RCRA DELISTING
TECHNICAL SUPPORT DOCUMENT

Office of Solid Waste

Prepared by

Multimedia Planning and Permitting Division
U.S. Environmental Protection Agency
Region 6
Dallas, TX 75202

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Updated by

Land and Chemicals Division
U.S. Environmental Protection Agency
Region 5
Chicago, IL 60404

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DISCLAIMER

This document provides describes to U.S. Environmental Protection Agency (U.S. EPA) regions and states on how to use the U.S. EPA Delisting Risk Assessment Software (DRAS) as a tool for the evaluation of hazardous waste delisting petitions. The document is not a substitute for U.S. EPA regulations, nor is it a regulation itself. Thus, it cannot impose legally binding requirements on U.S. EPA, states, or the regulated community. It may not apply to a particular situation based on the circumstances. U.S. EPA may change the DRAS in the future, as appropriate.

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ACRONYMS AND ABBREVIATIONS

ì g	Microgram
ì m	Micron
AADM	Ambient Air Dispersion Model
ADD	Average daily dose
AP-42	“Title”
API	American Petroleum Institute
AT	Averaging time
atm	Atmosphere
BAF	Bioaccumulation factor
BaP	Benzo(a)pyrene
BCF	Bioconcentration factor
BOD	Biochemical oxygen demand
BSAF	Biota-sediment bioaccumulation factor
BW	Body weight
CAS	Chemical Abstracts Service
CFR	*Code of Federal Regulations*
CLP	Contract Laboratory Program
cm	Centimeter
CSF	Cancer slope factor
DAD	Personally absorbed dose
DAF	Dilution attenuation factor
DF	Dilution factor
DRAS	Delisting Risk Assessment Software
DTSD	Delisting Technical Support Document
DW	Dry weight of soil or plant or animal tissue
EFH	Exposure Factors Handbook
EPACML	U. S. EPA Composite Model for Landfill
EPACMTP	U. S. EPA Composite Model for Leachate Migration with Transformation Products
EQL	Estimated quantitation limit
FR	*Federal Register*
g	Gram
GC	Gas chromatography
HBN	Health-based number
HEAST	Health Effects Assessment Summary Tables
HELP	Hydrologic Evaluation of Landfill Performance
HI	Hazard index
HQ	Hazard quotient
hr	Hour
HSWA	Hazardous and Solid Waste Amendments

ACRONYMS AND ABBREVIATIONS (Continued)

HWIR	Hazardous Waste Identification Rule
IDL	Instrument detection limit
IEUBK	Integrated Exposure Uptake Biokenitc Model
IRIS	Integrated Risk Information System
ISC	Industrial Source Complex
ISCST3	Industrial Source Complex Short Term 3
ISCSTDFT	Industrial Source Complex Short Term Draft
K	Kelvin
kg	Kilogram
km	Kilometer
L	Liter
LADD	Lifetime average daily dose
lb	Pound
m	Meter
MCL	Maximum contaminant level
MCLG	Maximum contaminant level goal
MDL	Method detection limit
Mg	Megagram
mg	Milligram
MIR	Maximum individual risk
mL	Milliliter
mm	Millimeter
MSWLF	Municipal solid waste landfill
NAPL	Nonaqueous-phase liquid
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NCEA	U. S. EPA National Center for Environmental Assessment
NPDES	National Pollutant Discharge Elimination System
NRC	National Research Council
NWT	Nonwastewater leachate
OAQPS	U. S. EPA Office of Air Quality Planning and Standards
OPPI	U. S. EPA Office of Policy Planning and Implementation
ORD	U. S. EPA Office of Research and Development
OSW	U. S. EPA Office of Solid Waste
PAH	Polynuclear aromatic hydrocarbon
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo(p)dioxin
PCDF	Polychlorinated dibenzofuran
PDF	Probability density function
pg	Picogram
PM	Particulate matter

ACRONYMS AND ABBREVIATIONS (Continued)

PM10	Particulate matter less than 10 microns in diameter
POE	Point of exposure
ppb	Parts per billion
ppm	Parts per million
ppmv	Parts per million by volume
ppt	Parts per trillion
PQL	Practical quantitation limit
QA	Quality assurance
QAPP	Quality assurance project plan
QC	Quality control
RAEPE	“Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites”
RAGS	“Risk Assessment Guidance for Superfund”
RCRA	Resource Conservation and Recovery Act
RfC	Reference concentration
RfD	Reference dose
RMC	Reynolds Metals Company
RME	Reasonable maximum exposure
RPF	Relative potency factor
s	Second
SIMS	Surface Impoundment Modeling System
SQL	Sample quantitation limit
STORET	Database Utility for STOrage and RETrieval of Chemical, Physical, and Biological Data for Water Quality
SVOC	Semivolatile organic compound
SW-846	“Test Methods for Evaluating Solid Waste”
TCDD	Tetrachlorodibenzo(p)dioxin
TCLP	Toxicity characteristic leaching procedure
TC Rule	Toxicity characteristics Rule
TDA	Toluenediamine
TDI	Toluene diisocyanate
TEF	Toxicity equivalent factor
TEQ	Toxicity equivalent quotient
Tetro Tech	Tetro Tech EM Inc.
THQ	Target hazard quotient
TIC	Tentatively identified compound
TLV	Threshold limit value
TOC	Total organic carbon
TSD	Treatment, storage, or disposal
TWA	Time-weighted average
UCL_{95}	95th percentile upper confidence limit
U. S. DOE	U. S. Department of Energy

ACRONYMS AND ABBREVIATIONS (Continued)

U.S. EPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
USLE	Universal soil loss equation
VOC	Volatile organic compound
WQC	Water quality criteria
yd	Yard
yr	Year

VARIABLES

A = area of waste management unit (acres)
a = coefficient for Stability Class D = 32.093
A_{eroded} = amount of soil and waste eroded (tons/acre/yr)
$A_{exposed}$ = area of waste management unit exposed (acres)
A_s = waste mass delivered to surface water (kg/acre/yr)
A_{si} = area of surface impoundment (m^2)
A_{skin} = exposed skin surface area (cm^2)
A_w = rate of waste erosion from landfill (kg/acre/yr)
ADD = average daily dose (mg waste constituent/kg BW/day)
$ADD_{c,p}$ = average daily dose for waste constituent c for pathway p (mg/kg-day)
AT = averaging time (days or yrs)

$\hat{a}$ = Proportionality constant (cm/sec)$^{-1/3}$
B = Bunge constant (unitless)
b = coefficient for Stability Class D = 0.81066
BAF = bioaccumulation factor (L/kg)
BD = soil dry bulk density (g soil/cm^3 soil)
BW = body weight (kg)

C = constant (m^2/s)$^{-2/3}$
C_{air} = constituent's maximum allowable respirable air concentration at POE (mg/m^3)
$C_{air\text{-}I}$ = constituent air concentration from compartments: shower, bathroom, and house (mg/L)
$C_{air\text{-}max}$ = maximum possible air concentration of waste constituent based on Henry's Law (mol/L)
C_{avg} = downwind concentration of waste constituent at POE (mg/m^3)
$C_{avg,s}$ = average constituent air concentration in shower (mg/L)
$C_{avg,b}$ = average constituent air concentration in bathroom (mg/L)
$C_{avg,h}$ = average constituent air concentration in house (mg/L)
$C_{dl\text{-}air\text{-}p}$ = pathway total concentration delisting level for respirable landfill air particulates (mg/kg)
$C_{di\text{-}air\text{-}si}$ = pathway leachate concentration delisting level for volatiles from surface impoundment (mg/L)
$C_{dl\text{-}air\text{-}v}$ = pathway total concentration delisting level for volatiles from landfill (mg/L)
$C_{dl\text{-}dermal}$ = pathway leachate concentration delisting level for groundwater dermal contact (mg/L)
$C_{dl\text{-}fish}$ = pathway total concentration delisting level for fish ingestion (mg/kg)
$C_{dl\text{-}ingest}$ = pathway leachate concentration delisting level for groundwater ingestion (mg/L)
$C_{dl\text{-}inhale}$ = pathway leachate concentration delisting level for shower inhalation (mg/L)
$C_{dl\text{-}soil}$ = pathway total concentration delisting level for soil ingestion (mg/kg)
$C_{dl\text{-}water}$ = pathway total concentration delisting level for ingestion of surface water (mg/kg)
C_{dw} = dissolved-phase water concentration (mg waste constituent/L water)
C_{fish} = maximum allowable concentration of waste constituent in fish tissue (mg/kg)
$C_{5th\text{-}stream}$ = concentration of waste in fifth-order stream (kg/L)

VARIABLES (Continued)

C_{gen} = generic constituent concentration — the medium average concentration contacted over the exposure period (for example, mg/kg for soil and mg/L for water)
C_{gw} = waste constituent concentration in groundwater (mg/L)
$C_{gw\text{-}dermal}$ = maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)
$C_{gw\text{-}ingest}$ = maximum allowable constituent concentration in groundwater for ingestion (mg/L)
$\mathrm{C}_{\mathrm{gw\text{-}inhale}}$ = maximum allowable constituent concentration in groundwater used for showering (mg/L)
C_i = vapor-phase concentration of *I* in landfill (g/m^3)
c_i = maximum allowable vapor-phase concentration of constituent in landfill (g/m^3)
C_{inh} = mass of waste constituent inhaled (mg/day)
C_L = leachate concentration (TCLP concentration) (mg/L)
C_l = Concentration of constituent *I* in liquid phase (mol.m^{-3})
C_{LMAX} = maximum allowable waste leachate (TCLP) concentration (mg/L)
C_s = vapor-phase concentration of *constituent I* at surface (g/m^3)
C_{sat} = soil saturation concentration (mg/kg)
C_{soil} = resulting soil concentration (mg/kg soil/yr)
$C_{soluble}$ = concentration of soluble fraction of constituent in waste (mg/kg)
C_{sw} = concentration of waste constituent in surface water (mg/L)
$C_{total\ waste}$ = total concentration of constituent in waste (mg/kg)
$Cancer\ Risk_i$ = individual lifetime risk indirect exposure to waste constituent *I* (unitless)
$Cancer\ Risk_{inh(I)}$ = individual lifetime cancer risk from direct inhalation of carcinogen waste constituent *I* (unitless)
CM = USLE cover management factor (unitless)
CR = water consumption rate (L/day)
CR_{fish} = fish water consumption rate (kg/day)
CR_{gen} = contact rate — the amount of contaminated medium contacted per unit time or per event (for example, kg/day for soil and L/day for water) (upper-bound value)
CR_{soil} = soil consumption rate (mg/day)
Cs = average soil concentration over exposure duration (mg waste constituent/kg soil)
C_{si} = saturation vapor concentration of *I* in landfill
CSF_c = cancer slope factor for waste constituent *c* (mg/kg day)$^{-1}$
CSF_{inhal} = constituent inhalation cancer slope factor (mg/kg day)$^{-1}$
CSF_{oral} = constituent oral cancer slope factor (mg/kg day)$^{-1}$

D = 100, distance to stream or river (m)
d = depth of soil cover (m)
D_a = diffusivity of constituent in air (cm^2/s)
D_{air} = diffusion coefficient of constituent in air (m^2/s)
d_e = effective diameter of surface impoundment (m)
D_{ether} = diffusion coefficient of ether (cm^2/s)
D_i = gas-phase diffusion coefficient (m^2/s)
D_l = diffusivity in water of a chemical (m^2/s)
D_w = diffusion coefficient in water (cm^2/s)
DA_{event} = dose absorbed per unit area per event (mg/cm^2-event)

VARIABLES (Continued)

DAD = dermally absorbed dose (mg/kg-day)
DAF = dilution attenuation factor (unitless)
DAF_{sf} = DAF scaling factor (unitless)
$\mathrm{DAF}_{\mathrm{va}}$ = waste volume-adjusted DAF (unitless)
DH = drop height of material from truck (m)
d_p = droplet diameter (cm)

E_i = landfill volatile emission flux of constituent (g/s)
E_l = particulate emissions from waste loading and unloading operations (kg/ton)
E_{l10} = waste loading and unloading emission rate of particulates up to 10 ì m (kg/ton)
E_{l30} = waste loading and unloading emission rate of particulates up to 10 ì m (kg/ton)
E_T = total emission rate of particulates that may be inhaled (g/hr)
E_{T10} = total emission rate of particulates up to 10 ì m (g/hr)
E_{T30} = total emission rate of particulates up to 30 ì m (g/hr)
E_v = particulate emissions from vehicle travel (g/hr)
E_{v10} = vehicle travel emission rate of particulates up to 10 ì m (g/hr)
$\mathrm{E}_{\mathrm{v30}}$ = vehicle travel emission rate of particulates up to 30 ì m (g/hr)
E_w = particulate emissions from wind erosion (g/hr)
E_{w10} = wind erosion emission rate of particulates up to 10 ì m (g/hr)
E_{w30} = wind erosion emission rate of particulates up to 30 ì m (g/hr)
ED = exposure duration (yr)
EF = exposure frequency (days/yr)
ET_{comp} = exposure time in each compartment (bath, shower, or house) (days/shower)
EV = event frequency (events/day)

F = frequency that wind blows from sector of interest (unitless)
F_c = fraction contaminated (unitless)
$\mathrm{f}_{\mathrm{em,b}}$ = fraction of constituent emitted from bathroom water use (unitless)
$\mathrm{f}_{\mathrm{emh}}$ = fraction of constituent emitted from house water use (unitless)
$\mathrm{f}_{\mathrm{em,s}}$ = fraction of constituent emitted from shower water use (unitless)
$F_{exposed}$ = fraction of area exposed to erosion (unitless)
F_{inhal} = fraction of particulates inhaled (Unitless)
f_{oc} = fraction organic carbon content of soil (g/g)
$f_{sat, I}$ = Fraction of gas phase saturation for each shower inhalation compartment I
$F(X)$ = dimensionless function obtained from plot in RAEPE

h = nozzle height (cm)
H = Henry's Law constant (atm-m^3/mol)
H' = dimensionless Henry's Law constant
HBN = health-based number (or MCL) (mg/L)
HI = hazard index (unitless)
HI_p = total hazard index for all waste constituents for specific exposure pathway p
HQ = hazard quotient (unitless)
$HQ_{\mathrm{c,p}}$ = hazard quotient for waste constituent c for exposure pathway p (unitless)
HI_{cum} = aggregate hazard index for all constituents and all exposure pathways

VARIABLES (Continued)

I = intake — amount of constituent at exchange boundary (mg/kg-day); for evaluating exposure to noncarcinogenic constituents, this intake is referred to as ADD; for evaluating exposure to carcinogenic constituents, this intake is referred to as LADD
I_b = bathroom water use (L/min)
I_h = house water use (L/min)
I_s = shower water use (L/min)
IFA_{adj} = inhalation factor, age-adjusted ([m^3-year]/[kg-day])
IFS_{adj} = soil ingestion factor ([mg-year]/[kg-day])
IFW_{adj} = water ingestion factor, age-adjusted [L •year]/[kg•day]
IR = inhalation rate (m^3/day or m^3/hr)

J_i = volatile emission flux of constituent (g/m^2/s)

k = constant— 0.36 for particulates up to 10 ì m and 0.8 for particulates up to 30 ì m
K = overall mass transfer coefficient (m/s)
K_d = soil-water partition coefficient (cm^3 water/g or L/kg)
K_{ef} = USLE erodibility factor (ton/acre)
K_{eq} = equilibrium constant (unitless)
K_G = gas-phase transfer coefficient (m/yr)
K_g = gas-phase mass transfer coefficient (m/s)
K_L = liquid-phase transfer coefficient (m/yr)
K_l = liquid-phase mass transfer coefficient (m/s)
K_{oc} = soil organic carbon-water partition coefficient (mL water/g soil)
k_{oc} = normalized distribution coefficient (L/kg)
K_{ol} = overall mass transfer coefficient (cm/sec)
K_{ow} = octanol-water partition coefficient (mg waste constituent/L octanol)/(mg waste constituent/L water)
K_p = batch drop particle size multiplier (dimensionless)
K_p^w = skin permeability constant in water (cm/hr)
Ke = equilibrium coefficient (s/cm-yr)

L = distance from center of uncovered waste area to 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)
L_v = distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)
$LADD$ = lifetime average daily dose (mg waste constituent/kg BW/day)
$LADD_{c,p}$ = lifetime average daily dose for waste constituent *c* (mg/kg-day) via pathway *p*
LS = USLE length-slope factor (unitless)

M = moisture content of waste (percent)
M_i = molecular weight (g.mol^{-1})
MW = molecular weight

n = total soil porosity (L_{pore}/L_{soil})

VARIABLES (Continued)

N_p = number of days per year with at least 0.01 inch of precipitation (days per year)

P = support practice factor (dimensionless)
p = Pasquill Stability coefficient for Category D (unitless)
P_a = air-filled sand porosity (dimensionless)
P_p = partial pressure of constituent (atm)
P_T = total sand porosity (dimensionless)
PF = USLE supporting practice factor (unitless)

q = Pasquill Stability coefficient for Category D (unitless)
q_d = rate of deposition ($mg/m^2/s$)
Q_{gs} = volumetric gas exchange rate between shower and bathroom (L/min)
Q_{gb} = volumetric gas exchange rate between bathroom and house (L/min)
Q_{gh} = volumetric gas exchange rate between house and atmosphere (L/min)
Q_p = emission rate of waste constituent particluates (mg/s)
Q_{p10} = emission rate of waste constituent particulates up to 10 ì m (mg/s)
Q_{p30} = emission rate of waste constituent particulates up to 30 ì m (mg/s)
Q_v = volatile emission rate (mg/s)
Q_{2nd} = flux of water in second-order stream (L/year)
Q_{stream} = volumetric flow of stream (L/year)

R = universal gas constant ($atm\text{-}m^3/mol\text{-}K$)
RF = rainfall erosion factor (1/year)
RfC = reference concentration (mg/kg)
RfD = reference dose (mg/kg-day)
RfD_c = reference dose for waste constituent c (mg/kg-day)
$Risk$ = cancer risk for carcinogens (unitless)
$Risk_{c,p}$ = risk for waste constituent c for specific exposure pathway p
$Risk_{cum}$ = aggregate risk for all constituents and all exposure pathways
$Risk_p$ = total risk for all constituents for specific exposure pathway p

S = mean vehicle speed (km/hr)
s = silt content of waste (percent)
S_{cg} = Schmidt number on gas side (unitless)
S_d = sediment delivery ratio (unitless)
SF = slope factor $(mg/kg\text{-}day)^{-1}$
Sol = solubility in water (mg/L water)

T = standard temperature (K)
t = soil thickness from which particles can be ingested (m)
$(t_{t+1} - t_t)$ = calculational time step (min)
t_{event} = duration of event (hr/event)
$TCLP$ = TCLP concentration of waste constituent (mg/L)
tf = time for constituent concentration to reach 1 percent of C_o
THQ = target hazard quotient (unitless)
TPD_{min} = minimum round trips per day
TR = individual target risk level (unitless)

VARIABLES (Continued)

TSS = total suspended solids concentration (mg/L)

U = mean annual wind speed (m/s)
U_{10} = wind speed at 10 m (m/s)
U_t = threshold value of wind speed at 7 m (m/s)

V = volume of landfilled waste (yd^3)
V_b = volume of bathroom (L)
V_h = volume of house (L)
V_s = volume of shower (L)
V_{si} = volume of liquid in surface impoundment (m^3)
v_d = deposition velocity (m/s)
v_t = terminal velocity (cm/sec)
Vf = fraction of disposal site covered with vegetation (unitless)
VKT = vehicle kilometers traveled-km trip x number of trips
VR_{comp} = ventilation rate for compartment (shower, bathroom, house) (L/hr)

W = mean vehicle weight (tons)
w = mean number of wheels per vehicle
W_{comp} = water used in one of three compartments (shower, bathroom, house) (L/hr)

X = dimensionless ratio
x = ½ width of area exposed (m)

$\mathrm{y}_{\mathrm{b,t}}$ = gas phase constituent concentration in the bathroom (mg/L)
$\mathrm{y}_{\mathrm{h,t}}$ = gas phase constituent concentration in the house (mg/L)
$\mathrm{y}_{\mathrm{s,t}}$ = gas phase constituent concentration in the shower (mg/L)
Yd = dumping device capacity (m^3)

$á$ = mass fraction of constituent in waste (unitless)
$Ö$ = mass transfer efficiency of chemical (unitless)
$Ö_{Rn}$ = mass transfer efficiency of radon (unitless)
r_a = density of air (g/cm^3)
$ñ_b$ = soil bulk density (mg/m^3)
$ñ_{db}$ = dry soil bulk density (kg/L)
$ñ_s$ = soil particle density (kg/L)
$\mathrm{ñ}_\mathrm{w}$ = waste density (tons per cubic yard)
$È_a$ = air-filled soil porosity (L$_{air}$/L$_{soil}$)
$È_w$ = water-filled soil porosity (L$_{water}$/L$_{soil}$)
$ô$ = lag time (hr)
m_a = viscosity of air (gm/cm-s)
$\sum_z$ = vertical dispersion coefficient (m)
10^{-2} = unit conversion factor (kg-cm^2/mg-m^2)
10^{-3} = unit conversion factor (kg-ì g/g^2)
10^{-4} = unit conversion factor (m^2/cm^2)
10^{-6} = unit conversion factor (g/ì g)
10^{-6} = unit conversion factor (kg/mg)
0.001 = unit conversion factor (g/mg)

0.004047 = unit conversion factor (km^2/acre)
0.31536 = unit conversion factor (m-g-s/cm-ì g-yr)
365 = unit conversion factor (days/yr)
907.18 = unit conversion factor (kg/ton)
3.1536×10^7 = unit conversion factor (s/yr)

Chapter 1
Introduction

What's covered in Chapter 1:

- Document Objective and Purpose
- Background
- Delisting Reference Documentation
- Document Organization

Under the regulations implementing Subtitle C of the Resource Conservation and Recovery Act (RCRA), wastes are designated as hazardous in two ways: (1) solid wastes that exhibit certain characteristics (those listed in 40 *Code of Federal Regulations* [CFR] Part 261, Subpart C) and (2) solid wastes that are specifically listed as hazardous (those listed in 40 CFR Part 261, Subpart D). As set forth in Subpart C, wastes that are characteristically hazardous remain so until they no longer exhibit any characteristic for which they are listed. Toxicity is one of the characteristics for which Subtitle C wastes are listed as hazardous. This document outlines a risk assessment procedure for determining whether a Subtitle C listed waste exceeds the U.S. Environmental Protection Agency (U.S. EPA) criteria for toxicity, a characteristic of RCRA listed wastes. Risk assessment is a science used to evaluate the carcinogenic risks and noncarcinogenic hazards to human health that are attributable to releases of hazardous chemicals. Risk assessments conducted for the delisting include evaluation of risks associated with direct and indirect exposures to waste constituents. The following definitions are adopted from the National Research Council's (NRC) 1983 report titled "Risk Assessment in the Federal Government: Managing the Process" (NRC 1983) for use throughout this guidance:

Risk assessment	The scientific evaluation of potential health impacts that may result from exposure to a particular substance or mixture of substances under specified conditions
Hazard	An impact to human health by waste constituents of concern
Risk	An estimation of the probability that an adverse health impact may occur as a result of exposure to chemicals in the amount and by the pathways identified

Dose	Constituent mass administered into the body per unit body weight per unit time (for example, in milligrams per kilogram per day)
Exposure	Exposure of identified receptors to chemicals via relevant pathways
Direct exposure	Exposure via immediate inhalation from a contaminated source
Indirect exposure	Exposure resulting from contact of human and ecological receptors with soil or water bodies on which an emitted chemical has been deposited or into which an emitted chemical has leached
Secondary exposure	Synonymic phrase for indirect exposure

This Delisting Technical Support Document (DTSD) was developed to be an integral part of and to provide the technical background for the Delisting Risk Assessment Software (DRAS). The DRAS can aid in determining whether a waste qualifies as being not characteristically toxic for the purposes of delisting under 40 CFR 260.20 and 260.22. The DRAS was developed to compute the risks and hazards associated with a specific waste stream for which a delisting petition has been submitted. The DRAS assesses the toxicity of a petitioned waste by estimating (1) chemical- and waste volume-specific screening exit values and (2) aggregate cancer risks and noncarcinogenic hazard indices (HI). To calculate the potential risks associated with a particular waste stream petitioned for delisting, specific information about the petitioned waste is required. U.S. EPA requires the following waste-specific information for DRAS in order to estimate risks and hazards associated with potential exposure to the petitioned waste stream:

- The maximum annual or total waste volume of the petitioned waste;
- The maximum total concentration of each chemical constituent in the petitioned waste;
- The maximum Toxicity Characteristic Leaching Procedure (TCLP) concentration of each chemical constituent in the petitioned waste; and
- The number of years the petitioned waste is projected to be generated

Section 1.1 discusses the objectives and purpose of this document. Section 1.2 provides background on the Hazardous Waste Delisting Program. Section 1.3 summarizes delisting reference documentation. An overview of the organization of this DTSD is provided in Section 1.4.

