Estimation of Waste Concentrations in Receiving Media

This chapter describes the methodology and assumptions used in the Delisting Risk Assessment Software (DRAS) program to compute releases of waste constituents from petitioned wastes and estimate waste constituent concentrations in media at the POEs. Source release systems are discussed in Section 2.1, releases to groundwater are discussed in Section 2.2, and releases to surface pathways are discussed in Section 2.3.

2.1 SOURCE RELEASE SYSTEMS

The risk-based delisting process involves performing a risk assessment for petitioned wastes that are disposed of in the two waste management units of concern to the U.S. EPA Region 6 Delisting Program: surface impoundments and landfills. The process determines whether a waste that is petitioned for an exclusion (delisting) is not characteristically toxic and is thus exempt from Subtitle C disposal requirements, assuming that the petitioned waste meets all other criteria for delisting (see U.S. EPA 1996e for other delisting criteria). Once delisted, the petitioned waste may be disposed of in any municipal or industrial nonhazardous waste Subtitle D disposal unit. U.S. EPA Region 6 is focusing its delisting petition evaluations on liquid and solid wastes disposed of in surface impoundments and landfill, respectively. Wastes disposed of in other waste management units will be evaluated on a case-by-case basis.

U.S. EPA Region 6 assumes that petitioned liquid wastes will be disposed of in surface impoundments and that petitioned solid wastes will be disposed of in landfills. Whether a waste is liquid or solid is determined using methods specified in U.S. EPA's "Test Methods for Evaluating Solid Waste" (SW-846) (U.S. EPA 1997g). U.S. EPA's SW-846 also is available on the internet at "http://www.epa.gov/epaoswer/hazwaste/test/main.htm". The assumptions used to quantify releases of chemicals from liquid-phase wastes in surface impoundments and solid-phase wastes in landfills are described in Sections 2.1.1 and 2.1.2, respectively.

2.1.1 Liquid-Phase Waste (Surface Impoundment)

The method used to compute releases from liquid-phase wastes assumes that liquid industrial wastes are disposed of in an unlined surface impoundment with a sludge or sediment layer at the base of the impoundment. The determination of whether a waste is a liquid waste is made using U.S. EPA-approved Test Method 9095, referred to as the Paint Filter Test (U.S. EPA 1997g). The four parameters used to characterize a surface impoundment are (1) the area of the impoundment, (2) the ponding depth of the liquid in the impoundment, (3) the thickness of a relatively low-permeability sludge or sediment layer at the base of the surface impoundment, and (4) the hydraulic conductivity of this sludge or sediment layer. Additional information regarding characterization and modeling of liquid wastes disposed of in surface impoundments is provided in the "EPACMTP Background Document and User's Guide" (U.S. EPA 1996b).

2.1.2 Solid-Phase Waste (Landfill)

The method used to compute releases from solid-phase wastes assumes that solid wastes are disposed of in a Subtitle D landfill and are covered with a 2-foot-thick native soil layer. It is assumed that the Subtitle D landfill is unlined or that any liner at the base of the landfill will eventually completely fail. The two parameters used to characterize landfills are (1) area and (2) depth (the thickness of the waste layer). Data to characterize landfills were obtained from a nationwide survey of industrial Subtitle D landfills (Westat 1987). Parameters and assumptions used to estimate infiltration of leachate from a landfill are provided in the "EPACMTP Background Document and User's Guide" (U.S. EPA 1996b).

2.2 RELEASES TO GROUNDWATER (EPACMTP)

This section describes the method used to compute the release and transport of chemicals from a waste management unit to the subsurface and their subsequent transport through the unsaturated and saturated zones to a theoretical downgradient receptor well.

2.2.1 Methodology for Estimation of Waste Constituent Concentration in Groundwater

This section summarizes the method used to calculate groundwater exposure concentrations resulting from release of waste constituents into the subsurface from two waste management units: surface impoundments and landfills. The exposure concentration is evaluated at a hypothetical groundwater-drinking water well located a specific distance from the downgradient edge of the waste management unit. This well is referred to hereafter as the receptor well, and the exposure concentration measured at that well is referred to as the groundwater receptor well concentration (C_{gw}). The model used was the EPACMTP (U.S. EPA 1996a). Receptor well concentrations for both carcinogens and noncarcinogens for finite-source degraders and nondegraders were determined with this model.

2.2.2 Overview of EPACMTP

The EPACMTP is a fate and transport model that simulates one-dimensional, vertically downward flow and transport of contaminants in the unsaturated zone beneath a waste disposal unit as well as two-dimensional or three-dimensional groundwater flow and contaminant transport in the underlying saturated zone. The model accounts for the following processes affecting contaminant fate and transport: advection, hydrodynamic dispersion, linear or nonlinear equilibrium sorption, chained first-order decay reactions, and dilution from recharge in the saturated zone. The EPACMTP incorporates a Monte Carlo module that allows assessment of the uncertainty associated with receptor well concentrations that result from variations in the model input parameters.

The EPACMTP consists of four major components:

- A module that performs one-dimensional analytical and numerical solutions for water flow and contaminant transport in the unsaturated zone beneath a waste management unit

- A numerical module for steady-state groundwater flow subject to recharge from the unsaturated zone

- A module of analytical and numerical solutions for contaminant transport in the saturated zone

- A Monte Carlo module for assessing the effect of the uncertainty resulting from variations in model parameters on predicted receptor well concentrations

The subsurface as modeled with the EPACMTP consists of an unsaturated zone beneath a waste unit and an underlying water table aquifer. Contaminants move vertically downward through the unsaturated zone to the water table. The EPACMTP allows simulation of flow and transport in the unsaturated zone and in the saturated zone, either separately or combined.

The EPACMTP is capable of simulating the fate and transport of dissolved contaminants from a point of release at the base of a waste disposal unit, through the unsaturated zone and underlying groundwater, to a receptor well at an arbitrary downstream location in the aquifer. The model accounts for the following mechanisms affecting contaminant migration: transport by advection and dispersion, retardation resulting from reversible linear or nonlinear equilibrium adsorption onto the soil and aquifer solid phase, and biochemical degradation processes. The latter may involve chain decay reactions if the contaminant or contaminants of concern form a decay chain. As is true of any model, the EPACMTP is based on a number of simplifying assumptions that make the model easier to use and that ensure its computational efficiency. The major simplifying assumptions used in the EPACMTP are summarized below.

1. **Soil and Aquifer Medium Properties**. 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. The EPACMTP does not account for the presence of preferential pathways such as fractures and macropores. Although the aquifer properties are assumed to be uniform, the model does allow for anisotropy in hydraulic conductivity.

2. **Flow in the Unsaturated Zone.** Flow in the unsaturated zone is assumed to be steady-state, one-dimensional, vertical flow from beneath the source toward the water table. The lower boundary of the unsaturated zone is assumed to be the water table. The flow in the unsaturated zone is assumed to be predominantly gravity-driven, and therefore the vertical flow component accounts for most of the fluid flux between the source and the water table. The flow rate is assumed to be determined by the long-term average infiltration rate through the waste management unit. In surface impoundments, the flow rate is assumed to be determined by the average depth of ponding.

3. **Flow in the Saturated Zone.** The saturated zone module of the EPACMTP is designed to simulate flow in an unconfined aquifer with constant saturated thickness. The model assumes regional flow in a horizontal direction with vertical disturbance resulting from recharge and infiltration from the overlying unsaturated zone and waste disposal facility. The lower boundary

of the aquifer is assumed to be impermeable. Flow in the saturated zone is assumed to be steady-state. The EPACMTP accounts for different recharge rates beneath and outside the source area. Groundwater mounding beneath the source is represented in the flow system by increased head values at the top of the aquifer. This approach is reasonable as long as the height of the mound is small relative to the thickness of the saturated zone.

4. **Transport in the Unsaturated Zone**. Contaminant transport in the unsaturated zone is assumed to occur by advection and dispersion. The unsaturated zone is assumed to be initially contaminant-free, and contaminants are assumed to migrate vertically downward from the disposal facility. The EPACMTP can simulate both steady-state and transient transport in the unsaturated zone with single-species or multiple-species chain decay reactions and with linear or nonlinear sorption.

5. **Transport in the Saturated Zone**. Contaminant transport in the saturated zone is assumed to be a result of advection and dispersion. The aquifer is assumed to be initially contaminant-free, and contaminants are assumed to enter the aquifer only from the unsaturated zone immediately beneath the waste disposal facility, which is modeled as a rectangular, horizontal plane source. The EPACMTP can simulate both steady-state and transient three-dimensional transport in the aquifer. For steady-state transport, the contaminant mass flux entering at the water table must be constant with time; for the transient case, the flux at the water table may be constant or may vary as a function of time. The EPACMTP can simulate the transport of a single species or multiple species, chain decay reactions, and linear sorption.

6. **Contaminant Phases**. The EPACMTP assumes that the dissolved phase is the only mobile phase and disregards interphase mass transfer processes other than adsorption onto the solid phase. The model does not account for volatilization in the unsaturated zone; this is a conservative approach for volatile chemicals. The model also does not account for the presence of a nonaqueous-phase liquid (such as oil) or for transport in the gas phase. When a mobile oil phase is present, significant contaminant migration may occur within it, and the EPACMTP may underestimate the movement of hydrophobic chemicals.

7. **Chemical Reactions**. The EPACMTP computes chemical reactions involving adsorption and decay processes. The EPACMTP assumes that sorption of organic compounds in the subsurface is represented by linear adsorption isotherms in both the unsaturated and saturated zones. It is assumed that adsorption of contaminants onto the soil or aquifer solid phase occurs instantaneously and is entirely reversible. The effect of geochemical interactions is especially important in fate and transport analyses of metals. For simulation of metals, the EPACMTP uses sorption isotherms generated by MINTEQA2 (Allison and others 1991). MINTEQA2 generates concentration-dependent effective partition coefficients for various combinations of geochemical conditions. This procedure is described in the background document for modeling of metal transport (U.S. EPA 1991d, 1996d).

The EPACMTP also accounts for chemical and biological transformation processes. All transformation reactions are represented by first-order decay processes. An overall decay rate is specified for the model; therefore, the model cannot explicitly consider the separate effects of multiple degradation processes such as oxidation, hydrolysis, and biodegradation. The user must determine the overall, effective decay rate when multiple decay processes are to be represented. To maximize its flexibility, the EPACMTP has the capability of determining the overall decay rate from chemical-specific hydrolysis constants using soil and aquifer temperature and pH values. The EPACMTP assumes that reaction stoichiometry is prescribed for scenarios involving chain decay reactions. The speciation factors are specified as constants by the user (see the "EPACMTP Background Document and User's Guide," U.S. EPA 1996b). In reality, these coefficients may change as functions of aquifer conditions (for example, temperature and pH), concentration levels of other chemical components, or both.

2.2.2.1 Contaminant Release and Transport Scenario

Two source release scenarios are considered in the EPACMTP: continuous (infinite) and finite-source. Only the finite-source scenario is considered for delisting. For finite-source scenarios, the release of contaminants occurs over a finite period of time, after which the leachate concentration becomes zero (that is, all the contaminants in the waste disposed of in the waste management unit have leached out).

Each type of waste management scenario is described by a relatively small number of parameters. The differences between waste management units are represented by different values or frequency distributions of the source-specific parameters. Source-specific stochastic parameters used by the EPACMTP for landfills include the capacity and dimensions of the waste management unit, the leachate concentration, infiltration and recharge rates, pulse duration, the fraction of hazardous waste in the waste management unit, the density of the waste, and the concentration of the chemical constituent in the hazardous waste. The source-specific stochastic parameters used for surface impoundments include the area, the ponding depth (such as the depth of liquid in the impoundment), and the thickness and hydraulic conductivity of the sludge or sediment layer at the bottom of the impoundment. Data on the areas, volumes, and locations of waste management units were obtained from the 1987 U.S. EPA survey of industrial Subtitle D waste facilities in the United States (Westat 1987). Derivation of the parameters for each type of waste management unit is described in the " EPACMTP Background Document and User's Guide" (U.S. EPA 1996b).