1.1 DOCUMENT OBJECTIVES

The objectives of the DTSD are to (1) describe the human health risk-based delisting methodology developed to perform a screening-level analysis and to compute aggregate risks and HIs for petitioned wastes and (2) provide documentation of data and default parameters selected for the risk analysis. The DTSD provides background information about the algorithms and equations used in conjunction with dilution attenuation factors (DAF) to compute cancer risks and hazard quotients (HQ) for individual chemicals. This information is intended to assist regulatory authorities, petitioners, and decision-makers in making hazardous waste delisting determinations.

1.2 BACKGROUND

Section 1004(5) of RCRA as amended by the Hazardous and Solid Waste Amendments (HSWA) of 1984 defines “hazardous waste” as “a solid waste, or combination of solid waste, which because of its quantity, concentration, or physical, chemical, or infectious characteristics may (a) cause, or significantly contribute to an increase in the mortality or an increase in serious irreversible, or incapacitating reversible, illness; or (b) pose a substantial present or potential hazard to human health or the environment when improperly treated, stored, transported, or disposed of, or otherwise managed.”

Section 3001 of RCRA requires U.S. EPA to identify those wastes that should be classified as “hazardous.” The Agency’s hazardous waste identification rules designate wastes as hazardous in one of two ways. First, the Agency has established four hazardous waste characteristics that identify properties or attributes of wastes that would pose a potential hazard if the wastes are improperly managed (see 40 CFR 261.21 through 261.24). Any generator of a solid waste is responsible for determining whether a solid waste exhibits any of these characteristics (see 40 CFR 262.11). Any solid waste that exhibits any of the characteristics remains hazardous until it no longer exhibits the characteristics (see 40 CFR 261.4(d)(1)).

The other mechanism that U.S. EPA uses to designate wastes as hazardous is “listing.” The Agency has reviewed data on specific waste streams generated from a number of industrial processes and has determined that these wastes, if mismanaged, would pose hazards for one or more reasons, including (1) the presence of significant levels of hazardous constituents listed in Appendix VIII to 40 CFR Part 261, (2) manifestation of one or more of the hazardous waste characteristics, or (3) the potential to impose detrimental effects on the environment (see generally 40 CFR 261.11). U.S. EPA has generally determined that these wastes contain toxic constituents at concentrations that potentially pose risks that are unacceptable for human or environmental exposure and that these constituents are mobile and persistent to the degree that they can reach environmental or human receptors.

As part of its RCRA Subtitle C hazardous waste regulations, U.S. EPA's Office of Solid Waste (OSW) gives facilities the flexibility to petition the Agency to exempt low-risk listed hazardous wastes that may not actually pose a threat to human health or the environment under the provisions of 40 CFR 260.20 and 260.22. This process is referred to as the "delisting" of a specific generator's listed waste. U.S. EPA's OSW was directed by statute to review petitions in order to determine whether the wastes may be delisted. The overall intent of the delisting process is to ease the regulatory burden on handlers of listed wastes that may have been improperly classified as hazardous by the broad listing definitions. In addition, the delisting process can be used to exclude listed wastes that are sufficiently treated, that they no longer pose a threat to human health or the environment. Listed hazardous wastes that exhibit any of the characteristics will continue to be regulated as hazardous wastes until the characteristic is removed. In a number of cases, wastes were listed because they contained toxic hazardous constituents and exhibited one or more of the hazardous waste characteristics that do not relate to chemical toxicity (for example, ignitability, corrosivity, and reactivity). If such a waste still exhibits any characteristic after the delisting criteria described herein have been applied, it must continue to be managed as a characteristically hazardous waste.

1.2.1 U.S. EPA OSW Hazardous Waste Delisting Program

The U.S. EPA OSW developed the Hazardous Waste Delisting Program to allow facilities that generate Subtitle C hazardous wastes to petition to have their wastes exempted from the requirements of the Subtitle C hazardous waste program (see 40 CFR 260.22). The delisting process evaluates whether a waste would release hazardous chemicals to groundwater at concentrations exceeding acceptable levels (health-based numbers or HBNs). Generally, the greatest risks determined for waste constituents considered for delisting resulted from potential groundwater exposure—that is, chemical releases to groundwater and subsequent exposure via groundwater exposure pathways. The U.S. EPA OSW originally applied the U.S. EPA Composite Model for Landfills (EPACML) fate and transport model to estimate constituent concentrations in groundwater at a receptor well located downgradient from a landfill or surface impoundment (U.S. EPA 1990c and 1990h). The EPACML fate and transport model was used to determine a Dilution-Attenuation Factor (DAF), which estimates the degree of dilution and attenuation that a constituent would 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 results of the EPACML analyses, the DAFs, were used to compute the maximum acceptable constituent concentration (that is, the exit level) in the leachate of a waste proposed for delisting.

The EPACML was originally developed to compute DAFs and set regulatory levels for specific constituents for the Toxicity Characteristics Rule (TC Rule) (U.S. EPA 1990d). Since the application of the EPACML to the TC Rule and to delisting, the Agency has developed a number of improvements in the modeling method and the input data. The U.S. EPA Composite Model for Leachate Migration with Transformation Products (EPACMTP) is the product of these improvements made to the EPACML fate and transport model.

1.2.2 Regional Authorization

On October 10, 1995, U.S. EPA Administrator Carol M. Browner delegated authorization of the Hazardous Waste Delisting Program to U.S. EPA's 10 regional offices (61 *Federal Register* [FR] 32798). The U.S. EPA OSW in Washington, DC, had previously administered the Delisting Program. As a result of the Administrator's action, delisting petitions that require a federal decision are now being reviewed by the appropriate U.S. EPA regions, and the regions, as of October 10, 1995, have the authority to make decisions on delisting petitions. The Agency believes that decentralizing the delisting authority to the Regional Administrators will result in more timely responses to delisting petitions.

Under RCRA, states authorized to administer a delisting program in lieu of the federal program also may exclude wastes from hazardous waste regulations. Facilities that manage their wastes in a state with RCRA delisting authorization should petition that state rather than U.S. EPA for an exclusion. Even in unauthorized states, U.S. EPA encourages petitioners to contact state authorities to determine what procedures might be necessary for delisting under state laws.

1.2.3 Regional Program Modifications

Previously, U.S. EPA OSW delisting evaluations applied the EPACML fate and transport model for determining potential chemical releases to groundwater (U.S. EPA 1991b). However, the EPACML had limitations, such as the inability to predict DAFs on a chemical-specific basis. After receiving authority to administer the Delisting Program, U.S. EPA Region 6 initially made two enhancements to the delisting process: (1) application of a new fate and transport model to calculate waste volume-specific DAFs, and (2) evaluation of additional exposure pathways. U.S. EPA Region 6 maintained the U.S. EPA OSW requirement to evaluate petitioned wastes on the basis of waste volume and investigated improvements made to the

EPACML that had been incorporated into the EPACMTP. Following review of the EPACMTP and the available literature, U.S. EPA Region 6 adopted the EPACMTP fate and transport model to develop DAFs in order to estimate the risk associated with exposure via groundwater pathways for delisting purposes.

The EPACMTP has been used to compute DAFs for the proposed Hazardous Waste Identification Rule (HWIR) (U.S. EPA 1995a) and the proposed Petroleum Refining Listing Rule (U.S. EPA 1998c). For the HWIR, the EPACMTP was used to determine (for 192 chemicals) waste volume-generic DAFs, which are based on a range of waste management unit areas (waste volumes) identified in a national survey of waste management units (including landfills and surface impoundments). A DAF represents the amount of dilution and attenuation expected to occur in groundwater as a chemical migrates to a potential exposure point at a downgradient receptor well.

The U.S. EPA evaluates petitions on the basis of a specific volume of waste. To do so, U.S. EPA Region 6 revised the EPACMTP to develop waste volume-specific DAFs. This was accomplished by using the EPACMTP to compute DAFs for a range of waste volumes for each waste disposal scenario (landfill and surface impoundment) and then developing regression equations for each disposal scenario that can be used to compute a DAF as a function of a specific waste volume (see Section 2.2.4). The U.S. EPA Region 6 Delisting Program performs two analyses of a petitioned waste: (1) a screening analysis that uses waste volume-specific DAFs to back-calculate maximum TCLP waste constituent concentrations at the prescribed risk levels for groundwater exposure pathway analyses and (2) an aggregate risk and hazard analysis that uses the waste volume-specific DAFs described herein. For further information on the development of waste volume-specific DAFs, refer to the document titled "Application of EPACMTP to Region 6 Delisting Program: Development of Waste Volume-Specific Dilution Attenuation Factors" (U.S. EPA 1996a).

In developing DRAS version 3, the EPACMTP was rerun with updated parameters, databases, and algorithms consistent with OSW use of the model. In addition, MINTEQA2 adsorption isotherms were used in conjunction with EPACMTP, resulting in some DAFs that vary based on the initial leachate concentration in the landfill or surface impoundment. The algorithms in DRAS have been modified accordingly to handle DAFs that vary by input concentration.

In a second enhancement to the delisting process, U.S. EPA Region 6 included additional exposure pathways in the delisting petition evaluation process to ensure that all potential exposure scenarios are addressed in the risk assessment. These additional pathways include (1) dermal contact with and inhalation of volatiles during bathing or showering with groundwater, (2) ingestion of drinking water from surface water bodies, (3) ingestion of contaminated fish, (4) inhalation of windblown particulates and volatiles from a waste management unit, and (5) ingestion of soils contaminated with windblown waste constituent particulates. These additional pathways allow a more complete evaluation of potential human health risks resulting from potential chemical releases of delisted wastes. For each exposure pathway, the appropriate chemical-specific factors are used to predict the risk to the sensitive receptor from the potential exposure to chemical contaminants. For instance, the effects of indoor inhalation exposure to volatile constituents may be comparable to or greater than those of ingestion exposure through drinking water (McKone 1987), and exposure from ingestion of contaminated fish may be significant because of bioaccumulation of each chemical constituent in fish tissue.

1.2.4 Delisting Risk Assessment Software (DRAS)

To evaluate delisting petitions in a timely manner, the U.S. EPA Region 6 Delisting Program developed a Windows-based program called the Delisting Risk Assessment Software (DRAS), that analyzes the risks and hazards posed by the constituents of a waste petitioned for delisting. Specifically, the DRAS performs two types of analyses: screening-level analyses and aggregate risk and hazard analyses. The results of these analyses may be viewed on screen, imported directly to word processing software, or printed in document-ready form. The screening-level analyses compute chemical-specific exit values or “delisting levels” for multi-year delistings. The aggregate risk and hazard analyses compute the aggregate carcinogenic risk and noncarcinogenic hazard indices (HI) for a waste petitioned for a one-time delisting. The delisting levels and aggregate risk and hazard estimates are calculated using modeled, medium-specific chemical concentrations and standard U.S. EPA exposure assessment and risk characterization algorithms.

Sections 1.2.4.1 and 1.2.4.2 provide additional discussion regarding calculation of delisting levels and calculation of aggregate risks and hazards, respectively.

1.2.4.1 Calculating Delisting Levels for Multi-year Delistings

In addition to alerting the user to the most limiting and most sensitive combination of exposure pathway and receptor, the DRAS provides the calculated chemical-specific delisting level for that combination. A delisting level is the maximum allowable concentration for each constituent of a waste petitioned for a multi-year delisting. For each waste constituent, the DRAS computes a total delisting level (in milligrams per kilogram) and a TCLP delisting level (in milligrams per liter). The TCLP delisting levels for the groundwater exposure pathways are calculated with standard risk assessment algorithms and with groundwater chemical concentrations at the point of exposure (POE) derived from waste volume-specific DAFs using the EPACMTP fate and transport model. The chemical-specific total delisting levels for the surface exposure pathways are calculated with standard risk assessment algorithms and with predicted chemical concentrations at the POE.

The analysis identifies the pathway-receptor combination that is the limiting combination or, in the case of multiple pathway-receptor combinations that fail the screening analysis, the most sensitive combination of pathway and receptor. This analysis shows the user the degree to which the waste's TCLP or total waste concentration exceeds the delisting level. The program also provides (in a print-ready summary table) all the calculated delisting levels for all pathway-receptor combinations.

1.2.4.2 Calculating Aggregate Risks and Hazards

In addition to calculating delisting levels for multi-year standard delistings, the DRAS performs a aggregate risk assessment for disposal of petitioned wastes in a landfill or surface impoundment waste management unit as a one-time delisting. If the delisting petition is for a one-time exclusion, the results of the aggregate risk assessment may be used in lieu of the delisting levels. A one-time delisting does not require the Agency to establish monitoring concentrations that must be met by each batch of waste to be managed under a promulgated exclusion. Therefore, the user may bypass the delisting levels, which are set at relatively conservative risk levels, in favor of the aggregate risk assessment process that employs the Agency's target risk levels (see Chapter 4 on target levels).

Computing the aggregate risk for a petitioned waste provides the user with detailed analysis of the petitioned waste. The DRAS indicates which chemicals and which pathways and/or receptors are driving the risk for a particular waste. The DRAS computes the aggregate carcinogenic risk by summing the carcinogenic risks for all waste constituents for a given exposure pathway and then summing the carcinogenic risks for each pathway analyzed in the delisting risk assessment. The DRAS computes the aggregate noncarcinogenic risk by summing the noncarcinogenic HQs for all waste constituents for a given exposure pathway and then summing the noncarcinogenic hazards associated with each exposure pathway analyzed. If the aggregate noncarcinogenic hazard exceeds the allowable level, the user should refer to Appendix A-4. Chemical-specific hazards may be apportioned by target organ.

1.3 DELISTING REFERENCE DOCUMENTATION

A number of delisting process documents have been developed to provide guidance specific to elements of the delisting process, including delisting petition preparation, waste sampling, sample quality assurance/quality control (QA/QC), and risk and hazard assessment. Delisting docket materials and other relevant reference documents are available and have also been used to support the delisting process. These documents and materials are briefly described below.

1.3.1 Delisting Process Documents

Documents available to guide a user through the delisting process are described below.

Delisting Guidance Manual. The “U.S. EPA Region 6 RCRA Delisting Program Guidance Manual for the Petitioner” (U.S. EPA 1996e) provides guidance to individuals who may be interested in submitting a petition to exclude or “delist” a listed hazardous waste generated at a particular facility from the lists of hazardous wastes in 40 CFR Part 261, Subpart D. U.S. EPA recognizes that a specific listed waste generated at a particular facility may not meet the criteria for which the waste was originally listed. The manual provides guidance on how to satisfy the procedures set forth in 40 CFR 260.20 and 260.22 whereby any individual can petition the Agency for a regulatory amendment to exclude a listed waste generated at a particular facility.

Risk Assessment Software User’s Manual. U.S. EPA Region 6 has developed a separate user’s manual to support the DRAS. The “U.S. EPA Region 6 RCRA Delisting Risk Assessment Software User’s Manual” (U.S. EPA 1998a) provides the user with the necessary information for installing and running the Windows-based risk assessment software. The user may access this manual directly through the Windows-based risk assessment software or may refer to a hard copy.

Application of Waste Volume-Specific DAF Document. U.S. EPA developed the report titled “Application of EPACMTP to Region 6 Delisting Program: Development of Waste Volume-Specific Dilution Attenuation Factors” to describe its approach for adapting the EPACMTP to the U.S. EPA Region 6 Delisting Program (U.S. EPA 1996a). The EPACMTP model computes individual DAFs that represent the decrease in concentration of a chemical as a result of its leaching from a waste management unit and its subsequent transport in the subsurface unsaturated and saturated zones to a receptor well. The EPACMTP was developed to compute DAFs as a function of a number of input parameters, including waste management unit area. However, waste volume is typically a derived input parameter, and the Delisting Program evaluates wastes based on specific waste volumes. Therefore, U.S. EPA Region 6 modified the EPACMTP model to compute DAFs for the range of waste volumes typically encountered in the evaluation of hazardous waste delisting petitions. The modifications to the EPACMTP and the analyses performed to derive 90th percentile DAFs for 192 chemicals (those listed in the proposed HWIR [U.S. EPA 1995a]) as a function of waste volume for wastes disposed of in landfills and surface impoundments are summarized in the 1996 report.

1.3.2 Delisting Docket Materials

Three delisting risk assessment evaluations have been performed to date by the U.S. EPA OSW to determine the risks and hazards associated with disposing of delisted wastes in nonhazardous waste management units. Specifically, the Agency has evaluated risks and hazards associated with contaminant releases to groundwater, surface water, and air and has documented the calculations performed in these evaluations in the dockets associated with each petition review. These three delisting risk evaluations performed by U.S. EPA are addressed in the following documents:

- U.S. EPA. 1993a. “Docket Report on Evaluation of Contaminant Releases to Surface Water Resulting from Conversion System’s Petitioned Waste.” August 27.
- U.S. EPA. 1993c. “Docket Report on Evaluation of Air Emissions Resulting from Conversion Systems, Inc.’s, Petitioned Waste.” September 9.
- U.S. EPA. 1994a. “Docket Report on Evaluation of Contaminant Releases to Air from U.S. Department of Energy Hanford’s Petitioned Waste.” May 27.

1.3.3 Other Relevant Reference Documents

The algorithms that the U.S. EPA Region 6 DRAS uses to compute the potential risks to human health associated with a waste petitioned for delisting are adapted from the following documents:

- U.S. EPA. 1988a. "Risk Assessment Guidance for Superfund (RAGS), Volume I: Human Health Evaluation Manual (Part A)." Office of Emergency and Remedial Response. Washington, DC EPA/540-1-89/002.

- U.S. EPA. 1991a. "Risk Assessment Guidance for Superfund (RAGS) Volume I: Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals)" (hereinafter referred to as RAGS Part B). Office of Emergency and Remedial Response. Washington, DC Publication No. 9285.7-013.

For dermal pathways, which are not covered in RAGS Part B, the DRAS uses the algorithms presented in the following document:

- U.S. EPA. 1992b. "Dermal Exposure Assessment: Principles and Applications, Interim Report." Office of Health and Environmental Assessment. Washington, DC. EPA/600/8-91/011B. January.

Additional exposure and risk assessment algorithms for shower inhalation of groundwater were obtained from the nongroundwater pathway risk assessment addressed in the following document:

- U.S. EPA. 1997a. "Supplemental Background Document; NonGroundwater Pathway Risk Assessment; Petroleum Process Waste Listing Determination." OSW. Research Triangle Park, North Carolina. March 20.

This background document contains the assumptions and equations used to evaluate the shower, bathroom, and house inhalation pathways for groundwater that were in turn used to determine the risks associated with specific petroleum refinery wastes. The docket materials contain the equations and assumptions that the U.S. EPA Headquarters Delisting Program used to evaluate delisting petitions with regard to the surface water and air exposure pathways.

Additional information on multipathway risk assessment algorithms and the EPACMTP fate and transport model used for the proposed HWIR (U.S. EPA 1995a) is provided in the following FR notice and background document for the HWIR:

- U.S. EPA. 1995a. “Hazardous Waste Management System: Identification and Listing of Hazardous Waste—Hazardous Waste Identification Rule (HWIR).” OSW. Washington, DC 60 FR 66344.

- U.S. EPA. 1995b. “Technical Support Document for HWIR: Risk Assessment for Human and Ecological Receptors.” Volumes I and II. OSW. Washington, DC

Details on the assumptions and input parameters used for the EPACMTP are provided in the following documents:

- U.S. EPA. 1997e. *EPA's Composite Model for Leachate Migration with Transformation Products, EPACMTP*: *User's Guide*. Office of Solid Waste, Washington, D.C.

- U.S. EPA. 1996b. *EPACMTP Background Document*. Office of Solid Waste. Washington, D.C. September.

- U.S. EPA. 1996c. *EPACMTP Background Document for the Finite Source Methodology for Chemicals with Transformation Products and Implementation of the HWIR*. Office of Solid Waste. Washington, D.C. September.

- U. S. EPA. 1997h. *Analysis of EPA's Industrial Subtitle D Databases used in Groundwater Pathway Analysis of the Hazardous Waste Identification Rule (HWIR).* Office of Solid Waste, Washington, DC. September.

- U.S. EPA. 1996d. *Background Document for EPACMTP: Metals Transport in the Subsurface, Volume 1: Methodology*. Office of Solid Waste. Washington, D.C. August.

- U.S. EPA. 1999. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) Background Document for Metals; Volume 2: Sorption Isotherms.* Office of Solid Waste, Washington, DC. August.

Every effort was made to maintain consistency with the U.S. EPA Region 6 “Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities - Peer Review Draft” (U.S. EPA 1998b) at the time DRAS was first released. The contaminant release and risk assessment algorithms and parameter values used for the DRAS have been compared to the draft Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (HHRAP) in order to ensure consistency between the two approaches. The HHRAP has since been finalized with a number of modifications and updates (U.S. EPA 2005), however, the changes do not impact the methodology used in the DRAS.

1.4 DOCUMENT ORGANIZATION

This section presents an overview of the DTSD's organization. The DTSD is arranged in a user-friendly format to present the technical procedures used to conduct a risk assessment for a petitioned waste:

- Chapter 1 describes the objectives of this DTSD and provides background and reference information for the U.S. EPA Delisting Program.
- Chapter 2 describes the methods used to estimate chemical releases from waste management units to groundwater, soils, air, and surface water as well as the calculation of contaminant concentrations in each of these media.
- Chapter 3 describes the selection of exposure scenarios, including the receptor locations, and the parameters and assumptions used to quantify exposure.
- Chapter 4 describes the methodology used to compute target carcinogenic risks and HQs as well as aggregate carcinogenic risks and HIs.
- Chapter 5 discusses the uncertainties involved in the risk and hazard analyses performed for the U.S. EPA Delisting Program.
- Chapter 6 contains full citations to the items referenced throughout the DTSD. the U.S. EPA Delisting Program.