For finite-source scenarios, simulations are performed for transient conditions, and the source is assumed to be a pulse of finite duration. In the case of landfills, the pulse duration is based on the initial amount of contaminant in the landfill, infiltration rate, landfill dimensions, waste and leachate concentration, and waste density (U.S. EPA 1996c). For surface impoundments, the duration of the leaching period is determined by the waste management unit's lifetime (the default value is 20 years for surface impoundments). For a finite-source scenario, the model can calculate either the peak receptor well concentration for noncarcinogens or an average concentration over a specified period for carcinogens. The finite-source methodology in the EPACMTP is discussed in detail in the finite-source background document (U.S. EPA 1996c).

2.2.2.2 EPACMTP Modeling Assumptions and Input Parameters

Specific EPACMTP modeling assumptions (in addition to the simplifying assumptions discussed in Section 2.2.1) are summarized in Table 2-1. This table also provides information on important input parameters as well as on their data sources or details. Overall, EPACMTP input parameters can be organized in the following four groups:

- Source-specific parameters
- Chemical-specific parameters
- Unsaturated zone-specific parameters
- Saturated zone-specific parameters

For delisting, the EPACMTP is run in Monte Carlo mode, and the source-, chemical-, unsaturated zone-, and saturated-zone specific parameters are represented by probability distributions reflecting variations on a national or a regional level. Specific capabilities and requirements associated with running the EPACMTP in the Monte Carlo mode are presented in Chapter 3 of the "EPACMTP User's Guide" (U.S. EPA 1997e). The Monte Carlo analysis determines the effect of the possible range of the input parameter of concern on the receptor well concentration. Output values produced for each iteration are sorted and ranked from highest to lowest in order to obtain a probabilistic distribution of receptor well concentrations. The different groups of input parameters are summarized below. For chemicals that were not modeled using the EPACMTP fate and transport model, the most conservative DAF was assigned (i.e., DAF = 18).

TABLE 2-1

EPACMTP MODELING ASSUMPTIONS AND INPUT PARAMETERS

Modeling Assumptions	
Modeling Element	Description or Value
Management Scenario	Landfill Surface impoundment
Modeling Scenario	Finite-source Monte Carlo; depleting source for organics, constant concentration pulse source for metals
Exposure Evaluation	Downgradient groundwater receptor well; maximum well concentration of noncarcinogens during modeling period, maximum 30-year average well concentration of carcinogens; 10,000-year exposure period
Regulatory Protection Level	90 percent
Source-Specific Parameters	
Parameter	Description or Value
Waste Unit Area Waste Unit Volume Infiltration Rate Landfill Surface Impoundment	Derived User-specified Site-based, derived from water balance using HELP model Site-based, derived from impoundment depth using Darcy's law
Leaching Duration Landfill Surface Impoundment	 Derived, continues until all constituents have leached out 20 years (operational life of unit)
Chemical-Specific Parameters	
Parameter	Description and Source
Decay Rate Organic Constituents Metals	 Hydrolysis rate constants compiled by U.S. EPA ORD No decay
Sorption Organic Constituents Metals	 K_{oc} constants compiled by U.S. EPA ORD MINTEQA2 sorption isotherm coefficients (K_d) for Pb, Hg (II), Ni, Cr (III), Ba, Cd, Ag, Zn, Cu (II), Be]; pH-dependent isotherm coefficients for As (III), Cr (VI), Se (VI), Th

Unsaturated Zone-Specific Parameters	
Parameter	Description and Source
Depth to Groundwater	Site-based, from API and USGS hydrogeologic database
Soil Hydraulic Parameters: Fraction Organic Carbon Bulk Density	U.S. EPA ORD data based on national distribution of three soil types (sandy loam, silt loam, silty clay loam)
Saturated Zone-Specific Parameters	
Parameter	Description and Source
Recharge Rate	Site-based, derived from regional precipitation and evaporation data and soil type
Aquifer Thickness	Site-based, from API and USGS hydrogeologic database
Hydraulic Conductivity	Site-based, from API and USGS hydrogeologic database
Hydraulic Gradient	Site-based, from API and USGS hydrogeologic database
Porosity	Effective porosity derived from national distribution of aquifer particle diameter
Bulk Density	Derived from porosity
Dispersivity	Derived from distance to receptor well
Groundwater Temperature	Site-based, from USGS regional temperature map
Fraction Organic Carbon	National distribution, from U.S. EPA STORET database
pH	National distribution, from U.S. EPA STORET database
Receptor Well Parameters	
Well Element	Description and Source
Radial Distance from Waste Management Unit	Nationwide distribution, from U.S. EPA OSW database
Angle Off-Center	Uniform within ± 90° from plume center line (no restriction within plume)
Depth of Intake Point	Uniform throughout saturated thickness of aquifer

Notes:

API = American Petroleum Institute
HELP = Hydrologic Evaluation of Landfill Performance
ORD = U.S. EPA Office of Research and Development
STORET = Database Utility for STORage and RETrieval of Chemical, Physical, and Biological Data for Water Quality
USGS = U.S. Geological Survey

Source-Specific Parameters. Source-specific parameters give information about the waste management unit. The most sensitive parameters in this group include the waste management unit's area, depth, and volume and the infiltration (U.S. EPA 1996b and 1996c). It is important to note the difference in leaching duration for each waste management unit (see Table 2-1).

Chemical-Specific Parameters. Chemical-specific parameters describe the degradation (decay), adsorptive, and diffusive characteristics of each of the chemical species being simulated. Thus, the most important variables accounting for chemical characterization are the hydrolysis rate (λ) (for organic constituents), the normalized distribution coefficient for organic carbon (K_{oc}) (for sorption estimation for organics), and the absorption isotherm coefficient (K_d) (for linear and nonlinear sorption estimation for metals). K_{oc} is also referred to as the organic carbon partition coefficient, and K_d is referred to as the effective partition coefficient.

Unsaturated Zone-Specific Parameters. The unsaturated zone is the first subsurface level below the waste management unit. It is unsaturated with groundwater, and therefore it does not have a water table. The available groundwater in this zone enables contaminants to move in the subsurface. The simulated process of fate and transport in this zone is based on the following assumptions:

- The flow of the fluid phase is isothermal and is governed by Darcy's law.
- The flow is one-dimensional, vertically downward, and steady-state.
- The fluid is slightly compressible and homogeneous.
- The dynamics of a second phase (such as air or nonaqueous liquid) can be disregarded.

The solutions for transient and steady-state transport in the unsaturated zone are based on the following assumptions.

- The leachate concentration entering the soil is either constant (with a finite or infinite duration) or decreasing with time following a first-order decay process.
- Sorption of contaminants onto the soil solid phase is described by a linear or nonlinear (Freundlich) equilibrium isotherm.

The required input parameters for this group are depth to groundwater and soil hydraulic parameters, including fraction organic carbon (f_{oc}) and bulk density.

Saturated Zone-Specific Parameters. The saturated zone is located below the unsaturated zone and beneath the water table. Groundwater flow in the saturated zone is simulated using a steady-state solution for predicting hydraulic head and Darcy velocities in a constant-thickness groundwater system subject to infiltration and recharge along the top of the aquifer and a regional (ambient) gradient defined by upstream and downstream head boundary conditions.

Simplifying assumptions used to simulate contaminant transport in the saturated zone are as follows:

- The flow field is steady-state.
- The aquifer is homogeneous and initially contaminant-free.
- Adsorption onto the solid phase is described by a linear or nonlinear equilibrium isotherm.
- Chemical and/or biochemical degradation of the contaminant can be described as a first-order process.
- For a multicomponent decay chain, the number of component species (parent and daughters) does not exceed seven.
- The mass flux of contaminants through the source is either constant or controlled by first-order decay until all mass has been released from the source.
- The chemical is dilute and present in the aqueous and aquifer solid phases only.

The required input parameters for this group are recharge rate, saturated thickness, hydraulic conductivity, hydraulic gradient, and groundwater temperature.

The receptor well location is also a saturated zone-specific parameter. As the location at which potential exposure to groundwater is measured, it can be anywhere downgradient of the waste management unit, within the areal extent of the contaminant plume, and/or along the contaminant plume center line. The receptor well location is determined using the following parameters:

- Radial distance from the waste management unit
- Angle off-center
- Depth of the intake point (z as well depth within the saturated zone)

The EPACMTP fate and transport model was used to determine the degree of dilution and attenuation that a chemical will undergo as it leaches from a waste management unit and is transported in the subsurface, into the saturated zone, and to a theoretical downgradient receptor well. The change in chemical concentration during the chemical's transport from the leachate to the receptor well is the DAF. The EPACMTP used Monte Carlo probability analyses to compute a DAF for each chemical in the DRAS chemical library. The EPACMTP determines the groundwater concentration at the theoretical drinking water well that is the 90th percentile of all predicted concentrations (C_{gw}) resulting from the Monte Carlo analyses. The DAF is then calculated as the function of the initial (C_L)waste concentration divided by the predicted groundwater concentration at the 90th percentile as shown in Equation 2-1.

$$DAF = \frac{C_L}{C_{gw}} \quad \textbf{(2-1)}$$

where:

			Default
DAF	=	dilution attenuation factor (unitless)	calculated
C_L	=	leachate concentration (that is, TCLP) (milligram per liter [mg/L])	waste- and chemical-specific
C_{gw}	=	waste constituent concentration in groundwater (mg/L)	computed with EPACMTP

If the maximum allowable concentration of a chemical at the groundwater receptor well *(C_{gw})* is assumed to be an HBN, Equation 2-1 can be rearranged, and the HBN (or a maximum contaminant level [MCL], if available) can be substituted for the receptor well concentration (C_{gw}) to back-calculate a maximum allowable leachate concentration (C_{LMAX}) for the ingestion exposure pathway. Equation 2-2 reflects this approach.

$$C_{LMAX} = HBN \cdot DAF \quad \textbf{(2-2)}$$

where:

			Default
C_{LMAX}	=	maximum allowable leachate concentration (TCLP concentration) (mg/L)	calculated
HBN	=	health-based number (or MCL) (mg/L)	chemical-specific

DAF	=	dilution attenuation factor (unitless)	computed with EPACMTP

2.2.2.3 Modification of EPACMTP to Derive Waste Volume-Specific DAFs

The EPACMTP developed for the proposed HWIR (U.S. EPA 1995a) requires as input the area of the waste management unit. Application of the EPACMTP to the U.S. EPA Region 6 Delisting Program allows evaluation of specific waste volumes. Therefore, to generate waste volume-specific DAFs, the EPACMTP was modified to convert a user-specified waste volume to a waste management unit area. This was accomplished by modifying the EPACMTP to divide the user-specified waste volume by a waste management unit depth randomly drawn from a regional site database. The regional site database contains waste management unit depth and area data obtained from a national survey of Subtitle D industrial facilities performed by the U.S. EPA Office of Policy Planning and Implementation (OPPI) (hereinafter referred to as the OPPI survey data) (Westat 1987; U.S. EPA 1988b). The OPPI survey data were compiled along with hydrogeologic data in the regional site database. The data for each waste management scenario are presented in Appendix A of the "EPACMTP Background Document and User's Guide" (U.S. EPA 1996b). For a complete description of the regional site-based modeling approach, refer to the "EPACMTP Background Document for Finite Source Methodology for Chemicals with Transformation Products and Implementation for the Hazardous Waste Identification Rule" (U.S. EPA 1996c).

Rather than compute DAFs for each of 192 chemicals for a range of waste volumes for landfills and surface impoundments, a method was developed to scale the DAFs based on a specific waste volume to the DAFs computed for the proposed HWIR (U.S. EPA 1995a). The EPACMTP fate and transport model was used to compute waste volume-specific DAF scaling factors for the landfill and surface impoundment waste management scenarios (U.S. EPA 1996a). Rather than compute the DAF scaling factors for fixed increments over the entire range of waste volumes, preliminary runs were made to plot the DAF scaling factors in order to determine the shape of the curve that predicts the DAF scaling factor as a function of waste volume. Initial runs indicated a nonlinear relationship of DAF scaling factors as a function of waste volume for each waste management scenario. Therefore, most of the EPACMTP analyses were performed for the range of waste volumes in which the change in DAF scaling factor was the greatest (that is, the lower range of waste volumes). This approach allowed development of regression equations to predict the DAF scaling factor as a function of waste volume for each waste management scenario. The DAF scaling factors and the regression equations developed for the landfill and surface impoundment waste management scenarios are presented in the following sections.