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, as incorporated in DRAS, involves performing a risk assessment for petitioned wastes that are disposed of in two waste management units of concern to the U.S. EPA 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. In preparing DRAS, U.S. EPA Region 6 focused its delisting petition evaluations on liquid and solid wastes disposed of in surface impoundments and landfills, respectively. Wastes disposed of in other waste management units will be evaluated on a case-by-case basis.

In DRAS, U.S. EPA Region 6 assumed 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 Technical Background Document" and the "EPACMTP Parameters/ Data Background Document" (U.S. EPA 2003a and 2003b).

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 and U.S. EPA 1997h). Parameters and assumptions used to estimate infiltration of leachate from a landfill are provided in the "EPACMTP Technical Background Document" and the "EPACMTP Parameters/ Data Background Document" (U.S. EPA 2003a and 2003b).

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 groundwater fate and transport model used was the EPACMTP (U.S. EPA 2003a). Receptor well concentrations for both carcinogens and noncarcinogens, and for both degraders and nondegraders were determined using the finite source option within the EPACMTP.

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 both uncertainty and variability in the model input parameter values.

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 uncertainty and variability in model parameter values on predicted receptor well concentrations

The subsurface as modeled with the EPACMTP consists of an unsaturated (vadose) 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. However, for the purposes of the delisting analysis, both the vadose zone and saturated zone were modeled.

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 toxic daughter products that are of concern as well. 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, this infiltration 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, respectively. 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. **Adsorption Reactions**. The EPACMTP computes chemical reactions involving adsorption. The EPACMTP assumes that sorption of organic compounds in the subsurface can be 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 one of two types of adsorption isotherms: 1) pH-dependent, empirically-derived isotherms (U.S. EPA 1990g); or 2) 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, and 1999).

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 rate constants using soil and aquifer temperature and pH values (U.S. EPA, 1993e and 1996c). 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 Technical Background Document” and the “EPACMTP Parameters/ Data Background Document” (U.S. EPA 2003a and 2003b)). 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, infiltration and recharge rates, pulse duration, the fraction of hazardous waste in the waste management unit, and the density of the 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 landfills were obtained from the 1987 U.S. EPA survey of industrial Subtitle D waste facilities in the United States (Westat 1987 and U.S. EPA 1997h). Relevant information for surface impoundments was obtained from the 2001 EPA’s industrial Surface Impoundments Study (US EPA 2001). Derivation of the parameters for each type of waste management unit is described in the “EPACMTP Technical Background Document” and the “EPACMTP Parameters/ Data Background Document” (U.S. EPA 2003a and 2003b).

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 mass of contaminant in the landfill, infiltration rate, landfill dimensions, and leachate concentration (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 50 years for surface impoundments). For a finite-source scenario, the model can calculate either the peak receptor well concentration for noncarcinogens or the highest 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. 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 probabilistic (Monte Carlo) mode, and many of 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 5 of the "EPACMTP Technical Background Document" (U.S. EPA 2003a). The Monte Carlo analysis enables parametric uncertainty and/or variability to be quantitatively accounted for. The flow and transport modules of EPACMTP are linked to a Monte Carlo driver that permits a probabilistic evaluation of variability and/or uncertainty in model input parameters, as described by specified (joint) probability distributions. The resulting sequence of receptor well concentrations (one 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 in Table 2-1 below.

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; LF: depleting source for organics, constant concentration pulse source for metals SI: constant concentration pulse for all constituents
Exposure Evaluation	Downgradient groundwater receptor well; maximum well concentration; 10,000-year modeling period
Regulatory Protection Level	90 percent
Source-Specific Parameters	
Parameter	Description or Value
Waste Unit Depth	Derived (for landfills), and site-based randomly selected from the EPA industrial SI data base for (surface impoundments)
Waste Unit Area	Site-based, randomly selected from OPPI database (for landfills), and from the EPA industrial SI data base (for surface impoundments)
Waste Unit Volume	User-specified
Infiltration Rate	
Landfill	Site-based, derived from water balance using HELP model
Surface Impoundment	Site-based, derived from impoundment depth using Darcy’s law
Leaching Duration	
Landfill	Derived, continues until all constituents have leached out
Surface Impoundment	50 years (operational life of unit)
Chemical-Specific Parameters	
Parameter	Description and Source
Decay Rate	
Organic Constituents	Hydrolysis rate constants compiled by U.S. EPA ORD (U.S. EPA 1993e)
Metals	No decay
Sorption	
Organic Constituents	K_{oc} constants compiled by U.S. EPA ORD (U.S. EPA 1993e)
Metals	MINTEQA2 sorption isotherm coefficients (K_d) for Ba, Cd, Cr (III), Hg, Ni, Pb, Ag, Zn, Cu, V, Be, Mo, As(III), Cr(VI), Se(VI), Tl, Sb(V), Mn, F, As(V), and Se(IV) (U.S. EPA 2003b); empirical K_d distributions for Fe and Sn

Unsaturated Zone-Specific Parameters

Parameter	Description and Source
Depth to Groundwater	Site-based, from API and USGS hydrogeologic database (Newell and others 1989 and 1990)
Soil and Unsaturated Zone Properties:	U.S. EPA ORD data, based on national distribution of three soil types (sandy loam, silt loam, silty clay loam)
Bulk Density	Carsel and others 1992
Saturated Conductivity	Carsel and Parrish 1992
Moisture Retention Parameters (alpha and beta)	Carsel and Parrish 1992
Residual Water Content	Carsel and Parrish 1992
Saturated Water Content	Carsel and Parrish 1992
Percent Organic Matter	Carsel and others 1992
Dispersivity	Gelhar’s analysis of EPRI field data (US EPA 2003b and EPRI 1985)

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 (Newell and others 1989 and 1990)
Hydraulic Conductivity	Site-based, from API and USGS hydrogeologic database (Newell and others 1989 and 1990)
Hydraulic Gradient	Site-based, from API and USGS hydrogeologic database (Newell and others 1989 and 1990)
Porosity	Effective porosity derived from national distribution of aquifer particle diameter
Bulk Density	Derived from porosity
Dispersivity	Derived from distance to receptor well (Gelhar and others 1992 and U.S. EPA 2003b)
Groundwater Temperature	Site-based, from USGS regional temperature map (Todd 1980 and US EPA 2003b)
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 (US EPA 1993)
Angle Off-Center	Uniform within ± 90° from plume center line (no restriction to always lie 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 and depth and the infiltration rate (U.S. EPA 1996b and 1996c). It is important to note the difference in leaching duration for each waste management unit type (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 (ë) (for organic constituents), the normalized distribution coefficient for organic carbon (K_{oc}) (for sorption estimation for organics), and the adsorption 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 also referred to as the effective partition coefficient. For details on metals isotherms from MINTEQA2, see Appendix A-6, *Application of CMTP and MINTEQA2 for determining DAFs*.

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 soil moisture 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 a vapor phase 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 values for this parameter group include depth to groundwater and soil hydraulic parameters and other soil properties, 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 concentration in groundwater is dilute, and the chemical is present in the aqueous and aquifer solid phases only.

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

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 and/or vertical 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 (X-dimension)
- Angle off of the plume centerline (Y-dimension)
- Depth of the intake point (Z-dimension; defined 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 decrease 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 (that is, 90 percent of the resulting receptor well concentrations are less than this concentration). The DAF is then calculated by dividing the initial (C_L) waste concentration by the predicted groundwater concentration at the 90th percentile as shown in Equation 2-1.

$$DAF = \frac{C_L}{C_{gw}} \tag{2-1}$$

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.

$$C_{LMAX} = HBN \cdot DAF \tag{2-2}$$

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

For further details about using the CMTP in conjunction with MINTEQA2 to derive DAFs, see Appendix A-6, *Application of CMTP and MINTEQA2 for determining DAFs*.

2.2.2.3 Derivation Waste Volume-Specific DAFs

Application of the EPACMTP to the U.S. EPA Delisting Program allows evaluation of specific waste volumes. To generate waste volume-specific DAFs, a number of EPACMTP runs were conducted, each run with a user-specified waste volume. For each Monte-Carlo realization, the user-specified waste volume was divided by a waste management unit area randomly drawn from a site in a regional site database, while retaining the site's original area/depth ratio. The regional site database contains waste management unit depth and area data obtained from a national survey of Subtitle D industrial landfill 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), and from the 2001 EPA's Industrial Surface Impoundment (SI) Study (US EPA 2001). The OPPI and SI survey data were compiled along with hydrogeologic data in the regional site database. The data for each waste management scenario are presented in "EPACMTP Parameters/ Data Technical Background Document" (U.S. EPA 2003b). For a complete description of the regional site-based modeling approach, refer to the "EPACMTP Technical Background Document" (U.S. EPA 2003a).

Rather than compute DAFs for each of 326 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 using the OPPI and SI data bases (U.S. EPA 1988b, 2001). 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 2003a, b). 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 lifetime 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 landfill lifetime waste volumes greater than 154,000 yd^3. U.S. EPA did not consider DAF scaling factors less than 1.0. Therefore, for lifetime landfill waste volumes greater than 154,000 yd^3, the waste volume-specific DAF is equal to the DAF computed based on the OPPI data base.

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

$$DAF_{sf} = 120{,}379 \cdot (V)^{-0.97952} \qquad \textbf{(2-3)}$$

where:

			Default
DAF_{sf}	=	DAF scaling factor (unitless)	computed
V	=	landfill lifetime 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 lifetime 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. For surface impoundment lifetime waste volumes greater than about 40,000 yd^3, the DAF scaling factor is equal to 1.0.

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

$$DAF_{sf} = 108{,}687 \cdot (V)^{-1.20644} \qquad \textbf{(2-4)}$$

where:

			Default
DAF_{sf}	=	DAF scaling factor (unitless)	computed
V	=	impoundment lifetime 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 Delisting Program: direct ingestion, dermal absorption, and shower inhalation. This section describes how the waste constituent concentration at the groundwater POE is calculated.

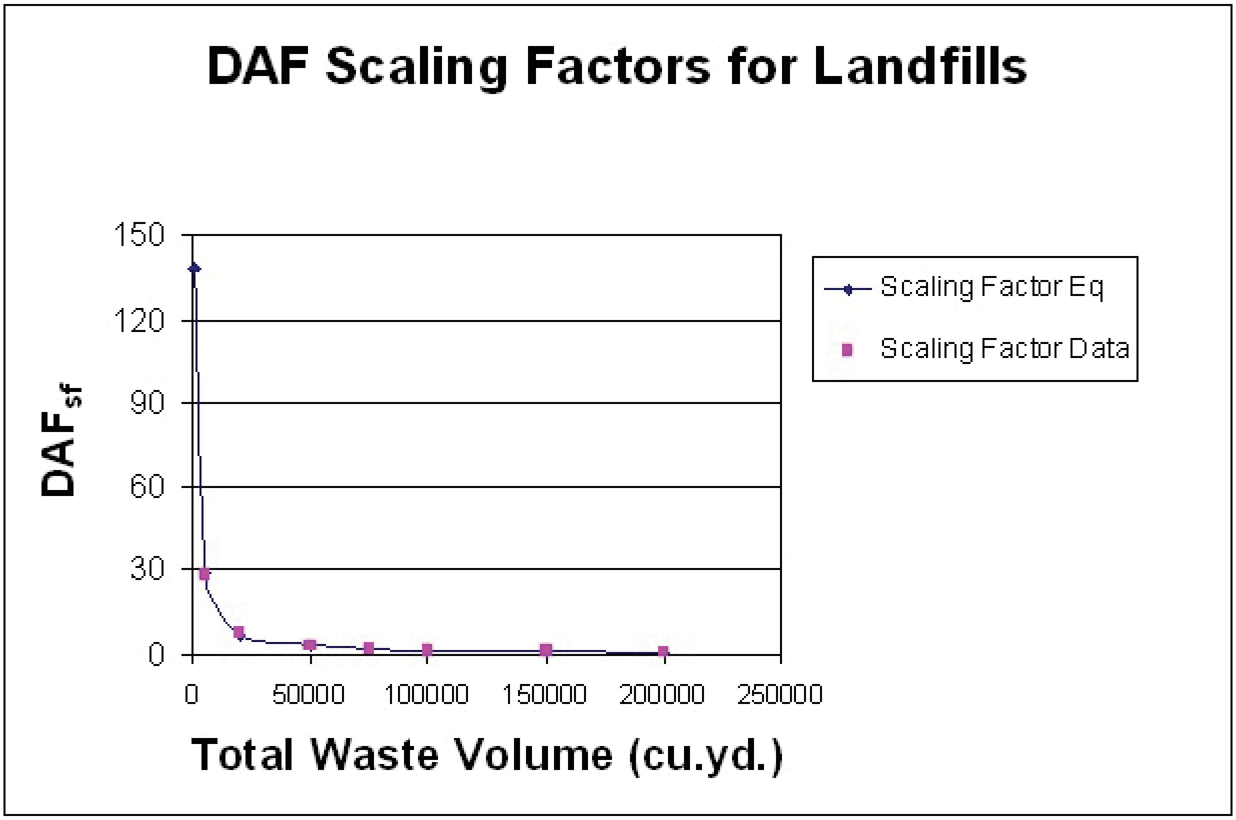

Figure 2-1. DAF Scaling Factors for Landfills

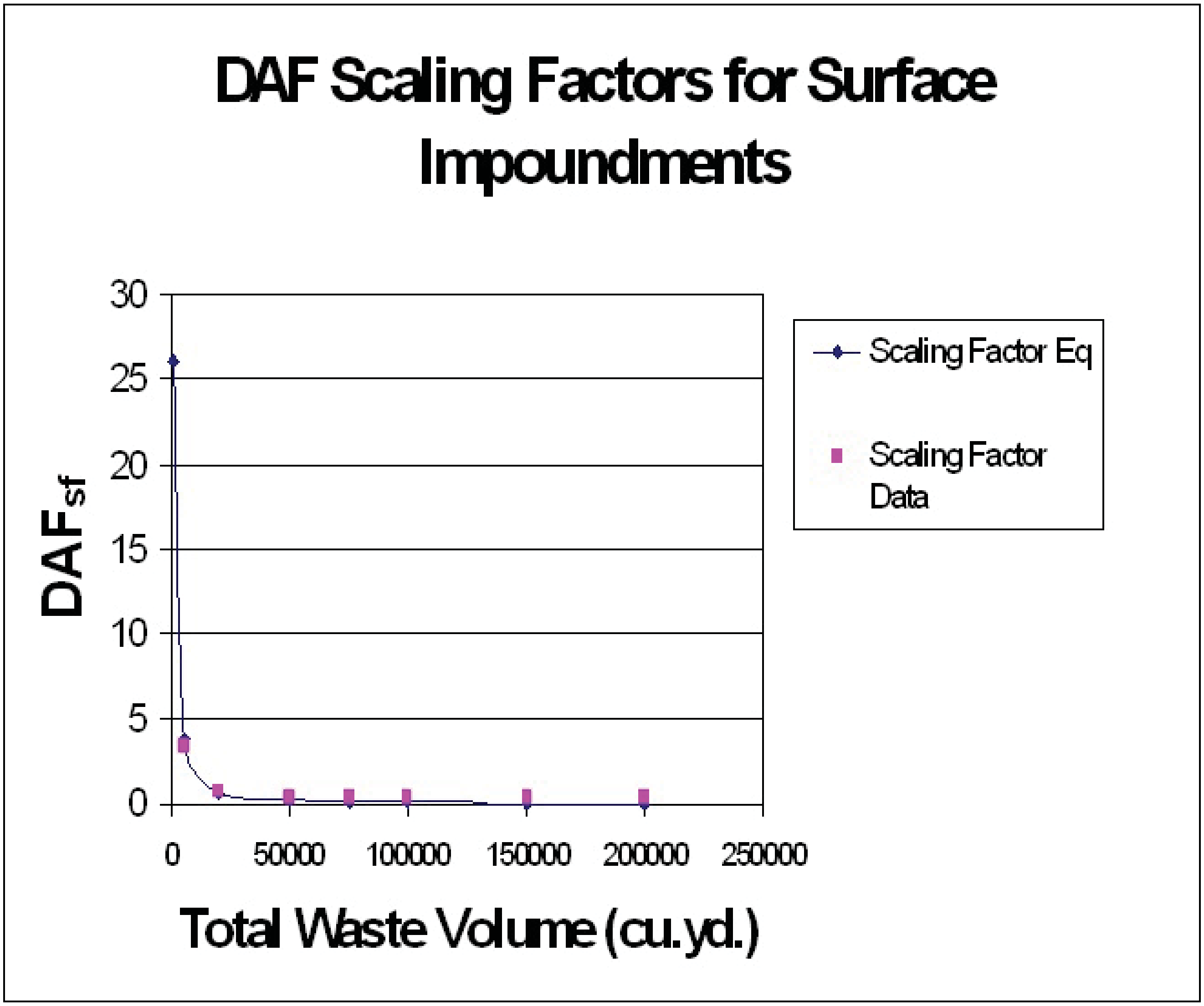

Figure 2-2. DAF Scaling Factors for Surface Impoundments

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

To adjust for the petitioned waste volume, the EPACMTP-modeled 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.

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 may be very high (that is, greater than 1,000,000) for some waste constituents.

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 — Particles

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 particles and their absorption into the lungs at the POE and (2) air deposition of particles and subsequent ingestion of the soil-waste mixture at the POE. To address inhalation and ingestion of particles, U.S. EPA Region 6 calculates particulate emissions resulting from wind erosion of soil-waste surfaces and from vehicular traffic over the waste 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, the DRAS 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 particles 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.

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

where:

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

and:

$$E_T = E_w + E_v + E_l \tag{2-8}$$

where:

			Default
E_T	=	total emission rate of particles 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 particles (those smaller than 10 microns [ì m] in diameter)[a] can be calculated using Equation 2-9.

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

where:

			Default
E_{w10}	=	wind erosion emission rate for particles up to 10 ì m (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 EPA 1985a)	1.33 (Appendix B, (U.S. 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 assumes 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:

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[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).

[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.

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

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

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

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	=	total 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 delisting program 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	=	annual volume of waste (yd^3/year)	delisting petition-specific

U.S. EPA delisting program 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.

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

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 (multi-year)

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

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

where:

			Default
E_{w30}	=	wind erosion emission rate for particles up to 30 ì m (g/hr)	calculated
E_{w10}	=	wind erosion emission rate for particles up to 10 ì m (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 particles up to 10 ì m and 0.8 for particles up to 30 ì m	(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 a minimum of 7.4 vehicles traverse the waste daily, based on the 95th percentile of vehicles per day reported in Table 3A-1 of the 1987 U.S. EPA survey of industrial Subtitle

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[c] Particles less than or equal to 30 ì m in diameter can be transported for considerable distances downwind; those larger than 30 ì m 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).

D waste facilities in the United States (Westat 1987 and U.S. EPA 1997h), and adding the number of daily trips based on the petitioned waste volume, *VKT* can be calculated using Equation 2-18.

$$VKT = Site\ Width \cdot \frac{1\ km}{1{,}000\ m} \cdot \frac{2\ trips}{roundtrip} \cdot \left[TPD_{\min} + \left(\frac{V \cdot \rho_w}{365\ days/year \cdot 10\ tons/load} \right) \right] \qquad \textbf{(2-18)}$$

where:

			Default
VKT	=	vehicle kilometers traveled (km-trips/day)	calculated
Site Width	=	$\sqrt{A_{exposed}}$ (m)	Equation 2-15
$\text{TPD}_{\min}$	=	minimum round trips per day	7.4 (95th percentile from Subtitle D Survey)
ñ$_w$	=	waste density (tons per cubic yard)	1.42 (U.S. EPA 1993a)
V	=	annual volume of waste (yd^3/year)	delisting petition-specific

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

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

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 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 realized 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/metric ton)	calculated
K_p	=	batch drop particle size multiplier (dimensionless): – 0.36 for particles up to 10 ì m – 0.73 for particles up to 30 ì m	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 particles up to 10 and 30 ì m in size as shown in Equations 2-22 and 2-23, respectively.

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

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

where:

			Default
E_{l10}	=	waste loading and unloading emission rate of particles up to 10 ì m (kg/metric ton)	calculated

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

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

where:

			Default
E_{l30}	=	waste loading and unloading emission rate of particles up to 30 ì m (kg/metric ton)	calculated

To convert the emission rate from kilogram per metric 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 tons per cubic yard (or 1.289 metric tons per cubic yard). 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:

$$\frac{kg}{metric\ ton} \cdot V \ \frac{yd^3}{yr} \cdot \frac{1.289\ metric\ tons}{yd^3} \cdot \frac{1{,}000\ g}{1\ kg} \cdot \frac{1\ yr}{365\ day} \cdot \frac{1\ day}{24\ hr}$$

Total Respirable Particulate Emissions

U.S. EPA Region 6 calculated the total annual average emissions of respirable particles (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.

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

where:

		Default
Q_{p10} =	emission rate of waste constituent particles up to 10 ì m (mg/s)	calculated
á	= mass fraction of waste constituent (unitless)	waste-specific (equal to total concentration in waste in mg/mg)
E_{w10}	= wind erosion emission rate of particles up to 10 ì m (g/hr)	Equation 2-9
E_{v10}	= vehicle travel emission rate of particles up to 10 ì m (g/hr)	Equation 2-19

E_{l10}	= waste loading and unloading emission rate of particles up to 10 ì m (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.

$$C_{avg} = \frac{2.03 \cdot Q_p}{\Sigma_z \cdot U \cdot L_v} \cdot F \tag{2-25}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	calculated
Q_p	=	emission rate of waste constituent particles (mg/s)	Equation 2-24
$\acute{O}_z$	=	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, $\acute{O}_z$, can be calculated using Equation 2-26 as described in the "Industrial Source Complex (ISC) Dispersion Model User's Guide" (U.S. EPA 1986a).

$$\Sigma_z = a \cdot L^b \tag{2-26}$$

where:

			Default
$\acute{O}_z$	=	vertical dispersion coefficient (m)	calculated
a	=	coefficient for stability class D	32.093 (from Appendix B, Table B-5)
L	=	distance from center of the uncovered waste area to compliance point 1,000 feet downwind (km)	(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 the (site width)/2 + 0.3048 km compliance point 1,000 feet (304.8 m) downwind (km)	(U.S. EPA 1986b)
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, ó$_y$, 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 Ó$_z$ 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 can be calculated using Equation 2-29. 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).

$$C_{inh} = C_{avg} \cdot 20 \; \frac{m^3}{day} \qquad \textbf{(2-29)}$$

where:

			Default
C_{inh}	=	concentration of waste constituents 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)

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 particles less than or equal to 30 ì m in size (that is, total suspended particles [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 particles up to 30 ì m 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 particles up to 30 ì m (mg/s)	calculated
á	=	mass fraction of waste constituent (unitless)	waste-specific (equal to total concentration in waste in mg/mg)
E_{w30}	=	wind erosion emission rate of particles up to 30 ì m (g/hr)	Equation 2-16
E_{v30}	=	vehicle travel emission rate of particles up to 30 ì m (g/hr)	Equation 2-20
E_{l30}	=	waste loading and unloading emission rate of particles up to 30 ì m (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 particles 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 particles less than or equal to 30 ì m 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 particles 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 particles) 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 particles) 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 ì m (0.001, 0.018, and 0.07 m/s, respectively).

To obtain an effective deposition velocity for particles less than or equal to 30 ì m in size, U.S. EPA Region 6 calculated the mean velocities for the two particle size ranges, 1 to 10 ì m (0.0095 m/s) and 10 to 30 ì m (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.

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

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
$\tilde{n}_b$	=	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.