2.2.2.3.1 Landfills

DAF scaling factors were developed for landfill waste volumes ranging from 10,000 to 1,000,000 cubic yards yd^3, based on the range of waste volumes encountered in the OPPI survey (U.S. EPA 1988b). The landfill DAF scaling factors are plotted in Figure 2-1. Figure 2-1 indicates that the DAF scaling factor is approximately 1.0 for waste volumes greater than 200,000 yd^3. U.S. EPA Region 6 did not want the DAFs used for delisting to be less than the HWIR DAFs, and therefore did not consider DAF scaling factors less than 1.0. Therefore, for landfill waste volumes greater than 200,000 yd^3, the waste volume-specific DAF is equal to the DAF computed for the proposed HWIR (U.S. EPA 1995a).

Equation 2-3 can be used to determine the DAF scaling factor (DAF_{sf}) as a function of waste volume for landfilled wastes.

$$DAF_{sf} = 6{,}152.7 \cdot (V)^{-0.7135} \qquad \textbf{(2-3)}$$

where:

			Default
DAF_{sf}	=	DAF scaling factor (unitless)	computed
V	=	volume of waste (yd^3)	delisting petition-specific

The correlation coefficient of this regression equation is 0.99, indicating that changes in the DAF scaling factor, are explained by changes in the waste volume and that the DAF scaling factor can be predicted with confidence as a function of waste volume.

2.2.2.3.2 Surface Impoundments

DAF scaling factors were developed for surface impoundment waste volumes ranging from 2,000 to 1,000,000 yd^3 based on the OPPI survey data (U.S. EPA 1988b). The surface impoundment DAF scaling factors are plotted in Figure 2-2. Figure 2-2 also indicates that the DAF scaling factor is approximately 1.0 for waste volumes greater than 100,000 yd^3.

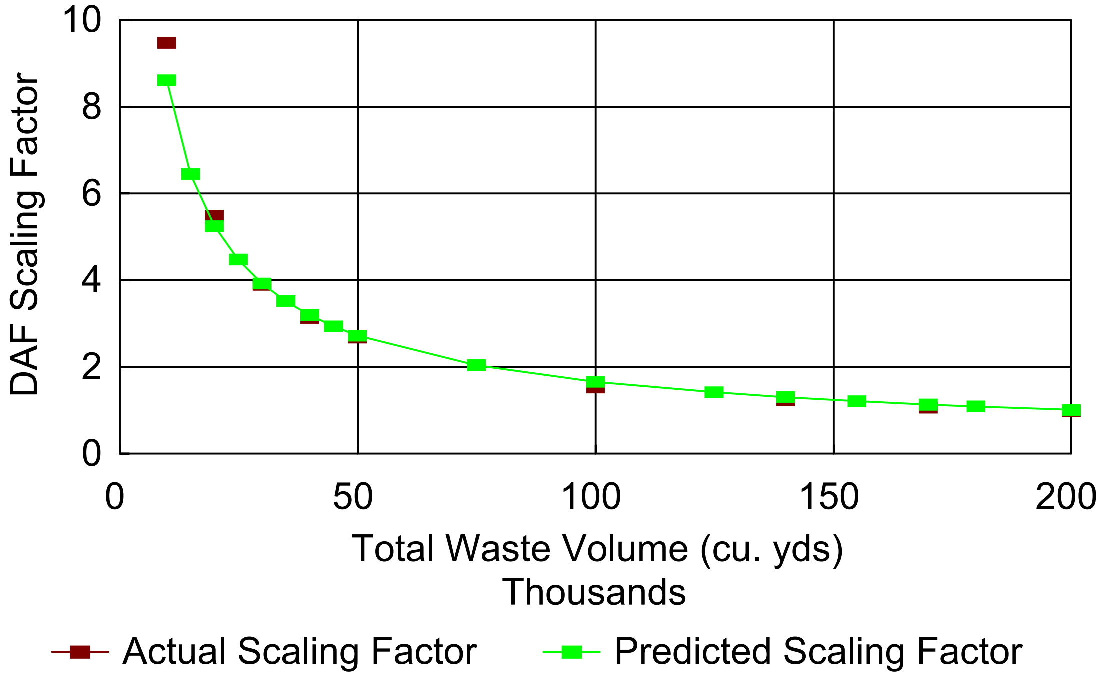

Figure 2-1. DAF Scaling Factors for Landfills

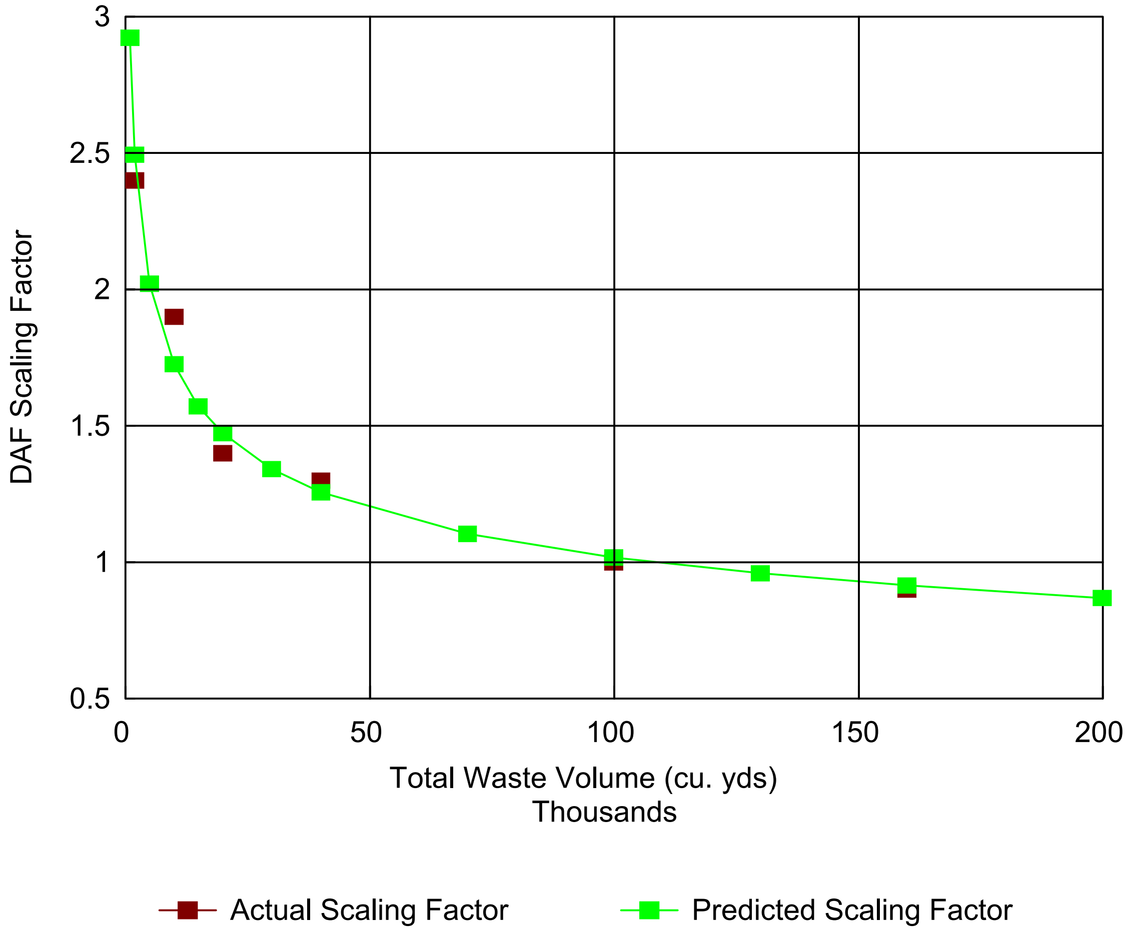

Figure 2-2. DAF Scaling Factors for Surface Impoundments

For surface impoundment waste volumes greater than 100,000 yd^3, the DAF scaling factor is equal to 1.0, and the waste volume-specific DAF is the same as the proposed HWIR DAF (U.S. EPA 1995a).

Equation 2-4 can be used to determine the DAF_{sf} as a function of waste volume for surface impoundments.

$$DAF_{sf} = 14.2 \cdot (V)^{-0.2288} \quad \textbf{(2-4)}$$

where:

			Default
DAF_{sf}	=	DAF scaling factor (unitless)	computed
V	=	volume of waste (yd^3)	delisting petition-specific

The correlation coefficient of this regression equation is 0.99, indicating a good fit of this line to the data points and that the DAF scaling factor for surface impoundments can be predicted as a function of waste volume with a high level of confidence.

2.2.3 Calculation of Groundwater Waste Constituent Concentration at the POE

As discussed above, the EPACMTP model estimates fate and transport of chemicals in groundwater. Within this medium, three human health exposure pathways are evaluated by the U.S. EPA Region 6 Delisting Program: direct ingestion, dermal absorption, and shower inhalation. This section describes how the waste constituent concentration at the groundwater POE is calculated.
To adjust for the petitioned waste volume, the EPACMTP DAF is multiplied by the DAF_{sf} determined as described above. The product is defined as the waste volume-adjusted DAF (DAF_{va}), which is specific for each waste constituent. Equation 2-5 is used for this calculation.

$$DAF_{va} = DAF_{sf} \cdot DAF \quad \textbf{(2-5)}$$

where:

			Default
DAF_{va}	=	waste volume-adjusted DAF (unitless)	calculated
DAF_{sf}	=	DAF scaling factor (unitless)	Equation 2-3 (landfills) or 2-4 (surface impoundments)

DAF	=	dilution attenuation factor (unitless)	Computed with EPACMTP

Using Equation 2-6, the TCLP leachate concentration of a petitioned waste constituent is divided by the waste volume-adjusted DAF to obtain the predicted groundwater receptor well concentration of the constituent.

$$C_{gw} = \left(\frac{TCLP}{DAF_{va}}\right) \qquad \textbf{(2-6)}$$

where:

			Default
C_{gw}	=	waste constituent concentration in groundwater (mg/L)	calculated
TCLP	=	TCLP concentration of waste constituent (mg/L)	waste constituent-specific
DAF_{va}	=	waste volume-adjusted DAF (unitless)	Equation 2-5

The waste constituent concentration in groundwater calculated using Equation 2-6 is used to determine exposure point concentrations for the three groundwater exposure pathways identified above: direct ingestion, dermal absorption, and shower inhalation. The exposures, in turn, are used to compute the risk and hazard associated with exposure to the waste constituent via each of these three pathways (see Chapter 4). It should be noted here that the EPACMTP DAFs are very high (that is, greater than 1,000,000) for some waste constituents. For these constituents, a TCLP concentration greater than 10,000 mg/L may result in risk estimates that are well within the target risk range even though the TCLP concentration may exceed a selected maximum waste target level. Therefore, it may be necessary to define a cap for the maximum allowable concentration of a waste constituent (see Section 4.2.3.2).

2.3 RELEASES TO SURFACE PATHWAYS

This section describes the equations used to predict releases of chemicals from landfill and surface impoundment waste management units via the air and surface water pathways. The equations predict medium-specific (soil and air) concentrations of the waste constituent or constituents at the POE. The release of chemicals to air via particulate matter from landfills is discussed in Section 2.3.1, the release of volatile chemicals to air from landfills and surface impoundments is discussed in Section 2.3.2, and the methodology for calculating surface water concentrations for waste constituents eroded from landfills is presented in Section 2.3.3.

2.3.1 Calculation of Waste Constituent Concentration in Air — Particulates

U.S. EPA Region 6 considers exposure to airborne particulate hazardous constituents released from wastes disposed of in landfills to be a function of (1) inhalation of particulates and their absorption into the lungs at the POE and (2) air deposition of particulates and subsequent ingestion of the soil-waste mixture at the POE. To address inhalation and ingestion of particulates, U.S. EPA Region 6 calculates particulate emissions resulting from wind erosion of soil-waste surfaces and from vehicular traffic and waste loading and unloading. To estimate the respirable particulate emissions resulting from wind erosion of surfaces with an infinite source of erodible particles, U.S. EPA Region 6 used the methodology documented in "Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites (RAEPE)" (U.S. EPA 1985a). To calculate the dust and particulate emissions resulting both from vehicular traffic and from waste loading and unloading operations at a facility, U.S. EPA Region 6 used the methodologies documented in "Compilation of Air Pollutant Emission Factors, Volume 1: Stationary Point and Area Sources" (AP-42) (U.S. EPA 1985b).

Particulate emission rates computed using these methodologies were summed and entered in the Ambient Air Dispersion Model (AADM), a steady-state, Gaussian plume dispersion model developed by U.S. EPA to predict the concentrations of constituents 1,000 feet downwind of a hypothetical land disposal facility (U.S. EPA 1985c). When evaluating delisting petitions, U.S. EPA Region 6 assumes conservative values for all variables that are likely to influence the potential for soil erosion, including wind velocity and vegetative cover. U.S. EPA Region 6, however, modified the AADM assumptions regarding unit dimensions to more closely resemble a landfill's dimensions.

The method used to calculate total respirable particulate emissions from a landfill is described in Section 2.3.1.1. The methods used to calculate the average downwind concentrations of particulates and the downwind concentrations of respirable particulate emissions are described in Section 2.3.1.2. Calculation of air deposition rates and resulting soil concentrations is described in Section 2.3.1.3.

2.3.1.1 Estimation of Particulate Emissions

U.S. EPA Region 6 calculates the mass flux (the source term or amount of waste material that becomes airborne) associated with particulate emissions using the methodology presented in RAEPE (U.S. EPA 1985a). The mass flux can be calculated using Equations 2-7 and 2-8.

$$Q_p = \alpha \cdot E_T \cdot \frac{1{,}000\ mg}{1\ g} \cdot \frac{1\ hr}{3{,}600\ s} \qquad \textbf{(2-7)}$$

where:

			Default
Q_p	=	emission rate of waste constituent particulates (milligrams per second [mg/s])	calculated
α	=	mass fraction of constituent in waste (unitless)	waste-specific (equal to total concentration in waste; mg/mg)
E_T	=	total emission rate of particulates that may be inhaled (gram/hour [g/hr])	Equation 2-8

and:

$$E_T = E_w + E_v + E_l \qquad \textbf{(2-8)}$$

where:

			Default
E_T	=	total emission rate of particulates that may be inhaled (g/hr)	calculated
E_w	=	particulate emission rate from wind erosion (g/hr)	Equation 2-9
E_v	=	particulate emission rate from vehicular traffic (g/hr)	Equation 2-17
E_l	=	particulate emission rate from waste loading and unloading operations (g/hr)	Equation 2-21

The equations used to compute emissions resulting from wind erosion (E_w), vehicular traffic (E_v), and waste loading and unloading (E_l) are presented in the following sections.

Wind Erosion Emissions

Wind erosion emissions of respirable particulates (those smaller than 10 microns [µm] in diameter)[a] can be calculated using Equation 2-9.

$$E_{w10} = 0.036 \cdot (1 - V_f) \cdot \left(\frac{U}{U_t}\right)^3 \cdot F(X) \cdot A_{exposed} \qquad \textbf{(2-9)}$$

[a] Particles less than or equal to 10 µm in aerodynamic diameter are defined as the respirable fraction. Refer to discussions in RAEPE (U.S. EPA 1985a), AP-42 (U.S. EPA 1985b), and the "Draft Superfund Exposure Assessment Manual" (U.S. EPA 1986b).

where:

			Default
E_{w10}	=	wind erosion emission rate for particulates up to 10 μm (g/hr)	calculated
V_f	=	fraction of disposal site covered with vegetation (unitless)	0 (U.S. EPA 1994a)
U	=	mean annual wind speed (meters/second [m/s])	4 (discussed below)
U_t	=	threshold value of wind speed at 7 m (m/s)	5.44 (Equation 2-10)
$F(X)$	=	dimensionless function obtained from a plot in RAEPE (U.S. EPA 1985a)	1.33 (Appendix B, Figure B-6)
$A_{exposed}$	=	area of the waste management unit exposed (m^2)	Equation 2-15

For the landfill waste disposal scenario, U.S. EPA Region 6 assumed that no vegetative cover is present, thereby assuming enhanced erodibility of soil or waste. Therefore, the fraction of the disposal site covered with vegetation (Vf) is equal to 0. The mean annual wind speed is assumed to be 4 m/s. This value represents the average of the wind speeds registered at U.S. climatological stations as documented in Table 4-1 of RAEPE (U.S. EPA 1985a). This value is also assumed to be associated with climate stability class D.[b]

The threshold value of wind speed, U_t , can be derived in the following manner as described in RAEPE (U.S. EPA 1985a) using Equation 2-10:

- The waste is assumed to exhibit a particle size of 0.2 millimeter (mm) (a typical size for fine sand). The wind erosion threshold friction velocity (U_{*t}) used in Equation 2-10 is derived from RAEPE (U.S. EPA 1985a) plots (see Appendix B, Figure B-6 of this DTSD) and is equal to 33 centimeter per second (cm/s).

- A roughness height (Z_o) of 1.0 cm (for a plowed field) is obtained from Appendix B.

- Based on Z_o, a ratio of wind speed at 7 m (U_t) to friction velocity (U_{*t}) can be obtained from Appendix B. In this case, $U_t/U_{*t} = 16.5$.

Therefore:

$$\begin{aligned} U_t &= 33 \ \frac{cm}{s} \cdot 16.5 \cdot \frac{1 \ m}{100 \ cm} \\ &= 5.44 \ \frac{m}{s} \end{aligned} \qquad \textbf{(2-10)}$$

[b] Stability class is a meteorological classification of atmospheric properties as those properties relate to dispersion of airborne materials. The classes range from A, extremely unstable, to F, moderately stable. The coefficients defined by the stability class are used in Equation 2-9 to calculate the downwind dispersion.

Using the values identified above, *X* is calculated as shown in Equation 2-11.

$$\begin{aligned} X &= 0.886 \cdot \left(\frac{U_t}{U}\right) \\ &= 0.886 \cdot \frac{5.44\ m/s}{4\ m/s} \\ &= 1.205 \end{aligned} \qquad \textbf{(2-11)}$$

Given the value of *X* computed using Equation 2-11, an F(X) value of 1.33 is obtained from the graph provided in RAEPE (U.S. EPA 1985a) (see Appendix B, Figure B-6 of this DTSD).

The landfill area is determined using an equation developed for the U.S. EPA OSW Hazardous Waste Delisting Program (U.S. EPA 1991b). Waste volume is correlated with disposal unit area by means of a regression equation developed from national data on disposal unit dimensions obtained in the OPPI survey. A method to convert waste volume to disposal unit area was developed for both landfills and surface impoundments (U.S. EPA 1991b). Equation 2-12 below presents the regression equation to determine the disposal unit area for a one-time delisting of landfill waste or surface impoundment waste for a RCRA delisting.

$$Ln(A) = -5.95477 + 0.676889 \cdot Ln(V) \qquad \textbf{(2-12)}$$

where:

			Default
A	=	area of the waste management unit (acres)	calculated
V	=	volume of waste (yd^3)	delisting petition-specific

The area of the waste management unit is based on the volume of waste reported in the petition. U.S. EPA Region 6 assumes that a petitioned waste is disposed of in one Subtitle D facility over a 20-year period (U.S. EPA 1988b). Based on maximum annual waste volumes and assuming waste disposal into the same landfill or surface impoundment over a 20 year period, Equation 2-13 below is used to determine the disposal unit area for multi-year RCRA delistings to a landfill or surface impoundment.

$$Ln(A) = -5.95477 + 0.676889 \cdot Ln(V \cdot 20\ yr) \qquad \textbf{(2-13)}$$

where:

			Default
A	=	area of the waste management unit (acres)	calculated
V	=	volume of waste (yd^3/year)	delisting petition-specific

U.S. EPA Region 6 believes that, at most, a month's (30 days') worth of waste would be uncovered at any one time. Therefore, the fraction of the area in the landfill that would be exposed and available as a source for particulate emissions ($F_{exposed}$) is calculated as shown in Equation 2-14.

$$F_{exposed} = \frac{30\,days}{(365\,days/year \cdot 20\,years)} = 0.0041 \qquad \textbf{(2-14)}$$

This fraction exposed is used to compute the exposed area ($A_{exposed}$) and the site width as shown in Equation 2-15.

$$A_{exposed} = F_{exposed} \cdot A\ \frac{4{,}047\ m^2}{acre} \qquad \textbf{(2-15)}$$

where:

			Default
$A_{exposed}$	=	area of the waste management unit exposed (m^2)	calculated
$F_{exposed}$	=	fraction of the area exposed (unitless)	Equation 2-14
A	=	area of the waste management unit (acres)	Equation 2-12 (one-time) or 2-13 (multiyear)

Although particulates greater than 10 µm in size generally are not considered respirable, U.S. EPA Region 6 also calculated the emission rate for particle sizes up to 30 µm in order to assess the potential impact of deposition and ingestion of such particulates (see Section 2.3.1.3). U.S. EPA Region 6 used the distributions of wind-eroded particulates presented in AP-42 (U.S. EPA 1985b) to estimate an emission rate for particulates up to 30 µm in size[c]. Specifically, these distributions indicate that the release rate for particulates up to 30 µm in size should be approximately twice the release rate calculated for particulates 10 µm in size. Equation 2-16 shows this relationship.

[c] Particles less than or equal to 30 µm in diameter can be transported for considerable distances downwind; those larger than 30 µm are likely to settle within a few hundred feet. See AP-42 (U.S. EPA. 1985b and the "Draft Superfund Exposure Assessment Manual" (U.S. EPA 1986b).

$$E_{w30} = E_{w10} \cdot 2 \qquad \textbf{(2-16)}$$

where:

			Default
E_{w30}	=	wind erosion emission rate for particulates up to 30 µm (g/hr)	calculated
E_{w10}	=	wind erosion emission rate for particulates up to 10 µm (g/hr)	Equation 2-9

Vehicle Emissions

U.S. EPA Region 6 used Equation 2-17 as described in AP-42 (U.S. EPA 1985b) to calculate vehicle emissions:

$$E_v = k \cdot 1.7 \cdot \left(\frac{s}{12}\right) \cdot \left(\frac{S}{48}\right) \cdot \left(\frac{W}{2.7}\right)^{0.7} \cdot \left(\frac{w}{4}\right)^{0.5} \cdot \frac{(365 - N_p)}{365} \cdot VKT \qquad \textbf{(2-17)}$$

where:

			Default
E_v	=	vehicle emissions (g/hr)	calculated
k	=	constant; 0.36 for particulates up to 10 µm and 0.8 for particulates up to 30 µm	U.S. EPA 1986b
s	=	percent silt content of waste (unitless)	8 (U.S. EPA 1986b)
S	=	mean vehicle speed (kilometers per hours [km/hr])	24 (U.S. EPA 1986b)
W	=	mean vehicle weight (tons)	15 (U.S. EPA 1986b)
w	=	mean number of wheels per vehicle	6 for W=15 tons (U.S. EPA 1986b)
N_p	=	number of days per year with at least 0.01 inch of precipitation (days/year)	90 (U.S. EPA 1986b)
VKT	=	vehicle kilometers traveled = kilometers/trip x the calculated number of round trips made per day (km-trips/day)	Equation 2-18

Conservatively assuming that 100 round trips are made daily (U.S. EPA 1994a), *VKT* can be calculated using Equation 2-18 by multiplying the disposal site width by 2 and by the number of round trips per day.

$$VKT = Site\ Width \cdot \frac{1\ km}{1{,}000\ m} \cdot 2 \cdot 100\ \frac{trips}{day} \qquad \textbf{(2-18)}$$

where:

			Default
VKT	=	vehicle kilometers traveled (km-trips/day)	calculated
Site Width	=	$\sqrt{A_{exposed}}$ (m)	Equation 2-15

Vehicle emissions, E_{v10} and E_{v30}, can then be calculated using Equation 2-17 as shown in Equations 2-19 and 2-20.

$$
\begin{aligned}
E_{v10} &= 0.36 \cdot 1.7 \cdot \left(\frac{8}{12}\right) \cdot \left(\frac{24}{48}\right) \cdot \left(\frac{15}{2.7}\right)^{0.7} \cdot \left(\frac{6}{4}\right)^{0.5} \cdot \frac{(365-90)}{365} \cdot VKT \\
&= 0.625\ kilogram(kg\frac{)}{day} \cdot 1{,}000\ \frac{g}{kg} \cdot \frac{day}{24\ hr} \cdot VKT
\end{aligned}
\qquad \textbf{(2-19)}
$$

and

$$
\begin{aligned}
E_{v30} &= 0.80 \cdot 1.7 \cdot \left(\frac{8}{12}\right) \cdot \left(\frac{24}{48}\right) \cdot \left(\frac{15}{2.7}\right)^{0.7} \cdot \left(\frac{6}{4}\right)^{0.5} \cdot \frac{(365-90)}{365} \cdot VKT \\
&= 1.39\ \frac{kg}{day} \cdot 1{,}000\frac{g}{kg} \cdot \frac{1\ day}{24\ hr} \cdot VKT
\end{aligned}
\qquad \textbf{(2-20)}
$$

U.S. EPA Region 6 assumes a maximum total annual generation rate for the petitioned waste to estimate a daily disposal amount in tons per day. Therefore, assuming that 15-ton vehicles are used, the number of trips can be determined. U.S. EPA Region 6 realizes that particulate emissions from vehicles are related to the size and weight of the vehicles used for waste transport as well as the number of vehicle trips made. The assumption of small vehicle use (with more trips) is conservative.