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

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 assumed 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* (atm)	chem-specific (Appendix A-1)
MW_i	= mole weight of waste constituent *I* (g/mole)	chem-specific (Appendix A-1)
R	= molar gas constant (atm-m^3/mole-°K)	8.21×10^{-5}
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 particles, biochemical oxygen demand (BOD), or photodegradable constituents is introduced into a lagoon. The particles 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 an [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 (days)	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.

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

where:

			Default
tf	=	time for constituent concentration to reach 1 percent of C_o (days)	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.

$$K_l = 2.611\ x\ 10^{-7} \cdot U_{10}^{\ 2} \cdot \left[\frac{D_w}{D_{ether}}\right]^{2/3} \tag{2-42}$$

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 x 10^{-6} (U.S. EPA 1994a)

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

$$K_{eq} = \frac{H}{R \cdot T} \tag{2-43}$$

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 x 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.

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

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)
$\tilde{n}_a$	=	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.

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

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.

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

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.

$$A_s = A_{eroded} \cdot 907.185 \frac{kg}{ton} \cdot S_d \tag{2-50}$$

where:

			Default
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.

$$A_w = A_s \cdot F_{exposed} \tag{2-51}$$

---

[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.

where:

Default

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 (kg/acre/yr) — 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 run-on and run-off 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}).

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

w
here:

			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 2-53 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 soil and waste mass delivered to the surface water from the landfill and the background suspended solids concentration. The TSS is calculated as follows:

$$TSS = \left(\frac{A_w \cdot A}{Q_{stream}} \right) \cdot 10^6 \ mg/kg + SS_b \qquad \textbf{(2-54)}$$

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 2nd 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 ppm (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):

$$Kd_{sw} = f_{oc,\,sw} \cdot K_{oc} \qquad \textbf{(2-55)}$$

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 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.

Chapter 3
Exposure Scenario Selection

What's covered in Chapter 3:

- Recommended Exposure Scenarios
  - Adult Resident
  - Child Resident
- Exposure Scenario Locations
- Quantifying Exposure

This chapter summarizes the exposure scenarios selected for U.S. EPA risk assessment evaluation of wastes petitioned for delisting. The purpose of this chapter is to identify the types of human exposure to waste constituent emissions associated with the selected waste management scenarios (landfill and surface impoundment). The chapter addresses both identification of recommended exposure scenarios and selection of standardized exposure scenario locations.

An exposure scenario is a combination of exposure pathways through which a single receptor may be exposed to a waste constituent. Receptors may come into contact with waste constituent emissions associated with the selected waste management scenarios via two primary exposure routes, either (1) directly via inhalation or ingestion of water or (2) indirectly via subsequent ingestion of soil and foodstuffs (such as fish) that become contaminated by waste constituents through the food chain. Receptors may also be exposed to waste constituents released from a waste management unit to surface media (for example, via volatilization to air or via windblown particulate matter) or to groundwater (for example, via ingestion of groundwater).

Exposure pathways represent combinations of receptors and exposure routes. Each exposure pathway consists of four fundamental components: (1) a source and mechanism of waste constituent release (see Chapter 2); (2) a retention medium, or a transport mechanism and subsequent retention medium in cases involving media transfer of waste constituents (see Chapter 2); (3) an exposure route; and (4) a point of potential human contact with the contaminated medium, which is referred to as the POE and involves exposure of a specific receptor at a specific point.

The exposure scenarios recommended for evaluation in this chapter are generally conservative in nature and are not intended to be entirely representative of actual scenarios at all sites. Rather, they are intended to allow standardized and reproducible evaluation of risks across most sites and land use areas. Conservatism is incorporated to ensure protection of potential receptors not directly evaluated, such as special subpopulations, and land uses that are region-specific. The U.S. EPA Delisting Program believes that the recommended exposure scenarios and associated assumptions presented in this chapter are reasonable and conservative and that they represent a scientifically sound approach that allows protection of human health and the environment while recognizing the uncertainty associated with evaluating real-world exposure. Unless site-specific conditions warrant an exception approved by the delisting authority, the U.S. EPA Delisting Program recommends that these scenarios be used, at a minimum, in an initial site evaluation to identify primary risk concerns. Any exceptions, particularly deletion or modification of a recommended exposure scenario, should be well documented and approved by the delisting authority.

The following sections discuss (1) the default exposure scenarios, (2) selection of the POEs to be evaluated in the risk assessment, and (3) quantification of exposure.

3.1 DEFAULT EXPOSURE SCENARIOS

Adult and child residents are the two receptors evaluated in this analysis. These receptors are discussed below along with special subpopulation characteristics.

3.1.1 Adult Resident

The adult resident exposure scenario is evaluated to account for the combination of exposure pathways to which an adult receptor may be exposed in an urban or rural (non-farm) setting. The adult resident is assumed to be exposed to waste constituents from an emission source through the following exposure pathways:

- Surface Pathways
    - Direct inhalation of vapors and particles
    - Ingestion of fish
    - Ingestion of drinking water from surface water sources
- Groundwater Pathways
    - Ingestion of drinking water from groundwater sources

- Dermal absorption from groundwater sources via bathing
- Inhalation from groundwater sources via showering

Further discussion of these exposure pathways, including numerical equations, parameter values, and waste constituent-specific inputs, can be found in Chapter 4. No other exposure pathways are evaluated for the adult resident exposure scenario.

3.1.2 Child Resident

For child receptors, the U.S. EPA Delisting Program evaluates two exposure pathways: (1) the dermal absorption while bathing with groundwater pathway and (2) the ingestion of soil contaminated with air particulate pathway. Child residents (1 to 6 years old) were not selected as receptors for the groundwater ingestion and inhalation pathways, the surface water pathways, or the direct air inhalation pathways because the adult resident receptor scenario has been found to be protective of children with regard to these pathways (U.S. EPA 1995a, 1995b). For most routes of exposure (*e.g.*, drinking water ingestion, air inhalation, food ingestion), the intake rate remains relatively proportional to body weight throughout the lifetime. Child intake rates may be slightly higher (U.S. EPA 1990f, 1997b). However, the exposure duration for children would be considerably shorter than for adults. After consideration of the weighted average of exposure over a lifetime, the adult evaluation becomes the more sensitive scenario and no increased adverse effects would be seen by considering the child exposure pathway. The dermal absorption pathway and the soil ingestion pathway are two exceptions.

The dermal absorption while bathing with groundwater exposure pathway is evaluated differently for child residents than it is for adult residents because of the following considerations: (1) the ratio of exposed skin surface area to body weight is slightly higher for children than for adults, resulting in a slightly larger average daily exposure for children than for adults; and (2) the exposure duration for such children is limited to 6 years, thus lowering the lifetime average exposure to carcinogens. Typically, the adult scenario is more protective with regard to carcinogens (because of the longer exposure duration), and the child scenario is more protective with regard to noncarcinogens (because of the greater skin surface area to body weight ratio) (U.S. EPA 1995b).

For soil ingestion, the receptor is a resident who ingests soil from childhood through adulthood. Young children (to 6 years) typically ingest a greater quantity of soil than adults while at the same time have a much smaller average body weight than an adult. This results in a greater average daily exposure for children than adults. However, the exposure duration for the child is limited to the 6 years that a young individual would fall into the appropriate age range, thus lowering lifetime average exposure for carcinogens. Therefore, OSW policy for assessing exposure via soil ingestion to carcinogenic constituents is to consider exposure to a single individual through childhood and adulthood for soil ingestion exposures. This practice accounts for both the greater average daily dose of childhood and the longer exposure duration of adulthood. For noncarcinogens, the greater average daily dose has the stronger influence on the effects assessment and the child exposure pathway is considered. Further discussion of the exposure pathways, including algorithms, parameter values, and waste constituent-specific inputs, can be found in Chapter 4.

3.1.3 Special Subpopulation Characteristics

Special subpopulations are defined as human receptors or segments of the population that may be at higher risk because of receptor sensitivity to waste constituents (for example, the elderly, infants and children, and fetuses of pregnant women). The assumptions and methodology specified in the DTSD to complete the risk assessment (such as the conservative EPACMTP assumptions discussed in Section 2.2.2 and the use of reference doses [RfD] developed to account for toxicity to sensitive receptors) have been developed to protect human health, including that of special subpopulations.

3.2 EXPOSURE SCENARIO LOCATIONS

This section describes the exposure locations (POEs) for the groundwater, surface water, and air pathways (Sections 3.2.1, 3.2.2, and 3.2.3, respectively). A delisted waste may be disposed of by the petitioner in any authorized landfill or surface impoundment without regulatory control (see Section 1.2.1); therefore, the POEs selected for the risk assessment of a waste petitioned for delisting are individually standardized.

3.2.1 Downgradient Groundwater-Drinking Water Well POE

The EPACMTP, a probabilistic groundwater fate and transport model, is used to predict groundwater constituent concentrations at a hypothetical receptor well located downgradient from a waste management unit (as described in Section 2.2). This receptor well represents the POE at which potential exposure to the groundwater is measured; that is, the estimated constituent concentration at the POE is used to assess risk. The distance to the well is based on the results of a U.S. EPA OSW survey performed to determine the distance to the nearest drinking water well downgradient from municipal landfills (U.S. EPA 1988b). The survey data are entered in the EPACMTP model as an empirical distribution: minimum = 0 m, median = 427 m, and maximum = 1,610 m (approximately 1 mile) (U.S. EPA 1997e). At most of the facilities included in the survey, the direction of ambient groundwater flow is not precisely known; therefore, it cannot be ascertained whether the nearest receptor well is located along the plume center line or even within the plume. To reflect uncertainties and variations in the location of the receptor well in relation to the direction of ambient groundwater flow, the modeled well is located anywhere on the downgradient side of the waste management unit within a 1-mile radius (U.S. EPA 1997e). In contrast to the 1990 TC Rule (U.S. EPA 1990d), there is no requirement that the well lie within the leachate plume.

For carcinogenic waste constituents, the exposure concentration is defined as the maximum 30-year average receptor well concentration; for noncarcinogens, the exposure concentration is taken to be the highest receptor well concentration during the modeled 10,000-year period (U.S. EPA 1996b). A 10,000-year limit was imposed on the exposure period; that is, the calculated exposure concentration is the peak or highest 30-year average concentration occurring within 10,000 years following the initial release from the waste management unit (U.S. EPA 1996b). The Monte Carlo fate and transport simulation provides a probability distribution of receptor well concentrations as a function of expected leachate concentration. Regulatory leachate concentration limits were determined using a back-calculation procedure corresponding to a 90th percentile protection level. These leachate concentration limits are the maximum leachate concentrations at which groundwater concentrations at 90 percent of the hypothetical downgradient receptor wells are not expected to exceed the given toxicity benchmark (usually either an MCL or HBN value) (U.S. EPA 1996b).

3.2.2 Downgradient Surface Water Body POE

Human exposure routes for surface water include ingestion of surface water used as drinking water and ingestion of fish from nearby surface water bodies. For the surface water ingestion exposure route, the surface water POE modeled is a fifth-order stream 100 m from the waste management unit (U.S. EPA 1993c). Fifth-order streams were chosen for analysis because U.S. EPA assumes that a fifth-order stream is the smallest stream capable of serving as a community water supply (U.S. EPA 1993c). The assumption of a 100-m distance to the nearest surface water body is a conservative assumption based on available data. A U.S. EPA survey of municipal landfill facilities showed that 3.6 percent of the surveyed facilities are located within 1 mile of a river or stream and that the average distance from these facilities to the closest river or stream is 586 m (U.S. EPA 1988b).

For the fish ingestion exposure route, a second-order stream was chosen for analysis. This stream segment was determined to be the smallest stream capable of supporting fisheries (U.S. EPA 1993c). The surface water body POE for collection of fish is assumed to be 100 m downgradient from the disposal facility (U.S. EPA 1993c).

3.2.3 Downwind POE

Human exposure to emissions of windblown particulate from landfills and to emissions of volatiles from landfills and surface impoundments includes inhalation of these particulate and volatiles. These are the exposure routes evaluated for the U.S. EPA Delisting Program. For the air pathway, the POE is 305 m (1,000 feet) downwind of the facility (U.S. EPA 1993c, 1994a).

3.3 QUANTIFICATION OF EXPOSURE

This section describes the factors to be evaluated in quantifying the exposure received under each of the exposure scenarios outlined above. The calculation of constituent-specific exposure rates for each exposure pathway evaluated is based on (1) the estimated concentration in a given medium as calculated in Chapter 2, (2) the contact rate, (3) receptor body weight, and (4) the frequency and duration of exposure. This calculation is repeated for each constituent and for each exposure pathway included in an exposure scenario. Exposure pathway-specific equations are presented in the appropriate sections of Chapter 4. Sections 3.3.1 through 3.3.8 describe a generic exposure rate calculation and the exposure pathway-specific variables that may affect this calculation. The default exposure parameters (*e.g.*, body weight, skin surface area) used in RCRA delisting risk assessment are reported in the *US EPA Exposure Factors Handbook* (EFH) (U.S. EPA 1990f, 1997b).

3.3.1 Generic Exposure Rate Equation

Exposure to hazardous constituents is assumed to occur over a period of time. To calculate an average exposure per unit of time, the total exposure can be divided by the time period. In this analysis, the exposures are intended to represent reasonable maximum exposure (RME) estimates for each applicable exposure route. The RME approach is intended to combine upper-bound and mid-range exposure factors so that the result represents an exposure scenario that is both protective and reasonable, not the worst possible case (U.S. EPA 1991a). All exposures quantified in this analysis are (1) normalized for time and body weight, (2) presented in units of milligrams per kilogram of body weight per day, and (3) termed "intakes." Equation 3-1 is a generic equation used to calculate constituent dose (U.S. EPA 1997b).

$$I = \frac{C_{gen} \cdot CR_{gen} \cdot EF \cdot ED}{BW \cdot AT} \qquad \textbf{(3-1)}$$

where:

I = intake — the amount of constituent at the exchange boundary (mg/kg/day); for evaluating exposure to noncarcinogenic constituents, this intake is referred to as the average daily dose (ADD); for evaluating exposure to carcinogenic constituents, this intake is referred to as the lifetime average daily dose (LADD)
C_{gen} = generic constituent concentration — the media average concentration contacted over the exposure period (for example, mg/kg for soil and mg/L for water)
CR_{gen} = generic contact rate — the amount of contaminated medium contacted per unit of time or per event (for example, kg/day for soil and L/day for water) (upper-bound value)
EF = exposure frequency (days/yr) (upper-bound value)
ED = exposure duration (yrs) (upper-bound value)
BW = body weight of the receptor over the exposure period (kg) (average value)
AT = averaging time — the period over which the exposure is averaged (days); for carcinogens, the averaging time is 27,375 days, based on a lifetime exposure of 75 yrs; for noncarcinogens, averaging time equals exposure duration (in yr) multiplied by 365 days/yr

3.3.2 Contact Rate

The contact rate is the amount of contaminated medium contacted per unit of time or event. Contact rates for subsistence food types (for example, fish for the fish ingestion pathway), are assumed to be 100 percent from the hypothetical assessment area (for example, water body). The following sections describe exposure pathway-specific considerations regarding contact rate.

3.3.2.1 Groundwater and Surface Water Exposure Pathway Parameters

This section describes the following variables specific to the groundwater and surface water exposure pathways: drinking water intake, other contact with groundwater, individual inhalation rate, and individual skin surface area.

Drinking Water Intake

For groundwater and surface water ingestion, the intake rate is assumed to be 2.0 L/day, the average amount of water that an adult ingests (U.S. EPA 1991a). This value, which is currently used to set drinking water standards, is close to the current 90th percentile value for adult drinking water ingestion (2.3 L/day) reported in the *US EPA Exposure Factors Handbook* (EFH) (U.S. EPA 1997b). Also, this value is comparable to the 8 glasses of water per day historically recommended by health authorities (U.S. EPA 1991a).

Other Contact with Groundwater

For the groundwater pathway of dermal exposure during bathing, the contact with water is assumed to occur in the shower. In this analysis, it is assumed that the average adult resident is in contact with groundwater during bathing for 0.25 hour per event and that the average child resident is in contact with groundwater during bathing for 0.33 hour per event, with one event per day (U.S. EPA 1991a, 1997b).

For the groundwater pathway of inhalation exposure during showering, the contact with water is assumed to occur principally in the shower and in the bathroom, although an additional longer exposure to much lower concentrations in the house following shower and bathroom use is also included. In this analysis, it is assumed that the average adult resident spends 11.4 minutes per day in the shower, an additional 48.6 minutes per day in the bathroom, and another 900 minutes (15 hours) in the house after shower and bathroom use (U.S. EPA 1991a, 1997b).

Individual Inhalation Rate

Daily inhalation rates vary depending on activity, gender, age, and so on (U.S. EPA 1997b). Citing a need for additional research, the latest EFH does not recommend a reasonable upper-bound inhalation rate value (U.S. EPA 1997b). The EFH-recommended value for the average inhalation rate is 15.2 m^3/day for males and 11.3 m^3/day for females (U.S. EPA 1997b). The inhalation rate of 20 m^3/day has been commonly used in past U.S. EPA risk assessments (U.S. EPA 1997b). The upper-bound value for an individual's inhalation rate has been established as 0.83 m^3/hr (20 m^3/day) (U.S. EPA 1991a). This value was derived by combining inhalation rates for indoor and outdoor activities in a residential setting (U.S. EPA 1991a). A conservative inhalation rate of 20 m^3/day has been selected for use in this analysis.

Individual Skin Surface Area

For dermal bathing exposure to contaminated groundwater, the selected receptors are an adult and a young child (1 to 6 years old). During bathing, generally all of the skin surface is exposed to water. The total adult body surface area can vary from about 17,000 to 23,000 cm^2. The EFH reports a value of 20,000 cm^2 as the median value for adult skin surface area (U.S. EPA 1997b). A value of 6,900 cm^2 has been commonly used for a child receptor in U.S. EPA risk assessments; this value is approximately the average of the median values for male children aged 2 to 6 years (U.S. EPA 1992b). The EHF presents a range of recommended values for estimates of the skin surface area of children by age (U.S. EPA 1997b). The mean skin surface area at the median for boys and girls 5 to 6 years of age is 0.79 m^2 or 7,900 cm^2 (U.S. EPA 1992b and 1997b). Given that the age for children is defined as 0 to 6 years (see Section 3.3.4), a skin surface area value for ages 5 to 6 years would be a conservative estimate of skin surface area for children. For calculation of dermal exposure to waste constituents, a value of 7,900 cm^2 is assumed for the skin surface area of children, and a value of 20,000 cm^2 is assumed for the skin surface area of adults.

The major factors that must be considered when estimating the intake associated with dermal exposure to groundwater (that is, the absorbed dose per event or DA_{event}) are the constituent concentration in contact with the skin, the surface area of the skin exposed, the duration of exposure, the absorption of the constituent through the skin, and the amount of constituent that can be delivered to a target organ. A detailed discussion of these factors can be found in “Guidelines for Exposure Assessment” (U.S. EPA 1992a) and “Dermal Exposure Assessment: Principles and Applications” (U.S. EPA 1992b). Detailed information concerning the algorithms and parameter values used to calculate DA_{event} is provided in Chapter 4.

3.3.2.2 Air Exposure Pathway Parameters

For air exposure pathways, receptors are assumed to be directly exposed to constituents as a result of normal respiration. Inhalation of vapors and particulate will be influenced by the relative amount of time that a receptor spends indoors. Although vapors entering buildings as a result of air exchange are likely to remain airborne and therefore may be inhaled, particulate entering these buildings are more likely to settle out and not be inhaled. However, for the purposes of this analysis, it is assumed that vapor and particulate may be inhaled throughout the day, both indoors and outdoors. Again, it is assumed that the upper-bound estimate respiration rate for an individual is 0.83 m^3/hr (20 m^3/day) (U.S. EPA 1991a, 1997d).

The U.S. EPA Delisting Program also considers child ingestion of soil contaminated with air-deposited particulate windblown from a nearby landfill. Two different receptors are used for carcinogens and noncarcinogens. For carcinogens, exposure in both childhood and adulthood is considered for the same individual. For noncarcinogens, only exposure in childhood is considered. For carcinogens, exposure is averaged over a lifetime (therefore, the averaging time is 75 years, the average lifetime recommended in the latest EFH). Due to the long-term cumulative effect of carcinogens, exposure to adults is usually assumed. For most routes of exposure (*e.g.*, drinking water ingestion, air inhalation, food ingestion), the intake rate remains relatively proportional to body wight throughout the lifetime. Child intake rates may be slightly higher. After consideration of the weighted average of exposure over a lifetime, little or no effect would be seen by considering child exposure. However, the intake rate of soil is inversely proportional to body weight, because children to 6 years of age are considered more likely to ingest greater quantities of soil than adults and their body weight is a fraction of an adult. Exposure to contaminants via soil ingestion is calculated as a weighted average of exposure for 6 years as a child and as an adult. The total exposure for carcinogenic risk is still averaged over a lifetime. For noncarcinogens, the daily intake is back-calculated by direct comparison to the reference dose (RfD). The RfD being the dose to which a person may be exposed daily over a lifetime or significant portion of a lifetime without adverse effect. Because the RfD is a exposure dose, and because the daily exposure dose from soil ingestion is likely to be considerably higher for children than adults, the daily intake (and therefore the delisting level) is calculated for children. The mean soil ingestion values for children range from 39 to 271 mg/day, with an average of 146 mg/day for soil ingestion and 191 mg/day for soil and dust ingestion (U.S. EPA 1989c, 1997b).

This group of mean values is consistent with the 200-mg/day value that U.S. EPA programs have used as a conservative mean estimate. However, considering pica behavior (deliberate ingestion of soil by children), the upper percentile recommended value for soil ingestion by children is 400 mg/day (U.S. EPA 1997b). Because the prevalence of pica behavior is not known, and because the EFH states that 200 mg/day may be used as a conservative estimate of the mean, a delisting risk assessment uses 200 mg/day as the soil ingestion rate for children.

3.3.3 Exposure Frequency

An exposure frequency of 350 days per year is applied to all exposure scenarios (U.S. EPA 1991a, 1997d). Until better data become available, the common assumption that residents take 2 weeks of vacation per year is used to support a value of 15 days per year spent away from home (leaving 350 days per year spent at home) (U.S. EPA 1991a).

3.3.4 Exposure Duration

The exposure duration reflects the length of time that an exposed individual may be expected to reside near the constituent source. For the adult resident, this value is taken to be 30 years, and for the child resident, this value is taken to be 6 years (U.S. EPA 1997d). The adult resident is assumed to live in one house for 30 years, the approximate average of the 90th percentile residence times from two key population mobility studies (U.S. EPA 1997b). For the child resident, the exposure duration is assumed to be 6 years, the maximum age of the young child receptor (U.S. EPA 1997d). For carcinogens, exposures are combined for children (6 years) and adults (24 years).

3.3.5 Averaging Time

For noncarcinogenic constituents, the averaging time (AT) equals the exposure duration in years multiplied by 365 days per year. For an adult receptor, the exposure duration is 30 years, and for a child receptor, the exposure duration is 6 years (U.S. EPA 1991a). For carcinogenic constituents, the AT has typically been 25,550 days, based on a lifetime exposure of 70 years at 365 days per year (U.S. EPA 1991a). The life expectancy value in the 1997 EFH is 75 years (U.S. EPA 1997b). Given this life expectancy value, the AT for a delisting risk assessment is 27,375 days, based on a lifetime exposure of 75 years at 365 days per year.

3.3.6 Body Weight

The choice of body weight for use in the risk characterization equations presented in Chapter 5 depends on the definition of the receptor at risk, which in turn depends on exposure and susceptibility to adverse effects. “Risk Assessment Guidance for Superfund (RAGS)” (U.S. EPA 1989) defines the body weight of the receptor as either adult weight (70 kg) or child weight (1 to 6 years, 15 kg) on the basis of data presented in “Evaluation and Selection of Models for Estimating Air Emissions from Hazardous Waste Treatment, Storage, and Disposal Facilities” (U.S. EPA 1984). However, most U.S. EPA guidance defines the child (1 to 6 years) weight as 15 kg (U.S. EPA 1991a). The EFH-recommended value of 71.8 kg for an adult differs from the 70-kg value commonly used in U.S. EPA risk assessments (U.S. EPA 1997b). The Integrated Risk Information System (IRIS) uses the 70-kg body weight assumption in deriving cancer slope factors and unit risks. In keeping with the latest EFH recommendation, DRAS uses a 72-kg adult weight and a 15-kg child weight in delisting risk assessments.

The daily intake for an exposure pathway is expressed as the dose rate per body weight. Because children have lower body weights, typical ingestion exposures per body weight, such as those for soil and foods, are higher for children. Therefore, average daily exposure normalized to body weight would be greater for children than adults. However, when factoring in the exposure duration for adults over children, the overall lifetime exposure would be less and the actual risk/hazard would be greater for adults. Use of two body weights may not account for significant differences between the weights of infants and toddlers or the weights of teenagers and adults. However, average body weight, not actual chronological age, defines a child in this context; obviously, the weight of a child changes significantly over the first 6 years of life. The average weight used is assumed to be a realistic average estimate for all 6 years that overestimates the weight of the child for the early years and then underestimates it for the later years (U.S. EPA 1997b).

3.3.7 Fish Consumption

Factors that affect human exposure via ingestion of fish from a surface water body include (1) sediment and water waste constituent concentrations and (2) the percentage of dietary fish caught in the surface water body affected by the waste management unit. The amount of fish consumed also affects exposure because people who eat large amounts of fish will tend to have higher exposures. Fish consumption rates vary greatly, depending on geographic region and social or cultural factors. Because 100 percent of a receptor's dietary fish may not originate from the surface water body near the waste management unit, the percentage of locally caught dietary fish is also a variable that must be considered in estimating exposure.

The EFH-recommended value for fish consumption for all fish is 0.28 g fish/kg body weight/day for an average adult (72 kg) (U.S. EPA 1997b). This value equates with a fish consumption rate of 20.1 g/day for all fish. For delisting risk assessments, U.S. EPA Region 6 estimates that an exposed individual eats 20 g of fish per day, representing one 8-ounce serving of fish approximately once every 11 days.

3.3.8 Age-adjusted Contact Rates

Because contact rates may be different for children and adults, carcinogenic risks during the first 30 years of life were calculated using age-adjusted factors (adj). Use of age-adjusted factors is especially important for soil ingestion exposures, which are higher during childhood and decrease with age. However, for the purposes of combining exposures across pathways, additional age-adjusted factors are used for inhalation and dermal exposures. These factors approximate the integrated exposure from birth until age 30 and combine contact rates, body weights, and exposure durations for two age groups: small children and adults. Age-adjusted factors are either obtained from RAGS Part B or developed by analogy.

For soils, noncarcinogenic waste constituents are evaluated for children separately from adults; no age-adjusted factor is used in this case. The focus on children is considered to be protective because of the higher daily rate of soil intake by children and their lower body weight.

Equation 3-2, 3-3, and 3-4 are used to develop age-adjustment factors for soil ingestion, inhalation, and water ingestion, respectively.

Soil ingestion ([mg·yr]/[kg·d]):

$$IFS_{adj} = \frac{ED_c \cdot IRS_c}{BW_c} + \frac{(ED_r - ED_c) \cdot IRS_a}{BW_a} \tag{3-2}$$

Inhalation ([m^3·yr]/[kg·days]):

$$IFA_{adj} = \frac{ED_c \cdot IRA_c}{BW_c} + \frac{(ED_r - ED_c) \cdot IRA_a}{BW_a} \tag{3-3}$$

Water ingestion ([L·yr]/[kg·d]):

$$IFW_{adj} = \frac{ED_c \cdot IRW_c}{BW_c} + \frac{(ED_r - ED_c) \cdot IRW_a}{BW_a} \tag{3-4}$$

where:

			Default
IFS_{adj}	=	Ingestion factor, soils ([mg·yr]/[kg·days])	113 (U.S. EPA 1991a)
ED_c	=	Exposure duration, child (yrs)	6 (U.S. EPA 1991a)
IRS_c	=	Soil ingestion rate, child (mg/day)	200 (U.S. EPA 1997b)
BW_c	=	Body weight, child (kg)	15 E (U.S. EPA 1997b)
ED_r	=	Exposure duration, residential (yrs)	30 (U.S. EPA 1991a)
IRS_a	=	Soil ingestion rate, adult (mg/day)	100 (U.S. EPA 1997b)
BW_a	=	Body weight, adult (kg)	72 (U.S. EPA 1997b)
IFA_{adj}	=	Inhalation factor ([m^3·yr]/[kg·days])	10.7 (U.S. EPA 1991a)
IRA_c	=	Inhalation rate, child (m^3/day)	10 (U.S. EPA 1989a)

IRA_a = Inhalation rate, adult (m^3/day) 20 (U.S. EPA 1991a)
IFW_{adj} = Ingestion factor, water ([L·yr]/[kg·days]) 1.07 (U.S. EPA 1991a)
IRW_a = Drinking water ingestion rate, adult (L/day) 2 (U.S. EPA 1989a)
IRW_c = Drinking water ingestion rate, child (L/day) 1 (U.S. EPA, 1991a)

Chapter 4
Risk and Hazard Assessment

What's covered in Chapter 4:

- Individual Risk and Hazard Estimation
- Target Risk Levels
    - Target Risk
    - Target Hazard
    - Delisting Levels
- Quantitative Estimation of Risk
- Quantitative Estimation of Potential Noncancer Effects

This chapter explains (1) back-calculation of pathway-specific delisting levels from target risk levels and subsequent selection of the final delisting level for each constituent in a petitioned waste and (2) aggregate risk analysis for one-time delisting petitions. Both determination of delisting levels and aggregate risk analysis involve risk assessment, but the calculations proceed in a different direction for the two cases. In determining delisting levels, the work begins by setting a target risk level, that reflects the maximum acceptable risk. Then, using waste volume and exposure parameters, the delisting level is back-calculated. This is done for each constituent and each potential exposure pathway (see Chapter 3), and the resulting constituent—and pathway-specific delisting levels are used to set the final delisting level for each constituent. It is important to note that even though the calculations proceed from the target risk to the delisting level (which is the reverse of the traditional risk assessment approach), the procedure is still based on standard risk assessment equations and methods — they are simply applied in reverse order.

The back-calculation procedure contrasts with the method used to compute the aggregate risk for a one-time delisting petition. To determine aggregate risk, the calculations proceed in a forward direction. Beginning with the leachate and total waste concentrations for each constituent in the waste (source concentrations), the waste volume and exposure parameters are used to estimate the upper-bound excess lifetime cancer risks (risk) and noncarcinogenic hazards (hazard). The risk is said to be aggregate because risks and hazards are summed separately for receptors (resident adults and children) across all applicable waste constituents and exposure pathways to obtain an estimate of the total individual risk and hazard for each receptor.

Risk is the probability that a receptor will develop cancer. Risk is estimated based on a unique set of exposure, model, and toxicity assumptions. A risk of 1×10^{-6} means that one person out of 1 million persons assumed to be exposed under similar conditions could develop cancer as a result of lifetime exposure to one or more potential carcinogens. Hazard is defined as the potential for noncarcinogenic health effects as a result of exposure to constituents of concern, averaged over an exposure period of less than an entire lifetime. A hazard is not a probability but rather a measure (expressed as a ratio) of the magnitude of a receptor's potential exposure relative to a standard exposure level. The standard exposure level is calculated over an exposure period such that there is no likelihood of adverse health effects to potential receptors, including sensitive populations (U.S. EPA 1989e).

Risks and hazards typically are characterized for a single receptor and are referred to as individual risks and hazards (U.S. EPA 1989a; NC DEHNR 1997). "Risk Assessment Guidance for Superfund (RAGS), Volume I: Human Health Evaluation Manual (Part A)" (U.S. EPA 1989a) states that human health risk assessments in the RCRA program have historically focused on high-end individual risk, and it has been recommended that risk assessors place their primary emphasis on characterizing high-end individual risks. All available guidance on target levels for risks and hazards is developed specifically for estimates of individual risk and hazard (see Sections 4.2 and 4.3). Therefore, the risk assessments performed for delisting using the DRAS will address only individual risks and hazards.

4.1 INDIVIDUAL RISK AND HAZARD ESTIMATION

Aggregate individual risk and hazard estimates are intended to convey information about the risks borne by individuals affected by releases from delisted wastes disposed of in Subtitle D nonhazardous waste management units. Aggregate risk assessments based on the equations described in this chapter provide quantitative risk and hazard estimates associated with potential exposure to constituents in wastes proposed for delisting. Both calculation of delisting levels and evaluation of aggregate risk involve use of risk equations described in the following sections. The cancer slope factors (CSF) for carcinogens and the oral and inhalation RfDs for noncarcinogens used in computing delisting levels and in the aggregate risk assessment are provided in Appendix A. Other chemical-specific variables (for example, bioaccumulation factors, Henry's Law constants, and so on) are also provided in Appendix A.

4.2 TARGET RISK LEVELS

Acceptable or target risk and hazard levels originated with the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (U.S. EPA 1990e). More recently, U.S. EPA has modified the target levels established in the NCP to reflect the contribution of background levels of contamination (U.S. EPA 1994f). However, these modified target values are not discrete indicators of observed adverse effects. If a calculated risk falls within the target values, U.S. EPA may without further investigation conclude that a proposed action (such as delisting a petitioned waste) does not present an unacceptable risk. A calculated risk that exceeds these targets, however, would not in and of itself indicate that the proposed action is not safe or that it presents an unacceptable risk. Rather, a computed risk that exceeds a target value triggers further careful consideration of the underlying scientific basis for the calculation. Target risk and hazard levels are addressed in more detail in the following sections. Sections 4.2.1 through 4.2.5 describe target risk, target hazard, delisting levels, DRAS aggregate risk, and special limitations to the computation of each of these levels.

4.2.1 Target Risk

Risk estimates represent the incremental probability that an individual will develop cancer over a lifetime as a result of exposure to a carcinogenic chemical (U.S. EPA 1989a). As defined in the NCP, U.S. EPA has established the range of generally acceptable risks for known or suspected carcinogens as 1×10^{-6} to 1×10^{-4} (U.S. EPA 1990e). Risks less than 1×10^{-6} (one additional cancer in 1,000,000 individuals) generally are not of concern, whereas risk levels exceeding 1×10^{-4} (one additional cancer in 10,000 individuals) are usually considered unacceptable. A risk level of 1×10^{-6} is used as a point of departure; that is, U.S. EPA generally must determine whether a waste qualifies for delisting if the petitioned waste's risks are between 1×10^{-6} and 1×10^{-4}.

For calculation of delisting levels for multi year (multiple-batch) delistings, the default target risk in the DRAS is set to 1×10^{-5}. At this default target risk, delisting levels for waste constituents represent waste concentrations that pose a potential risk of one additional cancer in 100,000 individuals exposed. The delisting authority may change the target risk in DRAS to match their program requirements. For a one-time delisting, U.S. EPA performs a aggregate risk assessment for the petitioned waste. If the aggregate risk for the petitioned waste exceeds 1×10^{-4}, then the waste may not qualify for delisting. The delisting authority may also set the requirement for an aggregate risk more conservatively.

U. S. EPA Region 6 distinguished between three measures of risk: constituent—and exposure pathway-specific risk, total exposure pathway risk, and aggregate risk in developing the DRAS. Constituent —and pathway-specific risk is calculated as shown in Equation 4-1.

$$Risk_{c,p} = LADD_{c,p} \cdot CSF_c \qquad \textbf{(4-1)}$$

where:

$Risk_{c,p}$ = risk for waste constituent *c* for specific exposure pathway *p*
$LADD_{c,p}$ = lifetime average daily dose for waste constituent *c* (mg/kg-day) via pathway *p*
CSF_c = cancer slope factor for waste constituent *c* $(\text{mg/kg-day})^{-1}$

Equation 4-1 is re-arranged to compute $LADD_{c,p}$ from a given target risk level, when used in calculating delisting levels.

Within a specific exposure pathway, an individual may be exposed to more than one chemical. The total risk associated with exposure to all chemicals through a single exposure pathway, or the total exposure pathway risk, is estimated as shown in Equation 4-2 (U.S. EPA 1989a).

$$Risk_p = \Sigma_c Risk_{c,p} \qquad \textbf{(4-2)}$$

where:

$Risk_p$ = total risk from all constituents for a specific exposure pathway *p*
$Risk_{c,p}$ = risk for waste constituent *c* for specific exposure pathway *p*

At specific exposure points, individual receptors may be exposed via multiple exposure pathways (see Table 2-2 in Chapter 2). Two steps are required to determine whether risks from multiple exposure pathways should be summed for a given individual receptor (U.S. EPA 1989a). The first step is to identify "reasonable exposure pathway combinations." The second step is to determine whether it is likely that the "same individuals would consistently face the reasonable maximum exposure (RME) via more than one pathway." For the purposes of one-time delisting risk assessments, U. S. EPA conservatively assumes that one individual would consistently encounter the RME via more than one pathway. Therefore, risks are summed across the receptor-exposure pathway combinations identified in Table 2-2. Hence, the aggregate risk from multiple exposure pathways is the sum of the total risks for the individual exposure pathways, expressed as shown in Equation 4-3.

$$Risk_{cum} = \Sigma_p Risk_p \qquad \textbf{(4-3)}$$

where:

$Risk_{cum}$ = Aggregate risk from all constituents and all exposure pathways
$Risk_p$ = Total risk from all constituents for specific exposure pathway p

The equations used to calculate dose and risk levels are presented in Section 4.3. Appendix A presents the oral and inhalation CSFs used to compute risks for specific waste constituents. CSFs for constituents not identified in Appendix A-1 can be obtained from the following sources (listed in the preferred order):

1. U.S. EPA's IRIS (U.S. EPA 1996g; accessible online at "http://www.epa.gov/ngispgm3/iris/index.html")
2. U.S. EPA (HEAST) (U.S. EPA 1997f)

If relevant information is not available from these sources, the petitioner should, with the approval of the appropriate regulatory agency, contact the U.S. EPA National Center for Environmental Assessment (NCEA) office in Cincinnati, Ohio. NCEA personnel may be able to assist in developing the necessary toxicity values.

4.2.2 Target Hazard

Standard risk assessment models assume that noncarcinogenic effects, unlike carcinogenic effects, exhibit a threshold; that is, they assume that there is a level of exposure below which no adverse effects will be observed (U.S. EPA 1989a). The potential for noncarcinogenic health effects resulting from exposure to a chemical is assessed by comparing daily exposure estimates to RfDs. An RfD is a daily intake rate that is estimated to pose no appreciable risk of adverse health effects, even to sensitive populations, over a specific exposure duration. Chronic RfDs are used to characterize noncarcinogenic hazards in order to maintain a conservative approach consistent with the purposes of the risk assessment.

Appendix A-1 presents RfDs for a number of chemicals. RfDs for other chemicals can be obtained from the following sources (listed in the preferred order):

1. U.S. EPA's IRIS (U.S. EPA 1996g; accessible on-line at "http://www.epa.gov/ngispgm3/iris/index.html")
2. U.S. EPA (HEAST) (U.S. EPA 1997f)

If relevant information is not available from these sources, the applicant should, with the approval of the appropriate regulatory agency, contact the U.S. NCEA office in Cincinnati, Ohio. NCEA personnel may be able to assist in developing the necessary toxicity values.

The ratio between the daily exposure estimate for a constituent and that constituent's RfD is known as a hazard quotient (HQ). When simultaneous exposure occurs to more than one constituent or through more than one pathway, a hazard index (HI), a summation of HQs, is used to express the potential for harm. The NCP target hazard for systemic toxicants is an *HQ* of 1.0 for a single chemical or an *HI* of 1.0. for multiple chemicals or pathways. U.S. EPA typically considers the need for reducing hazards if an *HQ* or *HI* exceeds 1.0 for human receptors who may reasonably be expected to be exposed. U.S. EPA usually requires remedial action at locations where *HQ* or *HI* values significantly exceed 1.0.

For a one-time delisting, U.S. EPA Region 6 performs a aggregate risk assessment for the petitioned waste. As in risk calculation, a distinction is made between three measures of hazard: constituent—and exposure pathway-specific hazard, total exposure pathway hazard, and aggregate hazard. The ultimate result of a aggregate risk assessment is an *HI* rather than an *HQ* because the assessment includes multiple pathways and may include multiple constituents. Consistent with the NCP and U.S. EPA guidance documents, if the *HI* for a waste exceeds 1.0, U.S. EPA considers whether to allow the waste to qualify for delisting. However, where *HI* values significantly exceed 1.0, U. S. EPA usually reviews the target organ hazard contribution, establishes specific delisting conditions, or denies the delisting petition. Constituent—and pathway-specific hazard is computed as shown in Equation 4-4.

$$HQ_{c,p} = \frac{ADD_{c,p}}{RfD_c} \tag{4-4}$$

where:

$HQ_{c,p}$ = hazard quotient for waste constituent c for exposure pathway p (unitless)
$ADD_{c,p}$ = average daily dose for waste constituent c for pathway p (mg/kg-day)
RfD_c = reference dose for waste constituent c (mg/kg-day)

As with carcinogenic constituents in a specific exposure pathway, a receptor may be exposed to multiple constituents associated with noncarcinogenic health effects. The total noncarcinogenic hazard for each exposure pathway is calculated using the procedures outlined by U.S. EPA (1989a). As in the procedure for calculating risks, the constituent—and pathway-specific hazards are summed for each receptor. The sum represents the total hazard for a given exposure pathway and is calculated as shown in Equation 4-5.

$$HI_p = \Sigma_c \, HQ_{c,p} \tag{4-5}$$

where:

HI_p = total hazard index for all waste constituents for specific exposure pathway p

$HQ_{c,p}$ = hazard quotient of waste constituent c for specific exposure pathway p

This summation methodology assumes that the health effects of the various chemicals to which a receptor is exposed are additive.

As discussed in Section 4.2.1 for risks, a receptor may be exposed to chemicals associated with noncarcinogenic health effects through more than one exposure pathway. For the purposes of a risk assessment, U.S. EPA assumes that it is reasonable to estimate a receptor's total hazard as the sum of the *HI*s for the exposure pathways identified in Table 2-2. Specifically, a receptor's aggregate HI is the sum of the hazards for the individual exposure pathways, expressed as shown in Equation 4-6.

$$HI_{cum} = \Sigma_p HI_p \tag{4-6}$$

where:

HI_{cum} = aggregate hazard index for all constituents and all exposure pathways
HI_p = total hazard index for all constituents for specific exposure pathway p

Generally, when the aggregate HI is assessed, an HI_{cum} value less than or equal to 1.0 is considered to be protective of human health, whereas an HI_{cum} value exceeding 1.0 indicates a potential for noncarcinogenic health effects (U.S. EPA 1989a). The more the HI_{cum} value exceeds 1.0, the greater the level of concern. However, because RfDs do not have equal accuracy or precision, and because they are not based on the same severity of effects, the level of concern does not increase linearly as an Hi_{cum} value approaches and exceeds 1.0 (U.S. EPA 1989a).

The HI_{cum} for a receptor can exceed 1.0 as a result of (1) the presence of one or more chemicals with an *HQ* exceeding 1.0 ($HQ_{c,p} > 1.0$ in Equation 4-5) or (2) summation of several chemical-specific *HQ*s (as in Equation 4-5) or pathway-specific *HI*s (as in Equation 4-6) that are each less than 1.0. In the first case, the presence of at least one chemical-specific hazard greater than 1.0 is interpreted as indicating the potential for noncarcinogenic health effects. In the second case, a detailed analysis is required to determine whether the potential for noncarcinogenic health effects is accurately estimated by HI_{cum} because the toxicological effects associated with exposure to multiple chemicals, often through different exposure pathways, may not be additive (U.S. EPA 1989a); therefore, HI_{cum} may overestimate the potential for noncarcinogenic health effects.

To address this issue, chemical-specific hazards may be summed according to major health effects and target organs or systems (U.S. EPA 1989a). Table A-4 in Appendix A identifies target organs and systems that are affected by listed waste constituents. It is especially important to consider any differences related to the exposure route; this process is referred to as segregation of the *HI*. If the highest segregated *HI* exceeds 1.0, there is a potential for noncarcinogenic health effects. However, if the highest segregated *HI* is less than 1.0, HI_{cum} (as calculated above) is overly conservative, and noncarcinogenic health effects are not likely to result from exposure to waste constituents, even if HI_{cum} is greater than 1.0. Therefore, when HI_{cum} for a specific volume of petitioned waste exceeds 1.0, it does not necessarily mean that the waste cannot be delisted; calculation of segregated *HI*s may show that HI_{cum} is too conservative. Calculation of segregated *HI*s is beyond the scope of this DTSD and should be performed by a qualified toxicologist or risk assessor. Segregated *HIs* are not used in calculating delisting levels or aggregate hazards in subsequent sections.

4.2.3 Delisting Levels

For all standard multi year delisting petitions, the DRAS calculates delisting levels for each constituent of a petitioned waste at the default risk levels (a cancer risk of 1×10^{-5} and a noncancer hazard of 0.1) or the risk levels required by the delisting authority. The waste constituent concentrations of each batch of delisted waste should not exceed the established delisting levels. In determining the allowable concentrations for each constituent of a standard multi year petitioned waste, U.S. EPA Region 6 sets two delisting levels for each constituent: (1) a total concentration delisting level and (2) a leachate concentration delisting level. The total concentration delisting level is the maximum allowable concentration of a constituent in the whole waste. The total concentration of each constituent in the waste should not exceed the total concentration delisting level. The leachate concentration delisting level is the maximum allowable concentration of a constituent in leachate derived from the waste. The TCLP concentration of each waste constituent should not exceed the leachate concentration delisting level. Note that leachate concentrations from leach tests other than the TCLP may also be considered.

The DRAS back-calculates a delisting level for each waste constituent and each exposure pathway that the program addresses. The DRAS generates leachate concentration delisting levels for all the groundwater exposure pathways (ingestion of groundwater, dermal absorption while bathing with groundwater, and inhalation of groundwater volatiles while showering) and for two of the air inhalation exposure pathways (inhalation of volatiles from a landfill and inhalation of volatiles from a surface impoundment). The DRAS generates total concentration delisting levels for the remaining air inhalation exposure pathway (inhalation of particulates from a landfill), the surface water exposure pathways (ingestion of surface water and ingestion of fish), and the soil ingestion exposure pathway.

To develop the final leachate concentration delisting level for a waste constituent, the DRAS calculates the constituent's pathway-specific leachate concentration delisting levels for the groundwater pathways and the two relevant air exposure pathways and then selects the lowest of the pathway-specific values (including delisting levels derived from the SDWA MCL) as the final leachate concentration delisting level. Similarly, to develop the final total concentration delisting level for a waste constituent, the DRAS computes pathway-specific values for the relevant pathways and then chooses the lowest as the final total concentration delisting level.