Waste Loading and Unloading Emissions

U.S. EPA Region 6 used Equation 2-21, which is based on an AP-42 (U.S. EPA 1985b) methodology, to calculate emissions from waste loading and unloading operations, E_l. In Equation 2-21, E_l is given in units of kilograms of waste released per metric ton of waste disposed of.

$$
E_l = K_p \cdot 0.0009 \cdot \left(\frac{\frac{s}{5} \cdot \frac{U}{2.2} \cdot \frac{DH}{1.5}}{\left(\frac{M}{2}\right)^2 \cdot \left(\frac{Yd}{4.6}\right)^{.33}} \right) \qquad \textbf{(2-21)}
$$

where:

			Default
E_l	=	emissions from waste loading and unloading operations (kg/ton)	calculated
K_p	=	batch drop particle size multiplier (dimensionless): – 0.36 for particulates up to 10 µm – 0.73 for particulates up to 30 µm	U.S. EPA 1986b
s	=	silt content of waste (percent)	8 (U.S. EPA 1986b)
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1986b)
DH	=	drop height of material from truck (m)	2 (U.S. EPA 1986b)
M	=	moisture content of waste (percent)	1 (U.S. EPA 1986b)
Yd	=	dumping device capacity (m^3)	10 (U.S. EPA 1986b)

Based on the petitioned waste volume, emissions from waste loading and unloading operations are calculated for particulates up to 10 and 30 µm in size as shown in Equations 2-22 and 2-23, respectively.

$$E_{l10} = 0.36 \cdot 0.0009 \cdot \left(\frac{\frac{8}{5} \cdot \frac{4}{2.2} \cdot \frac{2}{1.5}}{\left(\frac{1}{2}\right)^2 \cdot \left(\frac{10}{4.6}\right)^{.33}} \right) \tag{2-22}$$

$$= 0.00389 \ \frac{kg}{ton}$$

where:

			Default
E_{l10}	=	waste loading and unloading emission rate of particulates up to 10 µm (kg/ton)	calculated

$$E_{l30} = 0.73 \cdot 0.0009 \cdot \left(\frac{\frac{8}{5} \cdot \frac{4}{2.2} \cdot \frac{2}{1.5}}{\left(\frac{1}{2}\right)^2 \cdot \left(\frac{10}{4.6}\right)^{.33}} \right) \tag{2-23}$$

$$= 0.00789 \ \frac{kg}{ton}$$

where:

			Default
E_{l30}	=	waste loading and unloading emission rate of particulates up to 30 µm (kg/ton)	calculated

To convert the emission rate from kilogram per ton to the required grams per hour, U.S. EPA Region 6 assumes that the waste has an average density of 1.686 g/cm^3 (U.S. EPA 1993a), which is equal to 1.42 yd^3. Using the petitioned waste volume (V) in cubic yards per year and the waste density, the Agency determines the emission rate in grams per hour as follows:

$$\frac{kg}{ton} \cdot V\,\frac{yd^3}{yr} \cdot \frac{1.42\ ton}{yd^3} \cdot \frac{1{,}000\ g}{1\ kg} \cdot \frac{1\ yr}{365\ day} \cdot \frac{1\ day}{24\ hr}$$

Total Respirable Particulate Emissions

U.S. EPA Region 6 calculated the total annual average emissions of respirable particulates (E_{T10}) by summing E_{w10} for wind erosion, E_{v10} for vehicle travel, and E_{l10} for waste loading and unloading operations. Equations 2-7 and 2-8 were then combined to produce Equation 2-24.

$$Q_{p10} = \alpha \cdot (E_{w10} + E_{v10} + E_{l10}) \cdot \frac{1{,}000\ mg}{1\ g} \cdot \frac{1\ hr}{3{,}600\ s} \qquad \textbf{(2-24)}$$

where:

			Default
Q_{p10}	=	emission rate of waste constituent particulates up to 10 µm (mg/s)	calculated
α	=	mass fraction of waste constituent (unitless)	waste-specific (equal to total concentration in waste in mg/mg)
E_{w10}	=	wind erosion emission rate of particulates up to 10 µm (g/hr)	Equation 2-9
E_{v10}	=	vehicle travel emission rate of particulates up to 10 µm (g/hr)	Equation 2-19
E_{l10}	=	waste loading and unloading emission rate of particulates up to 10 µm (g/hr)	Equation 2-21 and 2-22

2.3.1.2 Calculation of Particulate Constituent Concentration in Air at the POE

U.S. EPA Region 6 used the AADM, modified for a landfill source (U.S. EPA 1985c), to calculate the downwind concentrations of constituents released from a land disposal facility. The model assumes that (1) the emission rate is constant over time, (2) the emissions arise from an upwind virtual point source with emissions occurring at ground level, and (3) that no atmospheric destruction or decay occurs.

U.S. EPA Region 6 calculated the downwind concentration of a constituent at the POE (C_{avg}) using Equation 2-25.

$$C_{avg} = \frac{2.03 \cdot Q_p}{\Sigma_z \cdot U \cdot L_v} \cdot F \qquad \textbf{(2-25)}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	calculated
Q_p	=	emission rate of waste constituent particulates (mg/s)	Equation 2-24
Σ_z	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1986b)
L_v	=	distance from the virtual point to the compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows from the sector of interest (unitless)	0.15

The Pasquill-Gifford vertical dispersion coefficient, Σ_z, can be calculated using Equation 2-26 as described in the "Industrial Source Complex (ISC) Dispersion Model User's Guide" (U.S. EPA 1986a).

$$\Sigma_z = a \cdot L^b \qquad \textbf{(2-26)}$$

where:

			Default
Σ_z	=	vertical dispersion coefficient (m)	calculated
a	=	coefficient for stability class D	32.093 (from Appendix B, Table B-5)
L	=	distance from the center of the uncovered waste area to the compliance point 1,000 feet downwind	(site width)/2 + 0.3048 km (U.S. EPA 1986b)
b	=	coefficient for stability class D	0.81066 (from Appendix B, Table B-5)

U.S. EPA Region 6 uses Equation 2-27 to calculate the distance from the virtual point to the compliance point 1,000 feet (304.8) downwind.

$$L_v = L + L' \qquad \textbf{(2-27)}$$

where:

			Default
L_v	=	distance from the virtual point to the compliance point located 1,000 feet downwind (km)	calculated
L	=	distance from center of the uncovered waste area to	(U.S. EPA 1986b)

		the (site width)/2 + 0.3048 km compliance point 1,000 feet (304.8 m) downwind (km)	
L'	=	virtual distance (the distance necessary to convert from an ideal point source to a volume source) (km)	Equation 2-28

The virtual distance is calculated as the distance required for the transverse standard deviation of the Gaussian plume, σ_y, to grow to half the width of the site. This distance will be different for each stability class and is calculated for Pasquill Stability Category D using Equation 2-28.

$$L' = \left(\frac{x}{p}\right)^{1/q} \qquad \textbf{(2-28)}$$

where:

			Default
L'	=	virtual distance (km)	calculated
x	=	½ the width of the area exposed (m)	$0.5*(A_{exposed})^{0.5}$
p	=	Pasquill Stability Category D coefficient (unitless)	68.26 (from Appendix B, Table B-6)
q	=	Pasquill Stability Category D coefficient (unitless)	0.919 (from Appendix B, Table B-6)

As stated previously, U.S. EPA Region 6 used a value of 4 m/s for wind speed (U). Based on information in the *Climatic Atlas of the United States* (Visher 1954), U.S. EPA Region 6 assumed that the average frequency (F) that the wind blows from various directions for many U.S. cities is 0.15. Using these values and the calculated values for Σ_z and L_v as inputs, the average downwind concentration of emissions can be calculated using Equation 2-25.

The concentration of each hazardous constituent actually inhaled and absorbed into the lungs can be calculated using Equation 2-29, assuming that the fraction of inhaled particulates (that is, particulates less than or equal to 10 μm in size) that would be absorbed into the lungs is 12.5 percent (see U.S. EPA 1985a, page 61). U.S. EPA Region 6 assumed that a moderately active person breathes 20 m^3 of air per day based on information in U.S. EPA's "Exposure Factors Handbook" (U.S. EPA 1997b).

$$C_{inh} = C_{avg} \cdot 12.5\% \cdot 20\,\frac{m^3}{day}$$

$$C_{inh} = C_{avg} \cdot 2.5 \qquad \textbf{(2-29)}$$

where:

			Default
C_{inh}	=	concentration of waste constituents inhaled (mg/m^3)	calculated
C_{avg}	=	downwind conc. of waste constituent at POE (mg/m^3)	Equation 2-25
20	=	Adult inhalation rate (m^3/day)	(US EPA, 1997b)

2.3.1.3 Calculation of Air Deposition Rates and Resulting Soil Concentrations at the POE

U.S. EPA Region 6 used the steps described below to investigate air deposition of the annual total emissions of particulates less than or equal to 30 µm in size (that is, total suspended particulates [E_{T30}]) to soil 1,000 feet from the edge of a disposal unit. First, U.S. EPA Region 6 summed E_{w30} for wind erosion, E_{v30} for vehicle travel, and E_{l30} for waste loading and unloading operations (calculated in Equations 2-16, 2-20, and 2-23, respectively) to arrive at the total emission rate of particulates up to 30 µm in size (Q_{p30}) as shown in Equation 2-30.

$$Q_{p30} = \alpha \cdot (E_{w30} + E_{v30} + E_{l30}) \cdot \frac{1{,}000\ mg}{1\ g} \cdot \frac{1\ hr}{3{,}600\ s} \qquad \textbf{(2-30)}$$

where:

			Default
Q_{p30}	=	emission rate of waste constituent particulates up to 30 µm (mg/s)	calculated
α	=	mass fraction of waste constituent (unitless)	waste-specific (equal to total concentration in waste in mg/mg)
E_{w30}	=	wind erosion emission rate of particulates up to 30 µm (g/hr)	Equation 2-16
E_{v30}	=	vehicle travel emission rate of particulates up to 30 µm (g/hr)	Equation 2-20
E_{l30}	=	waste loading and unloading emission rate of particulates up to 30 µm (g/hr)	Equation 2-23

U.S. EPA Region 6 calculated the average downwind particulate concentration in the air 1,000 feet from the disposal unit boundary using the AADM, as described previously (see Equation 2-25). U.S. EPA Region 6 then calculated the flux of particulates hitting the ground at the downwind point using Equation 2-31.

$$q_d = v_d \cdot C_{avg} \qquad \textbf{(2-31)}$$

where:

			Default
q_d	=	rate of deposition (mg/m^2/s)	calculated
v_d	=	deposition velocity (m/s)	0.03 (U.S. Department of Energy [U.S. DOE] 1984)
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	Equation 2-25

The effective deposition velocity is a function of friction velocity, surface roughness height, particle density, and particle size (U.S. DOE 1984). U.S. EPA Region 6 estimated a value of 0.03 m/s for the effective deposition velocity for all particulates less than or equal to 30 μm in size as described below.

The ranges of values for friction velocity and surface roughness height can be obtained from Figure B-4 in Appendix B of this document. To be conservative, U.S. EPA Region 6 assumed that the terrain on which particulates are deposited is a field of grass with blades up to about 5 cm high (a surface not susceptible to erosion but with a capacity for trapping particulates) and assumed a wind speed of 4 m/s. These assumptions led to a roughness height of 2.0 cm and a friction velocity of 50 cm/s (see Figure B-3 in Appendix B). A particle density of 2.61 g/cm^3 (an average particle density for clay particulates) was assumed for the petitioned waste (U.S. EPA 1994a). Figure B-6 in Appendix B presents curves of deposition velocity as a function of particle size for several particle densities and roughness heights. U.S. EPA Region 6 used these curves to obtain deposition velocities for particle sizes of 1, 10, and 30 μm (0.001, 0.018, and 0.07 m/s, respectively).

To obtain an effective deposition velocity for particulates less than or equal to 30 μm in size, U.S. EPA Region 6 calculated the mean velocities for the two particle size ranges, 1 to 10 μm (0.0095 m/s) and 10 to 30 μm (0.044 m/s), and then used the average of these two mean velocities to derive an effective deposition velocity of 0.03 m/s for use in Equation 2-31.

Finally, U.S. EPA Region 6 calculated the resulting soil concentration (C_{soil}) after 1 year of accumulation. U.S. EPA Region 6 conservatively assumed no constituent removal for 1 yr (that is, no leaching, volatilization, soil erosion, or degradation). To calculate C_{soil}, U.S. EPA Region 6 used Equation 2-32.

$$C_{soil} = \left(\frac{q_d}{\rho_b \cdot t} \right) \cdot 3.154 \ x \ 10^7 \ \frac{s}{yr} \ x \ 1 \ yr \qquad \textbf{(2-32)}$$

where:

			Default
C_{soil}	=	concentration of constituent in soil at the POE (mg/kg)	calculated
q_d	=	rate of deposition in $mg/m^2/s$	Equation 2-31
ρ_b	=	soil bulk density (kg/m^3)	1450 (Brady 1984)
t	=	soil thickness from which particles can be ingested (m)	0.01 (U.S. EPA 1994a)

U.S. EPA Region 6 selected a value of 1450 kg/m^3 for soil bulk density; this value is the midpoint of the range for soil bulk density (1200 to 1700 kg/m^3) cited in The Nature and Properties of Soils (Brady 1984). U.S. EPA Region 6 also selected a value of 1 cm as the thickness of the soil surface that will be available for mixing and ingestion. U.S. EPA Region 6 believes that a value of 1 cm is reasonable, given the assumption that no constituent removal would occur for 1 yr (U.S. EPA 1994a).

2.3.2 Calculation of Chemical Concentration in Air at the POE-Volatiles

Petitioned wastes potentially contain volatile organic compounds (VOC). Therefore, U.S. EPA Region 6 evaluates the potential threat to human health resulting from atmospheric transport and inhalation of volatile constituents from a petitioned waste. U.S. EPA Region 6 evaluates the potential influence of volatiles in petitioned waste on air quality for two source terms: landfills and surface impoundments. U.S. EPA Region 6 derives an annual waste generation rate and estimated emissions from landfills using Farmer's equation (Farmer and others 1978). Estimates of emissions of VOCs from disposal of wastewaters in surface impoundments are computed with U.S. EPA's Surface Impoundment Modeling System (U.S. EPA 1990a, 1990b).

The emission rates derived for the two disposal scenarios are entered in U.S. EPA's AADM, a steady-state, Gaussian plume dispersion model, to predict the concentrations of constituents 1,000 feet downwind of a hypothetical disposal facility. For a complete description and discussion of the AADM, refer to (U.S. EPA 1985c).

2.3.2.1 Calculation of Volatile Emissions from a Landfill Using Modified Farmer's Equation

Shen's modification of Farmer's equation, which was developed by U.S. EPA's Office of Air Quality Planning and Standards (OAQPS), is used to estimate the rate of emission of volatiles from a covered landfill (U.S. EPA 1984). This equation provides the rate of volatile emission instead of the flux rate by multiplying by the landfill surface area. U.S. EPA Region 6 determined that Farmer's equation would provide a reliable estimate of volatile emissions from a landfill. The rate of emission from the landfill is calculated using Equation 2-33.

$$E_i = D_a \cdot \frac{1\ cm^2}{1x10^4\ m^2} \cdot A_{exposed} \cdot C_{si} \cdot \frac{1}{L} \cdot \frac{P_a^{10/3}}{P_T^2} \cdot \frac{W_i}{W} \cdot \frac{1\ kg}{1000g} \cdot \frac{1\ g}{1000mg} \qquad \textbf{(2-33)}$$

where:

		Default
E_i	= emission rate of chemical *i* (g/sec)	calculated
D_a	= diffusion coefficient of constituent in air (cm^2/sec)	chem-specific (Appendix A-1)
$1/1x10^4$	= conversion factor for diffusivity from cm^2 to m^2	
$A_{exposed}$	= surface area (m^2)	Equation 2-15
C_{si}	= saturation vapor concentration of *i* in landfill (g/m^3)	Equation 2-35
L	= depth of soil cover (m)	0.1524 (U.S. EPA 1994a)
P_a	= air-filled sand porosity (dimensionless)	0.40 (U.S. EPA 1994a)
P_T	= total sand porosity (dimensionless)	0.40 (U.S. EPA 1994a)
W_i/W	= weight fraction of waste constituent *i* (mg/Kg)	waste-specific
1kg/1000g	= weight fraction conversion factor for (g/g) to (mg/Kg)	
1g/1000mg	=weight fraction conversion factor for (g/g) to (mg/Kg)	

It may be assuming that (1) the total porosity of dry sand is about 40 percent (P_T = 0.40) (U.S. EPA 1994a), (2) the percentage of air-filled pore space in the sand above the landfill is 40 percent (P_a = 0.40) (U.S. EPA 1994a), and the concentration of the constituent at the surface is negligible. Therefore, Equation 2-33 can be simplified to:

$$E_i = D_a \cdot A_{exposed} \cdot \frac{C_{si}}{L} \cdot 2.95x10^{-11} \cdot \frac{W_i}{W} \qquad \textbf{(2-34)}$$

A conservative assumption is made that the waste constituent *i* is in a state of pure component at saturation vapor concentration. The saturation vapor concentration of a waste constituent (C_{si}) in the landfill can be calculated as shown in Equation 2-35.

$$C_{si} = \frac{p_i \cdot MW_i}{R \cdot T} \qquad \textbf{(2-35)}$$

where:

			Default
C_{si}	=	saturation vapor concentration of i in landfill (g/m^3)	calculated
p_i	=	vapor pressure of waste constituent i (mm Hg)	chem-specific (Appendix A-1)
MW_i	=	mole weight of waste constituent i (g/mole)	chem-specific (Appendix A-1)
R	=	molar gas constant (mm Hg-L/mole-^{0}K)	62.36
T	=	standard temperature (K)	298 K (U.S. EPA 1994a)

2.3.2.2 Calculation of Downwind Volatile Waste Constituent Concentration in Air at the POE—Landfill

A landfill system may include a cover above the waste to limit volatile emissions. Therefore, U.S. EPA Region 6 conservatively assumed that a minimum of a 6-inch daily cover was present. U.S. EPA Region 6 used the area of the landfill to calculate atmospheric emissions and transport because the entire surface area will form a source of volatile waste constituents.

U.S. EPA Region 6 used the AADM, modified to estimate emissions from a landfill (U.S. EPA 1994a), to calculate the downwind concentrations of volatile constituents released from the landfill. The model assumes that (1) the emission rate is constant over time, (2) the emissions arise from an upwind virtual point source with emissions occurring at ground level and (3) no atmospheric destruction or decay of the constituent occurs. U.S. EPA Region 6 calculated the average downwind concentration of a constituent as shown in Equation 2-25. The volatile emission rate, Q_v, is calculated using Equation 2-36.

$$Q_v = E_i \cdot \frac{1{,}000\ mg}{g} \qquad \textbf{(2-36)}$$

where:

			Default
Q_v	=	volatile emission rate (mg/sec)	calculated
E_i	=	chemical flux (g/sec)	Equation 2-34

The emission rate is then used to compute the downwind concentration (C_{avg}) by employing the AADM (Equation 2-25). To calculate the POE and the mass inhaled (C_{inh}), U.S. EPA Region 6 assumes that a moderately active person breathes 20 m^3 of air per day (U.S. EPA 1997b). The mass inhaled (C_{inh}) is calculated as shown in Equation 2-37.

$$C_{inh} = C_{avg} \cdot 20 \left(\frac{m^3}{day}\right) \qquad \textbf{(2-37)}$$

where:

			Default
C_{inh}	=	mass of waste constituent inhaled (mg/day)	calculated
C_{avg}	=	downwind conc. of waste constituent at POE (mg/m^3)	Equation 2-25
20	=	Adult inhalation rate (m^3/day)	(US EPA, 1997b)

C_{inh} is computed for each contaminant in the petitioned waste and is then compared to the inhalation levels of concern.

2.3.2.3 Calculation of Volatile Emissions from a Hypothetical Surface Impoundment

U.S. EPA Region 6 uses the Surface Impoundment Modeling System (SIMS) (U.S. EPA 1990a, 1990b) to calculate emissions from a hypothetical disposal impoundment. SIMS was developed by U.S. EPA's OAQPS and is available for downloading along with its documentation from the OAQPS web site at "www.epa.gov.oar.oaqps". The equations contained in SIMS to estimate emissions are presented in this section.

Surface impoundments can be divided into two general categories: treatment and disposal impoundments. In treatment impoundments, wastewater containing particulates, biochemical oxygen demand (BOD), or photodegradable constituents is introduced into a lagoon. The particulates settle to the bottom of the unit, and a combination of biological, photochemical, and volatilization mechanisms causes destruction or removal of dissolved contaminants. An impoundment may be artificially aerated to speed up these processes. The partially treated and clarified wastewater is then drawn off for further treatment and is discharged via a permitted outfall (such as a [NPDES] outfall). The settled sludge may be dredged and landfilled separately. In disposal impoundments, the wastewater does not flow through the unit; instead, the impoundment is sized such that all the water evaporates or infiltrates to groundwater.

Disposal impoundments are defined as units that receive wastewater for ultimate disposal rather than for storage or treatment. Generally, wastewater is not continuously fed to or discharged from these types of impoundments. Therefore, the assumption of an equilibrium bulk concentration, which is applicable for flow-through impoundments, is not applicable for disposal impoundments; the concentrations of VOCs in a disposal impoundment decrease with time. The emission estimating procedure accounts for the decreasing liquid-phase concentrations the driving force for air emissions. For a disposal impoundment that contains no biomass, the biomass concentration equals zero, and no biodegradation of pollutants occurs in the impoundment.