The DRAS is a risk assessment tool that gives the user the flexibility to select alternative target risk levels. Delisting levels are determined according to the target risk level selected by the user. The selection of delisting levels is a risk management decision and, as such, should consider the selected risk level and other pertinent information.

The delisting levels for each waste constituent are the maximum concentrations (total and TCLP) allowed for the constituent in any batch of the petitioned waste, based on the most sensitive pathway associated with exposure dependent on the TCLP waste constituent concentration and the waste constituent total concentration. Although the DRAS calculates a delisting level (at the target risks) for each of the groundwater pathways, only the most sensitive pathway for each constituent is selected as that constituent's TCLP delisting level. The DRAS goes through the same procedure to determine the total delisting level for each waste constituent. These values are intended to be incorporated into the FR notice for a multi year delisting and are not to be exceeded in any batch sample of the delisted waste. Sections 4.3 and 4.4 present the algorithms used in the DRAS to calculate the delisting level concentrations for multi year delistings. If the concentration of each constituent in a petitioned waste is less than the total concentration delisting level, and if the waste TCLP concentration for each constituent is less than the leachate concentration delisting level, that waste may qualify to exit the hazardous waste management program. No single constituent concentration in the waste or TCLP concentration covered by a multi year delisting petition should exceed its delisting levels.

4.2.4 DRAS Aggregate Risk

In addition to calculating the delisting levels for standard multi year delistings, the DRAS can perform a aggregate risk assessment for one-time disposal of waste in a landfill or surface impoundment waste management unit. The DRAS requires that the user input the total and TCLP concentrations for each constituent of a petitioned waste. The program then computes the aggregate risk and hazard ($Risk_{cum}$ and HI_{cum}) for the waste.

Computing the aggregate risk and hazard for a petitioned waste provides the user with a detailed analysis of the risks and hazards associated with the waste. The DRAS indicates which chemicals, pathways, and receptors are driving the risk and hazard for the waste. The DRAS computes the aggregate risk by summing the risks for all waste constituents for a given exposure pathway and then summing the risks for the pathways analyzed in the delisting risk assessment (see Equations 4-1 through 4-3). The DRAS computes the aggregate hazard by summing the HQs for all waste constituents for a given exposure pathway to obtain exposure pathway-specific HIs and then summing the HIs associated with the exposure pathways analyzed (see Equations 4-4 through 4-6). Sections 4.3 and 4.4 detail the equations and default parameters used in the DRAS to calculate the aggregate risk and hazard, respectively, for a petitioned waste.

If a delisting petition is for a one-time exclusion, the results of the aggregate risk assessment may be used in lieu of the delisting levels. A one-time delisting does not require U.S. EPA to establish monitoring concentrations that must be met by each batch of waste to be managed under a promulgated exclusion. Therefore, the user may bypass the delisting levels, which are set at more conservative risk levels, in favor of the aggregate risk process that is conducted using U.S. EPA's target risk range (1×10^{-4} to 1×10^{-6} for carcinogenic waste constituents and an HI of 1.0 to 0.1 for noncarcinogenic waste constituents). Use of the aggregate risk analysis allows the risk associated with an individual waste constituent to extend to a less conservative risk level as long as the aggregate risk for the entire petitioned waste lies below or within U.S. EPA's target risk range.

4.2.5 Special Limitations and Exceptions

This section summarizes special limitations and exceptions to the methods outlined for calculating delisting levels, and risk and hazard. Earlier regulatory work performed using methods similar to those discussed herein has shown that a waste concentration ceiling may be needed to ensure that risk from direct exposure on site does not achieve unsafe levels, as is discussed in Section 4.2.5.1. Also, as described in Section

4.2.5.2, a special, constituent-specific waste concentration ceiling based on soil saturation levels is needed to prevent delisting of waste that may give rise to nonaqueous-phase liquids (NAPL) in groundwater. Because toxicity factors are not available for lead, its delisting level must be established outside the usual methods of the DRAS (See Section 4.2.5.3). Section 4.2.5.4 describes how risk and hazard levels are computed for PCDDs and PCDFs, and PCBs are addressed in Section 4.2.5.5. Finally other regulatory limits must be considered in the determination of delisting levels, including the Toxicity Characteristic regulatory levels (Section 4.2.5.6), maximum contaminant levels (MCLS) as described in Section 4.2.5.7. In addition, chemical detection limits are addressed in Section 4.2.5.8 and regulatory levels for aquatic biota are addressed in Section 4.2.5.9.

4.2.5.1 Waste Constituent Concentration Ceiling

Using standard chronic risk assessment algorithms, the DRAS generates upper-limit individual waste constituent source concentrations called delisting levels that are protective of the receptor for the defined exposure pathways. However, the delisting levels generated in this manner for certain constituents can be extremely high in their absolute concentrations (approaching those of pure compounds). Under certain circumstances, the generated delisting levels may not be the appropriate waste management concentration limits for the waste management unit. For example, a modeled chronic risk scenario (for example, volatile air emissions) cannot be relied upon if the source concentrations exceed certain limits (see Section 4.2.5.2 soil saturation). Above certain concentrations, the compounds may exist in free phase and exhibit behavior different from that assumed by the model (for example., dissolution and transport of immobile hazardous constituents). Moreover, were wastes allowed to be disposed of at very high concentrations, acute exposures could occur at the waste management unit itself. Some of these high constituent concentrations could interact with and damage liner systems. Finally, the proposed HWIR Media Rule states that "EPA believes it is reasonable to classify media as highly contaminated if 1 percent of the volume of the media is contaminated with a particular constituent" (U.S. EPA 1996b).

For these reasons, there is a need to assess the reasonableness of high concentration limits developed from the chronic risk models, particularly for liquid and volatile organics. The issue is a general policy concern for all chronic risk analyses, and its resolution is beyond the scope of the DRAS or this DTSD. However, the DRAS has been programmed to identify total waste constituents predicted at nominally high concentration levels (10,000 mg/kg or 1 percent by weight or greater). Delisting levels in excess of 10,000 mg/kg are highlighted in red on the Results Screen of the DRAS. When the DRAS computes a delisting level that exceeds 10,000 mg/kg, the petitioned waste containing constituents concentrations above this value will be reviewed on a case-by-case basis. In the DRAS, the delisting levels (exit values) are not altered to reflect this ceiling review. The DRAS provides the user with the calculated delisting level based on the waste constituent concentrations, the waste volume, and the risk level selected by the user.

The HWIR Waste Exemption Rule was proposed (in 60 FR 66344, December 21, 1995) with more than 80 compounds in the non-wastewater leachate (NWT) category, which had criterion limits that exceeded 10,000 mg/kg total waste concentration. As the DTSD methodologies are similar to those of the proposed HWIR, U.S. EPA Region 6 is aware that the chronic risk algorithms in the DRAS will generate delisting levels that exceed 10,000 mg/kg or 1 percent pure compound by weight when the constituent is not particularly toxic or does not readily migrate through the environment due to its physical and chemical properties. U.S. EPA Region 6 believes that delisting levels greater than 10,000 mg/kg, although they are predicted to be protective of chronically exposed resident receptors because they are back-calculated using acceptable risk assessment methodology, may pose a risk involving direct exposure on site. It is conceivable that some waste constituents for which high delisting levels are generated may exhibit another hazardous waste characteristic (such as ignitability) and that the petitioned waste may not be delisted as a result. However, not all waste constituents with elevated delisting levels will exhibit another hazardous waste characteristic and be excluded from delisting. Therefore, U.S. EPA will closely review any petition that includes a waste constituent concentration that exceeds 10,000 mg/kg or 1 percent pure chemical by weight.

4.2.5.2 Soil Saturation Evaluation

A soil saturation value represents the contaminant concentration in soil at which the adsorptive limits of the soil particles and the solubility limits of the available soil moisture have been reached. Above this concentration, pure or free-phase compound is expected in the soil. Three of the input parameters used to calculate soil saturation concentrations are constituent-specific: solubility (Sol), organic carbon partition coefficient (K_{oc}) and Henry’s Law constant (H'). A chemical-specific soil saturation concentration is therefore dependent on the reference value used for each of these parameters and can vary significantly depending on the values selected for these three input parameters. Delisting decisions based on the soil saturation concentrations reported in Appendix A-3 should be made with caution.

The derivation of the soil saturation concentration (C_{sat}) is shown in Equation 4-7.

$$C_{sat} = \frac{Sol}{\rho_b} \left(K_d \cdot \rho_b + \Theta_w + H'\Theta_a \right) \tag{4-7}$$

where:

			Default
C_{sat}	=	soil saturation concentration (mg/kg)	calculated
Sol	=	solubility in water (mg/L-water)	chem-specific (Appendix A-1)
$\tilde{n}_b$	=	dry soil bulk density (kg/L)	1.5 (U.S. EPA 1996f)
K_d	=	soil-water partition coefficient (L/kg)	$K_{oc} \cdot f_{oc}$ (chem-specific)
K_{oc}	=	normalized distribution coefficient (L/kg)	chem-specific (Appendix A-1)
f_{oc}	=	fraction organic carbon content of soil (g/g)	0.006 or site-specific
$\grave{E}_w$	=	water-filled soil porosity (L_{water}/L_{soil})	0.15 (U.S. EPA 1996f)
$\grave{E}_a$	=	air filled soil porosity (L_{air}/L_{soil})	0.28 or n-$\grave{E}_w$ (U.S. EPA 1996f)
n	=	total soil porosity (${}_{Lore}/L_{soil}$)	0.43 or 1 - ($\tilde{n}_b/\tilde{n}_s$)
$\tilde{n}_s$	=	soil particle density (kg/L)	2.65
H	=	Henry's Law constant (atm-m^3/mol)	chem-specific (Appendix A-1)
H'	=	Henry's Law constant (dimensionless)	H x 41, where 41 is a unit conversion factor

Equation 4-7 takes into account the amount of contaminant that is in the vapor phase in the pore spaces of the soil in addition to the amount dissolved in the soil's pore water and sorbed to soil particles. If the soil saturation limit is exceeded, formation of a NAPL plume is possible for the organic constituents. If NAPLs are present, significant contaminant migration may occur within this NAPL phase, which is not accounted for in the EPACMTP fate and transport model. Additionally, compound solubility assumptions for water may be inappropriate if another free-phase liquid or mixture is present. Therefore, EPACMTP modeling results may underestimate potential groundwater concentrations at downgradient receptor wells when waste constituent concentrations exceed soil saturation values.

The DRAS compares soil saturation values with calculated delisting levels and notifies the user when a waste constituent's total delisting level exceeds its soil saturation value. When the DRAS calculates a delisting level greater than the calculated soil saturation, the delisting manager should consider setting the delisting level equal to the soil saturation value in accordance with soil screening guidance (U.S. EPA 1996f).

4.2.5.3 Estimation of Potential Health Effects for Lead

Toxicity factors (for example, RfDs or CSFs) are not available for lead. Therefore, consistent with U.S. EPA (1994f), the U.S. EPA OSW recommends that the potential for adverse health effects associated with exposure to lead be characterized through a direct comparison with medium-specific health-based levels. Specifically, the target level for lead in drinking water is 0.015 mg/L. The drinking water target level is based on the lead action level established under the U.S. EPA primary drinking water regulations. The U.S. EPA OSW recommends target levels at the POE for lead in soil and air of 100 mg/kg and 0.2 (µg)/m^3, respectively.

The soil target level of 100 mg/kg is based on U.S. EPA guidance (1994g) that indicates that soil lead levels less than 400 mg/kg (based on lead concentrations in blood, as discussed below) are not of concern for remediation purposes. The U.S. EPA OSW has incorporated a margin of safety into the risk assessment process by allowing only 25 percent of the remediation threshold lead level to be attributable to releases from a waste management unit. Similarly, the protective air standard of 0.2 $\mu g/m^3$ is based on 25 percent of the National Ambient Air Quality Standard (NAAQS) quarterly average air concentration of 1.5 $\mu g/m^3$ (40 CFR 50.12) converted to an annual basis value (0.9 $\mu g/m^3$).

Estimates of the potential for human health effects associated with potential exposure to lead are typically based on lead concentrations in blood. U.S. EPA guidance (1994g) recommends a maximum lead concentration in blood of 10 µg/deciliter (µg/dL), which is at the low end of the range of concern for adverse health effects in children. "Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes" (U.S. EPA 1994e) presents a mathematical model called the "Integrated Exposure U/BK [IEUBK] Model" for estimating lead levels in the blood of children on the basis of total lead uptake from exposures through diet, drinking water, air, and soil. When run with standard recommended default values (these generally represent national averages or "typical" values), U.S. EPA's IEUBK model predicts that no more than 5 percent of children exposed to a lead concentration in soil of 400 mg/kg will have lead concentrations in their blood exceeding 10 µg/dL (U.S. EPA 1994e).

4.2.5.4 Estimation of Potential Health Effects for PCDDs and PCDFs

PCDDs and PCDFs are toxic compounds that were first discovered as thermal decomposition products of polychlorinated compounds, including (1) the herbicide 2,4,5-T, (2) hexachlorophene, (3) PCBs, (4) pentachlorophenol, and (5) intermediate chemicals used to manufacture these compounds. In recent years, as chemical analytical methods have become more sensitive, additional sources of PCDDs and PCDFs have been identified, including (1) effluent from paper mills that use chlorine bleaches and (2) combustion sources, including forest fires, municipal and medical waste incinerators, and hazardous waste combustion units. "Health Assessment Document for 2,3,7,8-TCDD and Related Compounds" (U.S. EPA 1994b) concludes that there is adequate evidence that exposure to PCDDs and PCDFs results in a broad spectrum of cancer and noncancer effects in animals, some of which may occur in humans. The procedures for estimating risks associated with PCDDs and PCDFs are discussed below.

There are 210 individual forms or "congeners" of PCDDs and PCDFs. U.S. EPA has developed procedures for assessing the cancer risks associated with exposure to the many PCDDs and PCDFs. These procedures are used to assess risk on the basis of the relative toxicity of 2,3,7,8-TCDD, which is generally believed to be the most toxic form (U.S. EPA 1994c). Each congener is assigned a value referred to as a toxicity equivalency factor (TEF) that corresponds to its toxicity in relation to 2,3,7,8-TCDD. For example, 2,3,7,8-TCDD has a TEF of 1.0, and other PCDDs and PCDFs have TEFs of 0.0 to 1.0. The U.S. EPA OSW and U.S. EPA guidance (1993d) recommend that all risk assessments include any PCDD or PCDF congener with chlorine molecules in the 2, 3, 7, and 8 positions. There are a total of 17 possible compounds that generally display dioxin-like toxicity. The TEF values for these 17 compounds are listed below.

Dioxin Congener	TEF (unitless)	Furan Congener	TEF (unitless)
2,3,7,8-TCDD	1.000	2,3,7,8-Tetrachlorodibenzofuran	0.100
1,2,3,7,8-Pentachlorodibenzo(p)dioxin	0.500	1,2,3,7,8-Pentachlorodibenzofuran	0.050
1,2,3,4,7,8-Hexachlorodibenzo(p)dioxin	0.100	2,3,4,7,8-Pentachlorodibenzofuran	0.500
1,2,3,6,7,8-Hexachlorodibenzo(p)dioxin	0.100	1,2,3,4,7,8-Hexachlorodibenzofuran	0.100
1,2,3,7,8,9-Hexachlorodibenzo(p)dioxin	0.100	1,2,3,6,7,8-Hexachlorodibenzofuran	0.100
1,2,3,4,6,7,8-Heptachlorodibenzo(p)dioxin	0.010	1,2,3,7,8,9-Hexachlorodibenzofuran	0.100
1,2,3,4,5,7,8,9-Octachlorodibenzo(p)dioxin	0.001	2,3,4,6,7,8-Hexachlorodibenzofuran	0.100
		1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.010
		1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.010
		1,2,3,4,6,7,8,9-Octachlorodibenzofuran	0.001

Source: U.S. EPA 1994c

Of the 2,3,7,8-substituted PCDD and PCDF congeners, a CSF has been developed for only 2,3,7,8-TCDD. The carcinogenic potential of the other sixteen 2,3,7,8-substituted congeners is derived by assigning TEF values that compare the toxicity of the 16 other toxic congeners to that of 2,3,7,8-TCDD; the CSF for 2,3,7,8-TCDD is based on evidence from experiments with rodents. The TEF values based on receptor binding studies or a sensitive measure of receptor binding,—specifically, induction of aryl hydrocarbon hydroxylase enzyme activity. The TEFs are based on the activity of the compounds in short-term toxicity assays that are considered predictive of the compounds' ability to cause cancer in animals during long-term experimental carcinogenicity studies. Consequently, "CSFs" derived from the TEF values for these 16 toxic congeners are less certain than the CSF for 2,3,7,8-TCDD. Consistent with previous guidance (U.S. EPA 1994c), U.S. EPA Region 6 recommends that a risk assessment for PCDDs and PCDFs be completed by calculating the (TEQ) for 2,3,7,8-TCDD. The total TEQ is calculated by converting the dioxin congener concentrations to a 2,3,7,8-TCDD concentration using the TEFs for all 17 dioxin congeners.

To assess the risk association with a mixture of PCDDs and PCDFs reported in a delisting petition using the TEF procedure, the following steps should be performed:

1. Review the delisting petition for all PCDD and PCDF analytical determinations for the delisting sample.
2. Multiply the congener concentrations in the sample by the TEFs listed above to express the concentrations as 2,3,7,8-TCDD equivalents (TEQs).
3. Sum the products calculated in Step 2 to obtain the total TEQ for the delisting sample.
4. Use the delisting sample's total TEQ as the sample concentration of 2,3,7,8-TCDD to be entered in the DRAS under the chemical name "TCDDioxin 2,3,7,8" or(CAS) number 1746016.

All of the 17 PCDD or PCDF congeners with chlorine molecules in the 2, 3, 7, and 8 positions are highly bioaccummulative compounds. For highly bioaccummulative chemicals, the accumulation of these chemicals up the food chain is an exposure pathway of potential concern. The DRAS does not consider potential impacts from ingestion of production terrestrial food products (fruit/vegetables) or animal food products (beef, pork, chicken). Ingestion of contaminated food products may be a potential exposure pathway for bioaccumulating compounds encountered under continuous exposure conditions. Bioaccummulative compounds may cause adverse impacts via the production terrestrial food products pathway if these compounds continuous long term releases from a landfill are expected to result in concentrations approaching some equilibrium at the point of exposure over a duration of years. Region 6 recommends analysis of the terrestrial food products pathway, in addition to the DRAS analysis, should the user have reason to expect that this compound's above ground releases from a landfill will occur continuously over a period of years to decades. Consult a risk assessor or toxicologist with questions regarding risk via the terrestrial food (fruit/vegetable and animal) products pathways.

4.2.5.5 Estimation of Potential Health Effects for PCBs

U.S. EPA guidance (1988c, 1994d, 1994e) recommends that all PCB congeners (209 different chemicals) be addressed as a mixture having a single carcinogenic potency in a risk assessment. This recommendation was based on the U.S. EPA drinking water criteria for PCBs (U.S. EPA 1988c), which in turn were based on available toxicological information with the following limitations:

- The only PCB for which a CSF had been developed was Aroclor 1260; there was no agreed— upon procedure for applying this CSF to similar mixtures with less chlorine content.
- Available physical, chemical, fate-and-transport, and toxicological information on individual congeners was limited, primarily because separation and synthesis of pure congeners can be technically difficult.
- The number of tests conducted with various PCB mixtures and specific PCB congeners to demonstrate similar toxicological effects was very limited.

4.2.5.5.1 Coplanar PCBs

Since the compilation of "Drinking Water Criteria Document for Polychlorinated Biphenyls (PCBs)" (U.S. EPA 1988c), additional research on PCBs has been reported. The most important result of this research is the finding that some of the moderately chlorinated PCB congeners can have dioxin-like effects (U.S. EPA 1992c; 1994d; ATSDR 1995). These dioxin-like congeners include are listed below.

PCB Congener	TEF (unitless)	PCB Congener	TEF (unitless)
3,3',4,4'-Tetrachlorobiphenyl	0.0005	2,3,3',4,4',5-Hexachlorobiphenyl	0.0005
2,3,3',4,4'-Pentachlorobiphenyl	0.0001	2,3,3',4,4',5'-Hexachlorobiphenyl	0.0005
2,3,4,4',5-Pentachlorobiphenyl	0.0005	2,3',4,4',5,5'-Hexachlorobiphenyl	0.00001
2,3',4,4',5-Pentachlorobiphenyl	0.0001	3,3',4,4',5,5'-Hexachlorobiphenyl	0.01
2',3,4,4',5-Pentachlorobiphenyl	0.0001	2,2',3,3',4,4',5-Hheptachlorobiphenyl	0.0001
3,3',4,4',5-Pentachlorobiphenyl	0.1	2,2',3,4,4',5,5'-Heptachlorobiphenyl	0.00001
		2,3,3',4,4',5,5'-Heptachlorobiphenyl	0.0001

Source: U.S. EPA 1996h

The listed congeners have four or more chlorine atoms with few substitutions in the ortho positions (positions designated as 2, 2', 6, and 6'). They are sometimes referred to as coplanar PCBs because the rings can rotate into the same plane if they are not blocked from rotation by ortho-substituted chlorine atoms; in this configuration, the shape of the PCB molecule is very similar to that of a PCDF molecule. Studies have shown that these dioxin-like congeners can then react with the aryl hydrocarbon receptor; this reaction is believed to initiate the adverse effects of PCDDs and PCDFs. As reported in "PCBs: Cancer Dose-Response

Assessment and Application to Environmental Mixtures" (U.S. EPA 1996h), the World Health Organization has used various test results to derive interim TEFs ranging from 0.1 to 0.00001 for the PCB congeners listed above. Additional congeners are suspected of producing similar reactions, but not enough data are available to derive TEF values for them. U.S. EPA Region 6 recommends that permitting authorities estimate risks associated with coplanar PCBs by computing a TEQ for PCBs and then applying a slope factor for dioxin on the chemical input screen of the DRAS.

4.2.5.5.2 Other PCBs

In addition to the coplanar (dioxin-like) PCB congeners, the remaining PCBs should be evaluated in a risk assessment. Based on consideration of the accumulated research on PCBs, (especially a recent carcinogenesis study of Aroclors 1016, 1242, 1254, and 1260 and a number of studies of the transport and bioaccumulation of various congeners) U.S. EPA derived three new CSFs to replace the former single CSF for PCBs (U. S. EPA 1996h). These new CSFs became effective in IRIS (U.S. EPA 1996g) on October 1, 1996. Additional studies are still being performed on PCBs. Therefore, the three CSFs are subject to revision as additional information becomes available. The three CSFs and the criteria for their use are listed below.

CSF (mg/kg-day)$^{-1}$	**Criteria for Use**
2	Food chain exposure; sediment or soil exposure Early-life (infant and child) exposure via all routes to all PCB mixtures
0.4 (less used)	Ingestion of water-soluble (less chlorinated) congeners Inhalation of evaporated (less chlorinated) congeners
0.07	Congeners with more than four chloride atoms per molecule less than 0.5 percent of the total PCBs

Source: U.S EPA 1996h

A CSF of 2 mg/kg-day^{-1} will be used for a risk assessment. The DRAS will perform the PCB risk analysis using the PCB CSF of 2 mg/kg-day^{-1} to generate a conservative estimate of the risk associated with PCB exposure. However, the CSF of 0.07 mg/kg-day^{-1} can be used in risk assessments where the PCB mixture contains less than 0.5 percent PCB congeners with more than four chlorine atoms. This risk calculation would have to be made by the delisting manager.