Further information can be found in the "Background Document for the Surface Impoundment Modeling System (SIMS) Version 2.0" (U.S. EPA 1990b). U.S. EPA Region 6 runs the SIMS model for each chemical until the fraction emitted to air reaches 1.0 percent, indicating that a significant fraction of the waste constituent has left the unit. For the lighter nonpolar compounds, this generally occurs after approximately 11 days. U.S. EPA Region 6 computes the average rate of emission from the impoundment by dividing the total mass of each contaminant by the number of days required for all of the contaminant to volatilize. The total rate of volatile emissions from the surface impoundment Q_v is computed using Equation 2-38.

$$Q_v = \frac{(V_{si} \cdot C_o \cdot 1000 mg/g)}{t_f \cdot 86{,}400 \sec/day} \cdot \exp\left(\frac{-K \cdot A_{si} \cdot 4{,}046.8 m^2/acre \cdot t_r \cdot 86{,}400 \sec/day}{V_{si}}\right) \quad \textbf{(2-38)}$$

where:

			Default
Q_v	=	total emission rate of volatiles (mg/sec)	calculated
C_o	=	initial surface impoundment concentration (gm/m^3)	waste-specific
t_f	=	time for constituent concentration to reach 1 percent of C_o (day)	Equation 2-39
K	=	overall mass transfer coefficient (m/s)	Equation 2-41
A_{si}	=	area of the surface impoundment (acres)	Equation 2-12 or 2-13
t_r	=	retention time for liquid in a surface impoundment (days)	Equation 2-40
V_{si}	=	volume of liquid in the surface impoundment (m^3)	delisting petition-specific
4046.8	=	conversion of acres to square meters	
86,400	=	conversion of days to seconds	
1000	=	conversion of grams to milligrams	

The time for the constituent concentration to reach 1 percent of C_o is computed with Equation 2-39.

$$tf = \frac{Ln(0.01) \cdot V_{si}}{-(K \cdot A_{si})} \cdot \frac{1 \ day}{86{,}400} \tag{2-39}$$

where:

			Default
tf	=	time for constituent concentration to reach 1 percent of C_o	calculated
V_{si}	=	volume of liquid in the surface impoundment (m^3)	waste-specific
K	=	overall mass transfer coefficient (m/s)	Equation 2-41
A_{si}	=	area of the surface impoundment (m^2)	waste-specific

The retention time for liquid in a surface impoundment (t_r) is computed with Equation 2-40.

$$t_r = \frac{V_{si}}{Q_f} \cdot \frac{365 \ day}{1 \ year} \tag{2-40}$$

where:

			Default
t_r	=	retention time for liquid in surface impoundment (days)	calculated
V_{si}	=	volume of liquid in the surface impoundment (m^3)	waste-specific (user-provided)
Q_f	=	flow rate of liquid into surface impoundment (m^3/yr)	waste-specific (user-provided)

The overall mass transfer coefficient is calculated using Equation 2-41.

$$K = \frac{1}{\left(\frac{1}{K_l} + \frac{1}{K_g \cdot K_{eq}}\right)} \tag{2-41}$$

where:

			Default
K	=	overall mass transfer coefficient (m/s)	calculated
K_l	=	liquid side mass transfer coefficient (m/s)	Equation 2-42
K_{eq}	=	equilibrium constant (unitless)	Equation 2-43
K_g	=	gas mass phase transfer coefficient (m/s)	Equation 2-44

K_l is calculated using Equation 2-42.

$$K_l = 2.611 \ x \ 10^{-7} \cdot U_{10}^{\ 2} \cdot [\frac{D_w}{D_{ether}}]^{2/3} \qquad \textbf{(2-42)}$$

where:

			Default
K_l	=	liquid side mass transfer coefficient (m/s)	calculated
U_{10}	=	wind speed at 10 m (m/s)	5.73 (U.S. EPA 1994a)
D_w	=	diffusion coefficient in water (cm^2/s)	chemical-specific
D_{ether}	=	diffusion coefficient of ether (cm^2/s)	8.56×10^{-6} (U.S. EPA 1994a)

K_{eg}, also called the dimensionless Henry's Law Constant (H'), is calculated using Equation 2-43.

$$K_{eq} = \frac{H}{R \cdot T} \qquad \textbf{(2-43)}$$

where:

			Default
K_{eq}	=	equilibrium constant (unitless)	calculated
H	=	Henry's Law constant (atm-m^3/g-mol)	chemical-specific
R	=	universal gas law constant (atm-m^3/g-mol K)	8.21×10^{-5}
T	=	standard temperature (K)	298

K_g is calculated using Equation 2-44.

$$K_g = 4.82 \ x \ 10^{-3} \cdot U_{10}^{\ 0.78} \cdot S_{cg}^{\ -0.67} \cdot d_e^{\ -0.11} \qquad \textbf{(2-44)}$$

where:

			Default
K_g	=	gas mass phase transfer coefficient (m/s)	calculated
U_{10}	=	wind speed at 10 m (m/s)	5.73 (U.S. EPA 1994a)
d_e	=	effective diameter of surface impoundment (m)	Equation 2-45
S_{cg}	=	Schmidt number on gas side (unitless)	Equation 2-46

The effective diameter of the surface impoundment is a function of the size of the surface impoundment, and therefore, is directly related to the to the volume of liquid waste petitioned for the delisting (V_{si}). The effective diameter is calculated in Equation 2-45.

$$d_e = \left(4 \cdot \frac{A_{si} \cdot 4{,}046.8 m^2/acre}{3.14} \right)^{0.5} \qquad \textbf{(2-45)}$$

where:

			Default
d_e	=	effective diameter of surface impoundment (m)	calculated
A_{si}	=	area of the surface impoundment (acres)	Equation 2-12 or 2-13
4,046.8	=	conversion from acres to square meters	

S_{cg} is calculated using Equation 2-46.

$$S_{cg} = \frac{\mu_a}{\rho_a \cdot D_a} \qquad \textbf{(2-46)}$$

where:

			Default
S_{cg}	=	Schmidt number on gas side (unitless)	calculated
μ_a	=	viscosity of air (gm/cm-s)	0.000181 (U.S. EPA 1990b)
ρ_a	=	density of air (g/cm^3)	1.2 x 10^{-3} (U.S. EPA 1990b)
D_a	=	diffusivity of constituent in air (cm^2/s)	chem-specific (Appendix A-1)

Average emission rates (in gm/hr) are computed using this methodology and are then used as input for the atmospheric dispersion modeling analysis described below.

2.3.2.4 Calculation of Downwind Waste Constituent Concentration in Air at the POE Surface Impoundment

Calculation of the downwind waste constituent concentration at the POE depends on the assumed size of the disposal unit. The assumptions necessary to determine the size of the unit depend on whether the unit is considered to be a covered landfill or a surface impoundment. U.S. EPA Region 6 calculates downwind concentrations separately for each configuration of the waste management unit. U.S. EPA Region 6 uses Equation 2-25 in Section 2.3.1.2 to calculate downwind concentrations for a landfill.

Section 2.3.2.4 presents U.S. EPA Region 6's calculation of downwind waste constituent concentrations at the POE for a hypothetical surface impoundment.

U.S. EPA Region 6 uses the AADM (U.S. EPA 1985c) to calculate the downwind concentrations of constituents released from a surface impoundment. The model assumes that (1) the emission rate is constant over time, (2) the emissions arise from an upwind virtual point source with emissions occurring at ground level, and (3) no atmospheric destruction or decay of waste constituents occurs. U.S. EPA Region 6 calculates the average downwind concentration of a constituent as shown in Equation 2-25. The downwind concentrations of emissions at the POE for a hypothetical surface impoundment (C_{avg}) can be calculated as shown in Equation 2-47.

$$C_{avg} = \frac{2.03 \cdot Q_v}{\sum_z \cdot U \cdot L_v} \cdot F \qquad \textbf{(2-47)}$$

where:

			Default
C_{avg}	=	downwind conc. of waste constituent at POE (mg/m^3)	calculated
Q_v	=	total emission rate of volatiles (mg/s)	Equation 2-38
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1986b)
L_v	=	distance from the virtual point to the compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows from the sector of interest (unitless)	0.15 (U.S. EPA 1986b)

The mass of a constituent inhaled (C_{inh}) downwind from a hypothetical surface impoundment is computed with the equation that is used to compute inhaled concentrations of volatiles emitted from landfills (Equation 2-37).

2.3.3 Calculation of Waste Constituent Concentration in Surface Water

Exposure through the surface water pathway results from erosion of hazardous materials from the surface of a solid waste landfill and transport of these constituents to nearby surface water bodies. U.S. EPA Region 6 uses the universal soil loss equation (USLE) (Wischmeier and Smith 1978) to compute long-term soil and waste erosion from a landfill in which delisted waste has been disposed. The USLE is used to calculate the amount of waste that will be eroded from the landfill. In addition, the size of the landfill is computed using the waste volume estimate provided by the petitioner and the preprocessing calculation for the EPACML (U.S. EPA 1991b). The volume of surface water into which runoff occurs is determined by estimating the expected size of the stream into which the soil is likely to erode (Keup 1985). The amount of soil delivered to surface water is calculated using a sediment delivery ratio (Mills and others 1982). Finally, a portion of the solid phase that is transported to a surface water body is assumed to be dissolved in surface water column. The dissolved fraction of the waste constituent in the water column is determined by a partitioning equation (U.S. EPA 1998b). U.S. EPA Region 6 uses conservative values for all variables likely to influence the potential for soil erosion and subsequent discharge to surface water. By using conservative values, U.S. EPA Region 6 is providing reasonably conservative estimates of the concentrations of waste constituents in surface water.

U.S. EPA Region 6 calculates erosion and discharge of contaminants to surface water using the USLE (Wischmeier and Smith 1978). The USLE is used to calculate the annual amount of soil and waste eroded, as shown in Equation 2-48.

$$A_{eroded} = RF \cdot K_{ef} \cdot LS \cdot CM \cdot P \qquad \textbf{(2-48)}$$

where:

			Default
A_{eroded}	=	soil and waste eroded (tons/acre/yr)	calculated
RF	=	rainfall erosion factor (1/yr)	300 (Wischmeier and Smith 1978)
K_{ef}	=	soil erodibility (tons/acre)	0.3 (Wischmeier and Smith 1978)
LS	=	slope length, or topographic factor (dimensionless)	petition-specific; see Appendix B, Table B-2 (Wischmeier and Smith 1978)
CM	=	cover & management factor (dimensionless)	1 (Wischmeier and Smith 1978)
P	=	support practice factor (dimensionless)	1 (Wischmeier and Smith 1978)

Rainfall erosion factor (RF) values range from 20 to 550 per year. A value of 300 was chosen as a conservative estimate (Wischmeier and Smith 1978). The distribution of rainfall erosion factor values for the United States is given in Figure B-1 Appendix B (Wischmeier and Smith 1978). Values greater than 300 occur in only a small portion of the southeastern United States. By selecting a value of 300 for its analysis, U.S. EPA Region 6 ensures that a reasonable worst-case scenario is provided for most possible landfill locations within the United States.

Soil erodibility (K_{ef}) factors range from 0.1 to 0.4 ton per acre (see Table B-1 in Appendix B). A value of 0.3 was selected for the analysis; it corresponds to the clay loam, clay, and silty clay loam soil types (Wischmeier and Smith 1978). U.S. EPA Region 6 believes that these soil types represent a reasonable worst-case approximation of the types of soil material present at a landfill.

Topographic factor (LS) values range from 0.06 to 12.9 (see Table B-2 in Appendix B) and account for the influence of slope length and steepness on erosion potential (Wischmeier and Smith 1978). The value is a function of the slope and slope length (see the equation at the bottom of Table B-2 in Appendix B). This parameter is partly petition-specific in that the slope length is calculated from the area of the landfill. The slope, however, is not petition-specific and is conservatively assumed to be 5%.

Cover and management factors (CM) range from 0.4 to 1.0 (see Table B-3 in Appendix B). A value of 1.0 reflects dedicated disposal practices at a facility.

Support practice factor (P) values range from 0.25 to 1.0 (see Table B-4 in Appendix B). This factor reflects the influence of conservation practices on erosion potential. If conservation practices are used (for example, contouring or terracing), the potential for erosion is lower. A support practice factor value of 1.0 means that no support practice is used. This value was therefore chosen as the most conservative value for erosion potential.

2.3.3.1 Computing the Amount of Soil Delivered to Surface Water

U.S. EPA Region 6 computes the percentage of eroded material that is delivered to surface water (the sediment delivery ratio, S_d) based on the assumption that some eroded material will be redeposited between the landfill and the surface water body. U.S. EPA Region 6 assumes a distance (D) of 100 m to the nearest surface water body[d] and uses the sediment delivery ratio equation developed by Mills and others (1982) as shown in Equation 2-49.

---

[d] Refer to the draft "National Survey of Solid Waste (Municipal) Landfill Facilities" (U.S. EPA 1988b). This report shows that 3.6 percent of the surveyed landfill facilities are located within 1 mile (1,609 m) of a river or stream and that the average distance from these facilities to the closest river or stream is 1,936 feet (586 m). Therefore, the assumption of D = 100 m is conservative.

$$
\begin{aligned}
S_d &= 0.77 \cdot (D)^{-0.22} \\
&= 0.77 \cdot 100^{-0.22} \\
&= 0.28
\end{aligned}
\qquad \textbf{(2-49)}
$$

where:

			Default
S_d	=	sediment delivery ratio (unitless)	calculated
0.77	=	constant (unitless)	assumed (U.S. EPA 1994a)
D	=	100, distance to stream or river (m)	assumed (U.S. EPA 1994a)

U.S. EPA Region 6 multiplies the total annual mass of eroded material by the sediment delivery ratio to the determine the mass of soil and waste delivered to surface water (A_s). Using a sediment delivery ratio of 0.28, A_s can be calculated as shown in Equation 2-50.

$$
A_s = A_{eroded} \cdot 907.185 \frac{kg}{ton} \cdot S_d \qquad \textbf{(2-50)}
$$

where:

			<u>Default</u>
A_s	=	soil and waste mass delivered to surface water (kg/acre/yr)	calculated
A_{eroded}	=	amount of soil and waste eroded (tons/acre/yr)	Equation 2-48
S_d	=	sediment delivery ratio (unitless)	0.28 (Equation 2-49)

The total annual amount of soil and waste eroded from the landfill is then calculated using Equation 2-51.

$$
A_w = A_s \cdot F_{exposed} \qquad \textbf{(2-51)}
$$

where:

			<u>Default</u>
A_w	=	rate of soil and waste erosion from landfill (kg/acre/yr)	calculated
A_s	=	soil and waste mass delivered to surface water calculated	Equation 2-50
$F_{exposed}$	=	fraction of area exposed to erosion (unitless)	0.0041 (U.S. EPA 1994a)

The fraction of waste exposed to erosion, $F_{exposed}$, is based on the assumption that 1 month's worth of waste is uncovered at any one time that is, 30 days/(365 days/yr x 20 yr) = 0.0041. U.S. EPA Region 6's assumption that 1 month's worth of waste would be left uncovered at any one time and thus would be readily transportable by surface water runoff is conservative. The minimum criteria for municipal solid waste landfills (MSWLF) set forth in 40 CFR Part 258 require that disposed waste be covered with 6 inches of earthen material at the end of each operating day or at more frequent intervals (40 CFR 258.21). Although a facility might request a temporary waiver of this cover requirement because of extreme seasonal weather conditions, it is highly unlikely that 1 month's worth of waste would be exposed all the time during the active life of the facility (that is, 20 years).

2.3.3.2 Determining the Volume of Surface Water

U.S. EPA Region 6 selected a representative volume or flux rate of surface water based on stream order, which is a system of taxonomy for streams and rivers. A stream that has no other streams flowing into it is referred to as a first-order stream. Where two first-order streams converge, a second-order stream is created. Where two second-order streams converge, a third-order stream is created. Stream order has proven to be a good predictor of flow parameters, including average length, drainage area, mean flow, width, depth, and velocity (Keup 1985). Data indicate that second-order streams have a flow rate of about 3.7 cubic feet per second (3.3×10^9 L/yr). The second-order stream was selected for analysis as the smallest stream capable of supporting recreational fishing. Fifth-order streams were also chosen for analysis as the smallest streams capable of serving as community water supplies. Fifth-order stream flow is on the order of 380 cubic feet per second (3.4×10^{11} L/yr) (U.S. EPA 1994a).

2.3.3.3 Computing the Waste Constituent Concentration in Surface Water

The waste constituent concentration in a surface water body near a land disposal facility is calculated using Equation 2-52.

$$C_{sw} = A \cdot \frac{A_w}{Q_{stream}} \cdot C_{total\ waste} \qquad \textbf{(2-52)}$$

where:

			Default
C_{sw}	=	concentration of waste constituent in surface water (mg/L)	calculated
A	=	area of the waste management unit (acres)	Equation 2-12 or 2-13
A_w	=	rate of waste erosion from landfill (kg/acre/yr)	Equation 2-51
Q_{stream}	=	volume of stream (L/yr)	(U.S. EPA 1994a)
		- for 5th order stream	3.4×10^{11}
		- for 2nd order stream	3.3×10^{9}
$C_{total\ waste}$	=	waste constituent concentration in delisted waste (mg/kg)	waste-specific

U.S. EPA Region 6's assumption that surface water runoff from a land disposal facility will be uncontrolled and will enter an adjacent surface water body also is conservative because 40 CFR 258.26 requires adequate runon and runoff controls at MSWLFs. In addition, 40 CFR 257.33 and 258.27 prohibit a point source or nonpoint source discharge of pollutants that violates any requirements of the Clean Water Act, including NPDES requirements and any requirement of an approved area-wide or state-wide water quality management plan.

2.3.4. Calculation of Dissolved Phase Waste Concentration in Surface Water (C_{dw})

U.S. EPA OSW (U.S. EPA 1998b) recommends the use of Equation 2-53 to calculate the fraction of the surface water concentration (C_{sw}) of a waste constituent that is dissolved in the water column (C_{dw}).

$$C_{dw} = \frac{C_{sw}}{1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}} \qquad \textbf{(2-53)}$$

where:

			Default
C_{dw}	=	Dissolved phase water concentration (mg/L)	Calculated
C_{sw}	=	Waste concentration in water column (mg/L)	Equation 2-52
Kd_{sw}	=	Suspended sediments/surface water partition coefficient (L water/kg suspended sediment)	chem-specific (Appendix A-1)
TSS	=	Total suspended solids concentration (mg/L)	Equation 2-54
1×10^{-6}		Units conversion factor (kg/mg)	

The use of Equation 2-51 to calculate the concentration of the waste constituent dissolved in the water column is consistent with U.S. EPA (1994e) and U.S. EPA (1998b). The total suspended solids concentration (*TSS*) is derived as a function of the soild and waste mass delivered to the surface water from the landfill and the background suspended solids concentration. The *TSS* is calculated as follows:

$$TSS = \frac{A_w \cdot A}{Q_{stream}} + SS_b \qquad \textbf{(2-54)}$$

where:

			Default
TSS	=	total suspended solids concentration (mg/L)	calculated
A_w	=	rate of soil and waste from the landfill (kg/acre/yr)	Equation 2-51
Q_{stream}	=	volume of stream (L/yr) - for 2^{nd} order stream	3.3×10^9 (U.S. EPA 1994)
A	=	area of the waste management unit (acres)	Equation 2-12 or 2-13
SS_b	=	background suspended solids concentration (mg/L)	10 ppt (U.S. EPA 1994a)

The total waste constituent concentration in the water column (C_{sw}) in a second order stream is calculated by using the Equation 2-52. The surface water partition coefficient (Kd_{sw}) is discussed below.

2.3.4.1 Partitioning Coefficients for Suspended Sediment-Surface Water (Kd_{sw})

Partition coefficients (*Kd*) describe the partitioning of a compound between sorbing material, such as soil, soil pore-water, surface water, suspended solids, and bed sediments. For organic compounds, *Kd* has been estimated to be a function of the organic-carbon partition coefficient and the fraction of organic carbon in the partitioning media. For metals, *Kd* is assumed to be independent of the organic carbon in the partitioning media and, therefore, partitioning is similar in all sorbing media.

The soil-water partition coefficient (Kd_s) describes the partitioning of a compound between soil pore-water and soil particles, and strongly influences the release and movement of a compound into the subsurface soils and underlying aquifer. The suspended sediment-surface water partition coefficient (Kd_{sw}) coefficient describes the partitioning of a compound between surface water and suspended solids or sediments.

Organics For organics (including PCDDs and PCDFs), soil organic carbon is assumed to be the dominant sorbing component in soils and sediments. Therefore, *Kd* values were calculated using the following fraction organic carbon (f_{OC}) correlation equations from *Review Draft Addendum to the*

Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions (U.S. EPA 1993d):

$$Kd_{sw} = f_{oc,\,sw} \cdot K_{oc} \tag{2-55}$$

U.S. EPA (1993d), from literature searches, states that f_{OC} could range as follows:

- 0.05 to 0.1 in suspended sediments - for which a mid-range value of $f_{oc,sw} = 0.075$ generally can be used.

Consistent with the the Region 6 Combustion Risk Assessment Protocol (U.S. EPA 1998b) guidance document, this DTSD uses mid-range f_{oc} values recommended by U.S. EPA (1993d). Kd_{sw} values were calculated using K_{oc} values recommended for each compound in this DTSD.

The variables (Kd_{sw}, TSS) in the Equation 2-53 are site-specific. Therefore, the use of the default values will introduce an under- or overestimation of C_{dw}. The degree of uncertainty associated with *TSS* is dependent on the suspended solids of the nearest second order stream and how this value compares to the default background suspended solids value (20 mg/L). Uncertainty associated with the variable Kd_{sw} is associated with estimates of the fraction organic content (f_{OC}). Because f_{OC} values can vary widely for different locations in the same medium, using default f_{OC} values may result in significant uncertainty in specific cases.

2.3.4.2 Concentration of Waste Constituent in Fish (C_{fish})

The waste constituent concentration in fish is calculated using either a waste-specific bioconcentration factor (*BCF*) or a waste-specific bioaccumulation factor (*BAF*). For compounds with a $log\ K_{ow}$ less than 4.0, *BCF*s are used. Compounds with a $log\ K_{ow}$ greater than 4.0 are assumed to have a high tendency to bioaccumulate, therefore, *BAF*s are used. Appendix A-1 provides a detailed discussion on the sources of the waste constituent-specific *BCF* and *BAF* values, and the methodology used to derive them. *BCF* and *BAF* values are generally based on dissolved water concentrations. Therefore, when *BCF* or *BAF* values are used, the waste constituent concentration in fish is calculated using dissolved water concentrations. The equations used to calculate fish concentrations are described in the subsequent subsections.

2.3.4.3 Calculation of Fish Concentration (C_{fish}) from Bioconcentration Factors Using Dissolved Phase Water Concentration

U.S. EPA OSW recommends the use of Equation 2-56 to calculate fish concentration from *BCF*s using dissolved phase water concentration. The use of this equation is further described in Appendix A-1.

$$C_{fish} = C_{dw} \cdot BCF_{fish} \tag{2-56}$$

where:

			Default
C_{fish}	=	Concentration of waste constituent in fish (mg/kg FW tissue)	calculated
C_{dw}	=	Dissolved phase water concentration of waste constituent (mg/L)	Equation 2-53
BCF_{fish}	=	Fish bioconcentration factor of waste constituent (L/kg)	chem-specific (Appendix A-1)

The dissolved phase water concentration (C_{dw}) is calculated by using Equation 2-53 above. Chemical-specific BCF_{fish} values are presented in Appendix A-1. The use of Equation 2-56 to calculate fish concentration is consistent with U.S. EPA (1994e) and U.S. EPA (1998b) .

2.3.4.4 Calculation of Fish Concentration (C_{fish}) from Bioaccumulation Factors Using Dissolved Phase Water Concentration

U.S. EPA OSW recommends the use of Equation 2-57 to calculate fish concentration from *BAF*s using dissolved phase water concentration.

$$C_{fish} = C_{dw} \cdot BAF_{fish} \tag{2-57}$$

where:

			Default
C_{fish}	=	Concentration of waste constituent in fish (mg/kg FW tissue)	calculated
C_{dw}	=	Dissolved phase water conc. of waste constituent (mg/L)	Equation 2-53
BAF_{fish}	=	Fish bioconcentration factor of waste constituent (L/kg)	chem-specific (Appendix A-1)

The dissolved phase water concentration (C_{dw}) is calculated by using Equation 2-53. Chemical-specific bioaccumulation factor *(BAF_{fish})* values are presented in Appendix A-1.