All of the coplanar (dioxin-like) PCB congeners are highly bioaccummulative compounds. For highly bioaccummulative chemicals, the accumulation of these chemicals up the food chain is an exposure pathway of potential concern. The DRAS does not consider potential impacts from ingestion of production terrestrial food products (fruit/vegetables) or animal food products (beef, pork, chicken). Ingestion of contaminated food products may be a potential exposure pathway for bioaccumulating compounds encountered under continuous exposure conditions. Bioaccummulative compounds may cause adverse impacts via the production terrestrial food products pathway if these compounds continuous long term releases from a landfill are expected to result in concentrations approaching some equilibrium at the point of exposure over a duration of years. Region 6 recommends analysis of the terrestrial food products pathway, in addition to the DRAS analysis, should the user have reason to expect that this compound's above ground releases from a landfill will occur continuously over a period of years to decades. Consult a risk assessor or toxicologist with questions regarding risk via the terrestrial food (fruit/vegetable and animal) products pathways.

4.2.5.6 Toxicity Characteristic Rule Evaluation

The TC Rule was published on March 29, 1990, in 55 FR 11798. The TC Rule established regulatory levels for 40 nonhydrolyzing or minimally hydrolyzing constituents (see Table 4-1). A solid waste exhibits the characteristic of toxicity if, the (TCLP), extract from a representative sample of the waste contains a concentration equal to or greater than the concentration listed in 40 CFR 261.24(a) for any of the waste constituents listed. The regulatory levels were generated based on a health-based chronic toxicity limit and a DAF of 100. The health-based chronic toxicity reference levels for the toxicity characteristic constituents were generated using chronic RfDs, carcinogenic risk-specific doses, or MCLs.

The TC rule regulatory levels are not to be exceeded. Because the DRAS generates levels using different modeling and exposure scenarios, delisting levels for the 40 constituents may be below the regulatory levels. Therefore, the DRAS compares a waste constituent's leachate concentration delisting level concentration, called the maximum allowable TCLP concentration, with the waste constituent's TC Rule regulatory level. If the maximum allowable (see p. 4-23) TCLP concentration exceeds the TC Rule regulatory level, the DRAS notifies the user of the exceedance.

TABLE 4-1

TC RULE REGULATORY LEVELS

Chemical	Regulatory Level (mg/L)
Arsenic	5.0
Barium	100
Benzene	0.5
Cadmium	1.0
Carbon Tetrachloride	0.5
Chlordane	0.03
Chlorobenzene	100
Chloroform	6.0
Chromium	5.0
Cresol	200
m-Cresol	200
o-Cresol	200
p-Cresol	200
2,4-D	10.0
1,4-Dichlorobenzene	7.5
1,2-Dichloroethane	0.5
1,1-Dichloroethylene	0.7
2,4-Dinitrotoluene	0.13
Endrin	0.02
Heptachlor	0.008
Hexachlorobenzene	0.13
Hexachloro-1,3-butadiene	0.5
Hexachloroethane	3.0
Lead	5.0
Lindane	0.4
Mercury	0.2
Methoxychlor	10.0
Methyl ethyl ketone	200.0
Nitorbenzene	2.0
Pentachlorophenol	100.0
Pyridine	5.0
Selenium	1.0
Silver	5.0
Tetrachloroethylene	0.7
Toxaphene	0.5
Trichloroethylene	0.5
2,4,5-Trichlorophenol	400
2,4,6-Trichlorophenol	2.0
2,4,5-TP Acid (Silvex)	1.0
Vinyl chloride	0.2

Source: U.S. EPA 1990d

4.2.5.7 MCL Evaluation

In 1974, Congress passed the Safe Drinking Water Act. This law requires U. S. EPA to determine safe levels of chemicals in drinking water that do or may cause health problems. These non-enforceable levels, which are based solely on possible health risks and exposures, are called MCL goals (MCLGs). Based on MCLGs, U. S. EPA sets enforceable standards called MCLs (see Appendix A-2). MCLs are set as close to the MCLGs as possible, considering the ability of public water systems to detect contaminants and remove them using suitable treatment technologies.

In addition to calculating a waste constituent's delisting level the ingestion of groundwater based on the preselected target risk level, the DRAS calculates the waste constituent's delisting level for ingestion of groundwater where the groundwater concentration is set at the existing MCL. In some cases, the waste constituent's groundwater ingestion delisting level based on the MCL will be less conservative than that calculated based on the selected risk level.

When the DRAS indicates that groundwater ingestion is the limiting groundwater pathway, the user has the option of considering either of the groundwater ingestion pathway delisting levels: the risk-based maximum acceptable TCLP concentration or the MCL-based maximum allowable TCLP concentration. By default, however, the DRAS will select the more conservative of the two in summarizing delisting levels.

4.2.5.8 Detection Limit Evaluation

Often a delisting petition contains waste sample analytical results that for a number of the constituents analyzed for but not detected. A nondetect may be reported at the method detection limit (MDL), an estimated quantitation limit (EQL), or a reporting level (RL). A number of variables influence a detection limit including the instrumentation used, the waste matrix, and the analytical test method. In some instances, the reported detection limit for a given chemical may exceed an allowable delisting level at the target risk level. If the user suspects the analyte to be a potential waste constituent, the user can employ DRAS to determine whether the reported detection limit for a waste constituent exceeds a delisting level at the target risk level.

The user enters a detection level concentration as the concentration in DRAS and selects a check box identifying the concentration as a detection limit. The DRAS determines the risk and hazard associated with a fraction of the waste constituent value. The default fraction used in the risk assessment is 0.5 (50 percent of the detection limit). The program tracks waste constituent risks associated with actual measured concentrations and risks associated with 50 percent of the entered detection limit values. The risk assessment results for a value entered as a waste constituent's detection limit are presented in italics to distinguish them from risk assessment results for measured waste constituent concentrations.

4.2.5.9 Estimation of Ecological Effects

A simple framework was developed for ecological assessment of wastes petitioned for delisting based on EPA's Framework for Ecological Risk Assessment (U.S. EPA, 1992d). During the problem formulation phase, a suite of ecological exposure scenarios were reviewed to determine potential ecological exposure pathways. At best, one can only infer that an ecosystem is protected from chemical stressors. The toxicological data support the evaluation of individuals, populations, and occasionally communities, but are inadequate to address the complexities of an ecosystem in most cases. Thus, the approach taken in the assessment was to estimate protective levels for the populations and communities expected to be exposed to wastes disposed to a landfill or surface impoundment, the WMUs evaluated in the DRAS program.

Ecological Exposure

The ecological risk assessment is based upon an approach that evaluates the movement of waste constituents from their WMUs, through different pathways, to the points where ecological receptors are expected to be exposed to these constituents. In selecting the ecological exposure pathways, U.S. EPA Region 6 considered the evaluated WMUs, the landfill and surface impoundment, and the potential for ecological exposure at these managed non-Subtitle C units. Potential exposure to two generic ecosystems was reviewed: A freshwater-based ecosystem and a terrestrial-based ecosystem. Terrestrial-based exposure from wastes disposed to landfills is minimized by regulations. Under 40 CFR §258, minimum criteria are established for all municipal solid waste landfills. These criteria, including wetland and floodplain controls, disease vector control and cover material requirements minimize terrestrial exposure. In Region 6, state regulations impose these criteria on industrial solid waste landfills. Given the measures employed to control terrestrial exposure, Region 6 believes terrestrial exposures to be minimal. Other regions should consult their respective states for regulations imposing criteria on industrial landfills. Nevertheless, Region 6 considered the potential for exposure to the freshwater ecosystem from wastewater runoff from a landfill receiving delisted wastes.

Although most RCRA delistings involve solid materials, liquid wastes may on occasion be delisted to surface impoundments. There are no federal regulations establishing criteria for municipal or industrial surface impoundments. The DRAS does not consider the potential for exposure to the freshwater ecosystem from wastewater runoff from a surface impoundment receiving delisted wastes. In the case of a delisting to a surface impoundment, Region 6 encourages the delisting manager to consider conducting a waste specific ecological risk assessment. The risk assessment should consider potential ecological pathways such as acute exposure to avian species.

Ecological Assessment Endpoint

The "Framework for Ecological Risk Assessment" defines assessment endpoints as explicit expressions of the environmental value that is to be protected (U.S. EPA 1992d). The Framework document goes on to define measurement endpoints as the measurable attributes related to the environmental value chosen as the assessment endpoint (U.S. EPA 1992d). Although it is conceptually useful to distinguish between the two types of endpoints, the practical implications in a predictive regulatory assessment require that both types be evaluated concurrently. In other words, assessment endpoints are not useful if they do not have measurable attributes, and measurement endpoints are not useful for assessment endpoints lacking ecological significance. For ecological receptors, an environmental value for the protection of populations or communities was selected as an assessment endpoint over a value to protect a given species.

Ecological Benchmarks

Ecological benchmarks were developed for the protection of the aquatic community. These ecological benchmarks are referred to as toxicity reference values (TRVs) and were developed from a variety of ecological receptors based on the availability of data for a given waste constituent. The TRV is developed to protect the entire community, not one particular species. In general, TRVs (the measurement endpoints) were selected for consistency with the Agency's "Framework for Ecological Risk Assessment" (U.S. EPA 1992d). Region 6 believes the ecological analysis is conservative with respect to the overall assessment endpoint (e.g., sustainability of the reproducing populations) because of the way the source, fate and transport parameters are set and how the TRVs are developed. However, the degree to which this conservativeness transfers to ecosystems is not known. See Appendix A-1 for the list of aquatic TRVs currently used in the DRAS analyses.

The ecological assessment focused on inferring the sustainability of populations and communities within the aquatic ecosystem. Therefore, TRVs were derived from measurement endpoints (i.e., reproductive, developmental, growth, survival, and mortality) from which such inferences could be made. Reproductive studies (e.g., number of viable young per female) were preferred over other endpoints. The aquatic TRVs defaulted to a more conservative no effects level (or concentration) approach for ecological receptors. For populations of fish and aquatic invertebrates (represented by daphnids), a hierarchical approach was taken for use of data sources in deriving aquatic TRVs. The first choice was the final chronic values (FCVs) from the Ambient Water Quality Criteria (AWQC) effort by the EPA Office of Water (U.S. EPA 1998d). If these benchmarks were not available, then a freshwater aquatic TRV was selected from the draft *Protocol for Screening Level Ecological risk Assessment at Hazardous Waste Combustion Facilities* (U.S. EPA. 1998e). Finally, TRVs were selected from the *Toxicological Benchmarks for Screening Potential contaminants of Concern for Effects on Aquatic Biota* (Suter and Tsao, 1996).

Ecological Example Calculation

The following outlines an example calculation for determining whether the waste constituent concentration is predicted to exceed the protective concentration of the constituent in the aquatic community at the point of exposure. The aquatic toxicity reference value (TRV) is the constituent concentration in the ecosystem that should not be exceeded to protect the aquatic community. The allowable ecological delisting level concentration is the delisting waste constituent concentration that will not exceed the aquatic TRV and is specific to the waste volume/chemical/waste management unit/receptor.

Receptor:	Aquatic ecosystem
Pathway:	Waste Constituent Concentration → WMU → Runoff → Surface water → Aquatic TRV
WMU:	Landfill

Equations 2-48 through 2-54 are used to calculate the predicted surface water concentration for a given waste constituent delisted to a landfill, based on the waste constituent's concentration and volume reported in the petition. The overland to surface water model used in this pathway encompasses both the WMU release and the fate and transport portions of the calculation. The DRAS compares the waste constituent's predicted surface water concentration to the constituent's aquatic TRV. If the predicted surface water concentration exceeds the aquatic TRV, the petitioned waste has exceeded the ecological assessment. The DRAS notifies the user of any waste constituent whose total concentration is predicted to exceed the aquatic TRV in the modeled aquatic ecosystem.

The DRAS compares the predicted ambient concentration with the TRV for protection of the ambient water community. For highly bioaccummulative waste constituents (log $K_{ow} > 4.0$ or $BCF > 100$), such as PCBs or dioxin, the user should consider that bioaccumulation of the waste constituent through the food web may occur. Under these circumstances, the ambient TRV value may not be protective. The current version of the DRAS does not account for bioaccumulation of waste constituents in the aquatic community. When assessing ecological impacts from the runoff of waste constituents having a log K_{ow} > 4.0 or a BCF > 100, the user should consider the potential for bioaccummulative impacts within the food web or benthic community.

4.3 QUANTITATIVE ESTIMATION OF RISK

This section presents the equations used to calculate pathway delisting levels and risks associated with exposures to carcinogenic chemicals constituents via groundwater, surface water, air, and soil pathways. Calculation of delisting levels and hazards associated with exposures to noncarcinogenic chemicals is addressed in Section 4.4. The equations used are adapted from the Technical Support Document HWIR (U.S. EPA 1995b) and U.S. EPA OSW delisting dockets (U.S. EPA 1993a, and 1994a). The general risk equations presented in Section 4.2.1 are implemented for specific constituents and exposure pathways. $Risk_{c,p}$ is first computed for each constituent using an expanded form of Equation 4-1, and is summed across all constituents to calculate $Risk_p$ for each exposure pathway as shown in Equation 4-2. Risks for all groundwater exposure pathways and surface exposure pathways are summed separately by the DRAS to assist the user in understanding which pathways contribute most to risk. The DRAS finally sums groundwater exposure pathway risk and surface exposure pathway risk as shown in Equation 4-3 to provide the user with the aggregate risk, $Risk_{cum}$, for all exposure pathways. Constituent- and pathway-specific subscripts are omitted in subsequent equations for simplicity of presentation; the context indicates whether the value being calculated is constituent- or pathway-specific.

For groundwater exposures, the DRAS computes constituent-specific delisting levels and aggregate risks for the following groundwater pathways: ingestion of groundwater, dermal exposure to groundwater via bathing (adult and child), and inhalation of volatile constituents released from groundwater during showering. The equations used to compute the risks associated with groundwater exposures are discussed in Section 4.3.1. For surface water, the DRAS computes delisting levels and aggregate risks for the following exposure pathways: ingestion of drinking water and ingestion of fish, both from surface water bodies receiving runoff from landfills. The equations used to calculate risks from exposure to surface water are discussed in Section 4.3.2. For air, the DRAS computes delisting levels and aggregate risks associated with exposure via inhalation of particulates and volatiles from solid wastes disposed of in landfills and inhalation of volatiles from liquid wastes disposed of in surface impoundments. Finally, the DRAS computes delisting levels and aggregate risks associated with exposure via (child) ingestion of soil contaminated by particles eroded from a landfill by wind and deposited on the soil from the air. The equations used to calculate risks associated with the inhalation exposure pathways and with soil ingestion are discussed in Sections 4.3.3 and 4.3.4, respectively.

4.3.1 Groundwater Exposure Pathways

Three groundwater exposure pathways are evaluated in the DRAS: ingestion of groundwater, dermal exposure to groundwater via bathing, and inhalation of volatile constituents released from groundwater during showering. The receptor is assumed to be an adult for the ingestion and showering pathways; both adult and child receptors are addressed for the bathing pathway. Each pathway is discussed below.

4.3.1.1 Ingestion of Groundwater

This section discusses computation of leachate concentration delisting levels and of risks associated with exposure of adults to carcinogenic waste constituents via the groundwater ingestion pathway. Section 4.3.1.1.1 presents the equations used to compute pathway delisting levels for standard multi year delistings. Section 4.3.1.1.2 presents the equations and methods used to compute constituent-specific risk and pathway-specific risk.

4.3.1.1.1 Delisting Level Analysis

A pathway leachate concentration delisting level for groundwater ingestion ($C_{dl\text{-}ingest}$) is calculated for each carcinogenic constituent using the maximum allowable constituent concentration in groundwater for ingestion exposure ($C_{gw\text{-}ingest}$). $C_{gw\text{-}ingest}$ is computed at the target risk level using Equation 4-8.

$$C_{gw-ingest} = \frac{TR \cdot AT \cdot 365\ days/yr}{EF \cdot IFW_{adj} \cdot CSF_{oral}} \tag{4-8}$$

where:

			Default
$C_{gw\text{-}ingest}$	=	maximum allowable constituent concentration in groundwater for ingestion (mg/L)	calculated
TR	=	individual target risk level (unitless)	User specified (default 1 x 10^{-5} See Section 4.2.3)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)

AT	=	averaging time (years)	75 (U.S. EPA 1997b)
IFW_{adj}	=	water ingestion factor, age-adjusted [L•year]/[kg•day]	1.07 (Equation 3-4)
CSF_{oral}	=	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)

The waste constituent's $C_{gw\text{-}ingest}$ is multiplied by its volume-adjusted DAF to generate the constituent's $C_{dl\text{-}ingest}$, as shown in Equation 4-9. Note that since some inorganic constituents now have DAFs that vary based on the landfill leachate concentration, DRAS now uses an iterative procedure to determine the leachate concentration-volume-adjusted DAF pairing that corresponds to the maximum allowable receptor well concentration. The DRAS may in some cases extrapolate between known leachate-DAF pairings in order to determine the appropriate delisting level.

$$C_{dl-ingest} = DAF_{va} \cdot C_{gw-ingest} \qquad \textbf{(4-9)}$$

where:

			Default
$C_{dl\text{-}ingest}$	=	pathway leachate concentration delisting level for groundwater ingestion (mg/L)	calculated
DAF_{va}	=	waste volume-adjusted dilution-attenuation factor (unitless)	Equation 2-5
$C_{gw\text{-}ingest}$	=	maximum allowable constituent concentration in groundwater for ingestion (mg/L)	Equation 4-8

If the TCLP concentration for a given waste constituent exceeds its pathway leachate concentration delisting level, $C_{dl\text{-}ingest}$, the waste may not qualify for a standard multi year delisting. The waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented Section 4.3.1.1. 2.

4.3.1.1.2 Risk Analysis

The basic risk equation used to develop Equations 4-8 and 4-9, in which the target risk is specified and the leachate concentration delisting level is back-calculated, is solved in the forward direction to compute the risk from groundwater ingestion for each carcinogenic constituent, given a concentration of that constituent in groundwater, as shown in Equation 4-10.

$$Risk = \frac{C_{gw} \cdot IFW_{adj} \cdot EF \cdot CSF_{oral}}{AT \cdot 365 \ days/year} \qquad \textbf{(4-10)}$$

where:

			Default
Risk	=	cancer risk for carcinogens (unitless)	calculated

C_{gw}	=	constituent concentration in groundwater (mg/L)	Equation 2-6
IFW_{adj}	=	water ingestion factor, age-adjusted [L•year]/[kg•day]	1.07 (Equation 3-4)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
CSF_{oral}	=	constituent oral cancer slope factor $(\text{mg/kg-day})^{-1}$	chem-specific (Appendix A-1)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)

To compute the individual constituent risks for groundwater ingestion, the user enters each waste constituent TCLP concentration into the DRAS at the specified prompt. The DRAS computes a waste constituent's C_{gw} by dividing the waste constituent TCLP concentration by the constituent's DAF_{va} (see Equation 2-6). Using Equation 4-10, the DRAS calculates the groundwater ingestion- specific risk for each constituent. Then the DRAS uses Equation 4-2 and sums the individual constituent risks to determine the total risk for the groundwater ingestion pathway.

4.3.1.2 Adult and Child Dermal Contact with Groundwater During Bathing

This section describes computation of leachate concentration delisting levels and risks associated with exposure of adults and children to carcinogenic waste constituents via dermal contact with groundwater during bathing. Section 4.3.1.2.1 presents the equations used to compute pathway delisting levels for standard multi year delisting. Section 4.3.1.2.2 presents the equations used to compute constituent-specific risk and pathway-specific risk for one-time delistings.

4.3.1.2.1 Delisting Level Analysis

Pathway leachate concentration delisting levels for dermal contact with groundwater used for bathing ($C_{dl\text{-}dermal}$) are calculated using the target risk level. The DRAS computes $C_{dl\text{-}dermal}$ for the adult and child receptors using appropriate exposure parameters for each and then sets the final $C_{dl\text{-}dermal}$ using the results for the more sensitive of the two receptors. $C_{dl\text{-}dermal}$ is calculated for each carcinogenic constituent and is a function of the maximum allowable constituent concentration in groundwater for dermal exposure, $C_{gw\text{-}dermal}$, which is computed so as not to exceed the acceptable risk level for dermal exposure during bathing. $C_{gw\text{-}dermal}$ is multiplied by the constituent's DAF_{va} to generate $C_{dl\text{-}dermal}$, as shown in Equation 4-11.

$$C_{dl-dermal} = DAF_{va} \cdot C_{gw-dermal} \tag{4-11}$$

where: Default

$C_{dl\text{-}dermal}$ = pathway leachate concentration delisting level for calculated

		groundwater dermal contact (mg/L)	
DAF_{va}	=	waste volume-adjusted dilution attenuation factor (unitless)	Equation 2-5
$C_{gw\text{-}dermal}$	=	maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)	Equations 4-12 through 4-14

$C_{gw\text{-}dermal}$ is computed differently for organic and inorganic constituents. For organic constituents, $C_{gw\text{-}dermal}$ is calculated using one of two methods, depending on whether the elapsed time from the beginning of bathing (called the event duration, t_{event}) is greater or less than the time required for the flux of chemical through the skin to reach steady state (a chemical-specific constant called t^*). For inorganic constituents, the method used is independent of t_{event}.

Equation 4-12 is used to compute $C_{gw\text{-}dermal}$ for an organic constituent when t_{event} is less than t^*.

$$C_{gw-dermal} = \frac{DA_{event} \cdot 10^3 cm^3/L}{2K_p^{\,w}\sqrt{\dfrac{6\tau t_{event}}{\pi}}} \tag{4- 12}$$

where:

			Default
$C_{gw\text{-}dermal}$	=	maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)	calculated
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	Equation 4-15
$K_p^{\,w}$	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
ô	=	lag time (hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25(adult) or 0.33 (child) (U.S. EPA 1997b)

$C_{gw\text{-}dermal}$ is computed for organic constituents using Equation 4-13 when t_{event} is greater than or equal to t^*.

$$C_{gw-dermal} = \frac{DA_{event} \cdot 10^3 cm^3/L}{K_p^{\,w}\left[\dfrac{t_{event}}{1+B} + 2\tau\left(\dfrac{1+3B+(3B)^2}{1+B^2}\right)\right]} \tag{4-13}$$

where:

			Default
$C_{gw\text{-}dermal}$	=	maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)	calculated
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	Equation 4-15

K_p^w	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25 (adult) or 0.33 (child) (U.S. EPA 1997b)
ô	=	lag time (hr)	chem-specific (Appendix A-1)
B	=	Bunge constant (unitless)	chem-specific (Appendix A-1)

$C_{gw\text{-}dermal}$ is computed for inorganic constituents using Equation 4-14.

$$C_{gw-dermal} = \frac{DA_{event} \cdot 10^3 cm^3/L}{K_p^w \cdot t_{event}} \qquad \textbf{(4-14)}$$

where:

			Default
$C_{gw\text{-}dermal}$	=	maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)	calculated
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	Equation 4-15
K_p^w	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25(adult) or 0.33 (child) (U.S. EPA 1997b)

To generate $C_{gw\text{-}dermal}$, one must first generate the dose absorbed per unit area per event (DA_{event}.). DA_{event} is calculated separately for adults and children using Equation 4-15.

$$DA_{event} = \frac{AT \cdot DAD \cdot BW \cdot 365 \ days/year}{EV \cdot EF \cdot ED \cdot A_{skin}} \qquad \textbf{(4-15)}$$

where:

			Default
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	Calculated
AT	=	averaging time (years)	75 (U.S. EPA 1997b)
DAD	=	dermally absorbed dose (mg/kg-day)	Equation 4-16
BW	=	body weight (kg)	
		- adult	72 (U.S. EPA 1997b)
		- child	15 (U.S. EPA 1997b)
EV	=	event frequency (events/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	
		- adult	30 (U.S. EPA 1997b)
		- child	6 (U.S. EPA 1997b)

A_{skin}	=	exposed skin surface area (cm^2)	
		- adult	20,000 (U.S. EPA 1997b)
		- child	7,900 (U.S. EPA 1997b)

The *DAD*, or dermally absorbed dose for contact with water during bathing, is computed for carcinogens using Equation 4-16.

$$DAD = \frac{TR}{CSF_{oral}} \qquad \textbf{(4-16)}$$

where:

			Default
DAD	=	dermally absorbed dose (mg/kg-day)	calculated
TR	=	individual target risk level (unitless)	user specified (1×10^{-5} default)
CSF_{oral}	=	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)

If the TCLP concentration for a given waste constituent exceeds its pathway leachate concentration delisting level, $C_{dl\text{-}dermal}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk steps calculations presented Section 4.3.1.2.2..

4.3.1.2.2 Risk Analysis

In the risk analysis, the DRAS uses the equations used in computation of the delisting levels, but instead of back-calculating from the target risk level to a maximum allowable constituent concentration in leachate, the equations are used to calculate the risk given some constituent concentration expected in the groundwater. The risk for a given carcinogenic constituent associated with dermal contact during bathing is computed for the adult and child receptors using Equation 4-17.

$$R = DAD \cdot CSF_{oral} \qquad \textbf{(4-17)}$$

where:

			Default
R	=	risk from dermal contact during bathing for constituent (unitless)	calculated
DAD	=	dermally absorbed dose (mg/kg-day)	Equation 4-18
CSF_{oral}	=	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)

The dermally absorbed dose (*DAD*), needed in Equation 4-17, is calculated as shown in Equation 4-18 (U.S. EPA 1992b).

$$DAD = \frac{DA_{event} \cdot EV \cdot EF \cdot ED \cdot A_{skin}}{AT \cdot 365 days/year \cdot BW} \qquad \textbf{(4-18)}$$

where:

			Default
DAD	=	dermally absorbed dose (mg/kg-day)	calculated
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	Equations 4-19 through 4-21
EV	=	event frequency (events/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	
		- adult	30 (U.S. EPA 1997b)
		- child	6 (U.S. EPA 1997b)
A_{skin}	=	exposed skin surface area (cm^2)	
		- adult	20,000 (U.S. EPA 1997b)
		- child	7,900 (U.S. EPA 1997b)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)
BW	=	body weight (kg)	
		- adult	72 (U.S. EPA 1997b)
		- child	15 (U.S. EPA 1997b)

The dose absorbed per unit area per event (DA_{event}) must be calculated using Equation 4-19, 4-20, or 4-21, depending on the type of waste constituent in question and the relationship between the duration of the bath (t_{event}) and the time required for the flux of chemical through the skin to reach steady state (t^*, a chemical-specific constant). For organic constituents, when t_{event} is less than t^*, DA_{event} is computed using Equation 4-19 (U.S. EPA 1992b).

$$DA_{event} = \frac{C_{gw} \cdot 2K_p^{\,w} \sqrt{\dfrac{6\tau t_{event}}{\pi}}}{10^3 cm^3/L} \qquad \textbf{(4-19)}$$

where:

			Default
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	calculated
C_{gw}	=	constituent concentration in groundwater (mg/L)	Equation 2-6
$K_p^{\,w}$	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
ô	=	lag time (hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25(adult) or 0.33 (child) (U.S. EPA 1997b)

For organic constituents, when t_{event} is greater than or equal to t^*, Equation 4-20 is used (U.S. EPA 1992b).

$$DA_{event} = \frac{C_{gw} \cdot K_p^{w} \left[\frac{t_{event}}{1 + B} + 2\tau \left(\frac{1 + 3B + (3B)^2}{1 + B^2} \right) \right]}{10^3 cm^3/L} \qquad \textbf{(4-20)}$$

where:

			Default
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	calculated
C_{gw}	=	constituent concentration in groundwater (mg/L)	Equation 2-6
K_p^{w}	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25 (adult) 0.33 (child) (U.S. EPA 1997b)
B	=	Bunge constant (unitless)	chem-specific (Appendix A-1)
ô	=	lag time (hr)	chem-specific (Appendix A-1)

For inorganic constituents, DA_{event} is calculated using Equation 4-21.

$$DA_{event} = \frac{C_{gw} \cdot K_p^{w} \cdot t_{event}}{10^3 cm^3/L} \qquad \textbf{(4-21)}$$

where:

			Default
DA_{event}	=	dose absorbed per unit area per event (mg/cm^2-event)	calculated
C_{gw}	=	waste constituent conc. in groundwater (mg/L)	Equation 2-6
K_p^{w}	=	skin permeability constant in water (cm/hr)	chem-specific (Appendix A-1)
t_{event}	=	duration of event (hr/event)	0.25 (adult) 0.33 (child) (U.S. EPA 1997b)

Using Equation 4-17, the DRAS calculates the risk associated with groundwater exposure via dermal absorption during bathing for each waste constituent. Subsequently, the DRAS uses Equation 4-2 and sums the constituent-specific risks to determine the total risk for the groundwater dermal absorption pathway.

4.3.1.3 Adult Inhalation of Volatiles from Groundwater During Showering

This section describes computation of pathway leachate concentration delisting levels and risks associated with exposure to carcinogenic constituents in groundwater via inhalation of volatiles by adults while showering. Section 4.3.1.3.1 presents the equations used to calculate the delisting levels for standard multi year delistings. Section 4.3.1.3.2 presents the equations used to calculate constituent-specific and pathway-specific risk for one-time delistings.

The method used to determine waste constituent-specific exposures resulting from inhalation of constituents in tap water is based on information used to support the "Petroleum Refinery Listing Determination" (U.S. EPA 1997a). This project used a method developed by McKone (1987) to estimate a time-varying constituent concentration in three compartments: the shower, the bathroom, and the house. Therefore, U.S. EPA Region 6 uses simplified versions of the equations that compute a conservative constant concentration for each of the three compartments. The simplified equations are based on the steady-state model developed by McKone and Bogen (1992), which calculates a chemical concentration in air for each of the three compartments.

4.3.1.3.1 Delisting Level Analysis

The pathway total waste constituent concentration delisting levels for carcinogenic constituents inhaled when groundwater is used for showering ($C_{dl\text{-}inhale}$) are calculated for each constituent using the maximum allowable concentration in groundwater for inhalation ($C_{gw\text{-}inhale}$) at the specified target risk level. To generate $C_{dl\text{-}inhale}$, $C_{gw\text{-}inhale}$ is multiplied by the constituent's DAF_{va} as shown in Equation 4-22.

$$C_{dl-inhale} = DAF_{va} \cdot C_{gw-inhale} \tag{4-22}$$

where:

			Default
$C_{dl\text{-}inhale}$	=	pathway leachate concentration delisting level for shower inhalation (mg/L)	calculated
DAF_{va}	=	waste volume-adjusted dilution attenuation factor (unitless)	Equation 2-5
$C_{gw\text{-}inhale}$	=	maximum allowable constituent concentration in groundwater used for showering (mg/L)	Equation 4-24

To generate $C_{gw\text{-}inhale}$, one must first generate the sum of products of the maximum allowable air concentration and corresponding exposure time in three compartments during showering ($CET_{\max}$) using Equation 4-23.

$$CET_{\max} = \frac{TR \cdot AT \cdot 365 \ days/year}{EV \cdot EF \cdot IFA_{adj} \cdot CSF_{inhal}} \tag{4-23}$$

where:

		Default
CET_{max} =	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-23)
TR =	individual target risk level (unitless)	1 x 10^{-5} (U.S. EPA 1997b) or user-specified
AT =	averaging time (years)	75 (U.S. EPA 1997b)
EV =	event frequency (events/day)	1 (U.S. EPA 1997b)
EF =	exposure frequency (days/year)	350 (U.S. EPA 1997b)
IFA_{adj} =	inhalation factor, age-adjusted ([m^3-year]/[kg-day])	10.7 (Equation 3-3)
ET_{comp} =	exposure time in each compartment (days/shower)	(McKone and Bogen 1992)
	- shower	0.00792
	- bathroom	0.0338
	- house	0.625
CSF_{inhal} =	constituent inhalation cancer slope factor (mg/kg-day)$^{-1}$	chem-specific (Appendix A-1)

Once CET_{max} for a given waste constituent is known, $C_{gw\text{-}inhale}$ is back-calculated based on the linear relationship of the transient shower concentration equations as described in Equations 4-31 through 4-33. In other words, the linearity of these equations allows us to simply calculate the fractions of the average compartment concentrations to the groundwater concentration in the forward calculations and then divide CET_{max} by the sum of products of concentration fraction and corresponding exposure time to get $C_{gw\text{-}inhale}$. This requires that one forward calculation be performed at an arbitrary TCLP concentration before the backward calculations begin. The allowable groundwater concentration can then be computed using Equation 4-24.

$$C_{gw-inhale} = \frac{CET_{\max}}{\sum_{comp} F_{GW,\ comp} \cdot ET_{comp}} \tag{4-24}$$

where:

		Default
$C_{gw\text{-}inhale}$ =	maximum allowable constituent concentration in groundwater used for showering (mg/L)	calculated
CET_{max} =	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Eq. 4-23)
$F_{GW,\ comp}$ =	fraction of average waste constituent concentration in the compartment over the groundwater waste constituent concentration based on a forward calculation	Equation 4-25
ET_{comp} =	exposure time in each compartment (days/shower)	(McKone and Bogen 1992)

- shower	0.00792
- bathroom	0.0338
- house	0.625

The relationship between the average waste constituent concentration in a given compartment (determined using Equations 4-28 to 4-33) divided by an established waste concentration in the groundwater is calculated in Equation 4-25. This relationship is unique to each chemical since it is based on the volatilization of the chemical in the compartment. However, the relationship between this calculated fraction and the allowable delisting concentration is linear.

$$F_{GW,\ comp} = \frac{C_{F-avg,\ comp}}{C_{F-gw-inhale}} \qquad \textbf{(4-25)}$$

where:

$F_{GW,\ comp}$ = fraction of average waste constituent concentration in the compartment over the groundwater waste constituent concentration based on a forward calculation — Equation 4-25
$C_{F\text{-}avg,\ comp}$ = average compartment concentration from a forward calculation (mg/m^3) — Equations 4-28 - 4-30
$C_{F\text{-}gw\text{-}inhale}$ = groundwater concentration that C_{Favg} is based on (mg/L)

Once $C_{gw\text{-}inhale}$ is determined from Equation 4-24, the DRAS performs the back-calculation described in Equation 4-22 to generate the pathway delisting level ($C_{dl\text{-}inhale}$). If the TCLP concentration of a given waste constituent exceeds its pathway leachate concentration delisting level, $C_{dl\text{-}inhale}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented Section 4.3.1.3.2.

4.3.1.3.2 Risk Analysis

In the risk analysis, the DRAS computes the risk for an adult receptor exposed to carcinogenic waste constituents through inhalation of air concentrations contaminated during showering with the groundwater and subsequent exposure. The shower model used in this analysis is based on the equations presented in McKone (1987). The shower model estimates the change in the shower air concentration based on the mass of waste constituent lost by the contaminated groundwater and the air exchange rate between the various model compartments (shower, bathroom, and house). The constituent-specific risk is calculated for each inhalation compartment using Equation 4-26.

$$R = \frac{CET \cdot 1000 L/m^3 \cdot CSF_{inhal} \cdot IFA_{adj} \cdot EF \cdot EV \cdot}{AT \cdot 365\ days/year} \qquad \textbf{(4-26)}$$

where:

			Default
R	=	risk from inhalation during showering for constituent (unitless)	calculated
CET	=	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-27)
1000L/m^3	=	conversion from cubic meters to liters	
CSF_{inhal}	=	constituent inhalation cancer slope factor (mg/kg-day)$^{-1}$	chem-specific (Appendix A-1)
IFA_{adj}	=	inhalation factor, age-adjusted ([m^3-year]/[kg-day])	10.67 (Equation 3-3)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
EV	=	event frequency (events/day)	1 (U.S. EPA 1997b)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)

$C_{air,i}$ is the sum of the exposure concentrations in the shower, bathroom, and house compartments computed using Equation 4-27.

$$CET = \sum_{comp} C_{avg,\ comp} \cdot ET_{comp} = C_{avg,\ s}\ ET_s + C_{avg,\ b}\ ET_b + C_{avg,\ h}\ ET_h \qquad \textbf{(4-27)}$$

where:

			Default
CET	=	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-27)
$C_{avg,s}$	=	average constituent air conc. in shower (mg/L)	Equation 4-28
$C_{avg,b}$	=	average constituent air conc. in bathroom (mg/L)	Equation 4-29
$C_{avg,h}$	=	average constituent air conc. in house (mg/L)	Equation 4-30
ET_{comp}	=	exposure time in each compartment (days/shower)	U.S EPA 1997b
		- shower	0.00792
		- bathroom	0.0338
		- house	0.625

To calculate the total constituent air concentration inhaled, the average constituent-specific air concentration ($C_{avg,i}$) that the individual is exposed to in each inhalation compartment (shower, bathroom, and house) is calculated using Equations 4-28 through 4-30, respectively.

$$C_{avg,\,s} = \frac{\sum_{t\,=\,1}^{N} y_{s,\,t}}{N} \qquad \textbf{(4-28)}$$

where:

			Default
$C_{avg,s}$	=	average air concentration in shower (mg/L)	calculated
t	=	calculational time step index	
N	=	total number of time steps for the time in the shower	57 (n/ Ä*t*)
Ät	=	time step (min)	0.2 (McKone 1987)
n	=	time in shower (min)	11.4 (U.S. EPA 1997b)
$y_{s,t}$	=	air concentration in shower at time *t* (mg/L)	Equation 4-31

and:

$$C_{avg,b} = \frac{\sum_{t\,=\,1}^{N} y_{b,\,t}}{N} \qquad \textbf{(4-29)}$$

where:

			Default
$C_{avg,b}$	=	average air concentration in bathroom (mg/L)	calculated
t	=	calculational time step index	
N	=	total number of time steps for the time in the bathroom excluding time in shower	243 (n/Ä*t*)
Ät	=	time step (min)	0.2 (McKone 1987)
n	=	time in bathroom excluding time in shower (min)	48.6 (U.S. EPA 1997b)
$y_{b,t}$	=	air concentration in bathroom at time *t* (mg/L)	Equation 4-32

and:

$$C_{avg,\,h} = \frac{\sum_{t\,=\,1}^{N} y_{h,\,t}}{N} \qquad \textbf{(4-30)}$$

where:

			Default
$C_{avg,h}$	=	average air concentration in house (mg/L)	calculated
t	=	calculational time step index	
N	=	total number of time steps for the time in the house excluding time in the bathroom and shower	4500 (n/Ä*t*)
n	=	time in house excluding shower and bathroom time (min)	900 (U.S. EPA 1997b)

Ät = time step (min) — 0.2 (McKone 1987)
$y_{h,t}$ = air concentration in house at time *t* (mg/L) — Equation 4-33

For each of the compartments, the average concentration of the waste constituent in the air is dependent on calculation of the gas phase of the waste constituent in that compartment. For example, calculation of the average air concentration in the shower, $C_{avg,s}$, is dependent on calculation of the gas phase constituent concentration in the shower at time *t* ($y_{s,t+1}$) as derived using Equation 4-31. The air exchange rate between the shower and the bathroom is included in the estimation of the gas phase concentration of the constituents in the shower (McKone 1987).

$$y_{s,t+1} = y_{s,t} + \left[(Q_{gs} \cdot (y_{b,t} - y_{s,t})) + (I_s \cdot C_{gw} \cdot f_{em,s})\right] \cdot \left(\frac{t_{t+1} - t_t}{V_s}\right) \qquad \textbf{(4-31)}$$

where:

Default
$y_{s,t+1}$ = gas phase constituent concentration in the shower at end of time step (mg/L) — calculated
$y_{s,t}$ = gas phase constituent concentration in the shower at beginning of time step (mg/L) — calculated (0.0 for 1st time step)
Q_{gs} = volumetric gas exchange rate between shower and bathroom (L/min) — 100 (McKone 1987)
$y_{b,t}$ = gas phase constituent concentration in bathroom at beginning of time step (mg/L) — calculated (0.0 for 1st time step)
$(t_{t=1} - t_t)$ = calculational time step (min) — 0.2 (McKone 1987)
I_s = shower water use (L/min) — 5.5 (McKone 1987)
C_{gw} = waste constituent concentration in the receptor well (mg/L) — Equation 2-6
$f_{em,s}$ = fraction of constituent emitted from shower water use (unitless) — (Equation 4-34)
V_s = volume of shower stall (L) — 2,300 (U.S. EPA 1997b)

To calculate the average concentration of the waste constituent in the air of the bathroom, $C_{avg,b}$, the gas phase constituent concentration in the bathroom at time *t* ($y_{b,t+1}$) is calculated using Equation 4-32. Equation 4-32 is calculated based on total bathroom water use of 125 L/day and exposure duration in bathroom of 1 hr/day (U.S. EPA 1997b).

$$y_{b,t+1} = y_{b,t} + \left[(Q_{gs} \cdot (y_{s,t} - y_{b,t})) - (Q_{gb} \cdot (y_{b,t} - y_{h,t})) + (I_b \cdot C_{gw} \cdot f_{em,b})\right] \cdot \left(\frac{t_{t+1} - t_t}{V_b}\right) \qquad \textbf{(4-32)}$$

where:

			Default
$y_{b,t+1}$	=	gas phase constituent concentration in the bathroom at end of time step (mg/L)	calculated
$y_{b,t}$	=	gas phase constituent concentration in the bathroom at for previous time step (mg/L)	calculated (0.0 for 1st time step)
Q_{gs}	=	volumetric gas exchange rate between shower and bathroom (L/min)	100 (McKone 1987)
$y_{s,t}$	=	gas phase constituent concentration in the shower for previous time step (mg/L)	calculated (0.0 for 1st time step)
Q_{gb}	=	volumetric gas exchange rate between bathroom and house (L/min)	300 (McKone 1987)
$y_{h,t}$	=	gas phase constituent concentration in bathroom at for previous time step (mg/L)	calculated (0.0 for 1st time step)
$(t_{t+1} - t_t)$	=	calculational time step (min)	0.2 (McKone 1987)
I_b	=	bathroom water use (L/min)	2.08 (McKone 1987)
C_{gw}	=	waste constituent concentration in the receptor well (mg/L)	Equation 2-6
$f_{em,b}$	=	fraction of constituent emitted from bathroom water use (unitless)	(Equation 4-34)
V_b	=	volume of bathroom (L)	13,600 (U.S. EPA 1997b)

To calculate the average air concentration of a waste constituent in the house, $C_{avg,h}$, the gas phase constituent concentration in the house at time t ($y_{h,t+1}$) is calculated using Equation 4-33. Equation 4-33 is calculated based on total bathroom water use of 201 L/day and 16 hours of household exposure (U.S. EPA 1997b).

$$y_{h,t+1} = y_{h,t} + \left[(Q_{gb} \cdot (y_{b,t} - y_{h,t})) - (Q_{gh} \cdot (y_{h,t} - y_{a,t})) + (I_h \cdot C_{gw} \cdot f_{em,h})\right] \cdot \left(\frac{t_{t+1} - t_t}{V_h}\right) \qquad \textbf{(4-33)}$$

where:

			Default
$y_{h,t+1}$	=	gas phase constituent concentration in the house at end of time step (mg/L)	calculated
$y_{h,t}$	=	gas phase constituent concentration in the house for previous time step (mg/L)	calculated (0.0 for1st time step)
Q_{gb}	=	volumetric gas exchange rate between the bathroom and house (L/min)	300 (McKone 1987)
$y_{b,t}$	=	gas phase constituent concentration in the bathroom for previous time step (mg/L)	calculated (0.0 for1st time step)
Q_{gh}	=	volumetric gas exchange rate between house and atmosphere (L/min)	2,325 (McKone 1987)
$y_{a,t}$	=	gas phase constituent concentration in the atmosphere (mg/L)	assumed 0.00
$(t_{t+1} - t_t)$	=	calculational time step (min)	0.2 (McKone 1987)
I_h	=	water use in house - other than bathroom (L/min)	0.21 (McKone 1987)
C_{gw}	=	waste constituent concentration in the receptor well (mg/L)	Equation 2-6
$f_{em,h}$	=	fraction of constituent emitted from household	(Equation 4-34)

		water use - other than bathroom (unitless)	
V_h	=	volume of house (L)	310,000 (U.S. EPA 1997b)

In order to calculate the gas phase constituent concentration in the three compartments (shower, bathroom and house) at time *t*, the fraction of the waste constituent emitted from the contaminated groundwater (f_{em}) must be calculated for each compartment. The fraction emitted for the shower compartment ($f_{em,\ s}$) is calculated using the formula in Equation 4-34. The fraction emitted for the bathroom compartment and the house compartment are calculated using the general formula in Equation 4-38. The fraction of the gas phase saturation (f_{sat}) is unique to each compartment, thereby determining a unique fraction emitted (f_{em}) for each compartment.

$$f_{em,i} = (1 - f_{sat,i}) \cdot (1 - e^{-N}) \qquad \textbf{(4-34)}$$

where:

			Default
$f_{em,\ i}$	=	fraction of constituent emitted from contaminated groundwater for each compartment *i* (unitless)	calculated
N	=	Dimensionless mass transfer coefficient (unitless)	Equation 4-36
f_{sat}	=	Fraction of gas phase saturation for each compartment *i* (unitless)	Equation 4-35

The fraction of the gas phase saturation (f_{sat}) for each compartment is calculated using Equation 4-35.

$$f_{sat,s} = \frac{y_{i,t}}{H' \cdot C_{gw}} \qquad \textbf{(4-35)}$$

where:

			Default
f_{sat}	=	fraction of gas phase saturation for each compartment *i* (unitless)	calculated
$y_{i,t}$	=	gas phase constituent concentration in each compartment *i* (mg/L)	
		$y_{s,t}$ = for shower compartment	Equation 4-31
		$y_{b,t}$ = for bathroom compartment	Equation 4-32
		$y_{h,t}$ = for house compartment	Equation 4-33
H'	=	dimensionless Henry's law constant (H × 41)	chem-specific (Appendix A)
C_{gw}	=	waste constituent concentration in the receptor well (mg/L)	Equation 2-6

To calculate the fraction of waste constituent emitted from the contaminated groundwater (f_{em}), the dimensionless overall mass transfer coefficient (*N*) is determined using Equation 4-36.

$$N = K_{ol} \cdot \left(\frac{6}{d_p} \right) \cdot \left(\frac{h}{v_t} \right) \qquad \textbf{(4-36)}$$

where:

			Default
N	=	dimensionless mass transfer coefficient (unitless)	calculated
K_{ol}	=	overall mass transfer coefficient (cm/sec)	Equation 4-37
d_p	=	droplet diameter (cm)	0.098 (McKone 1987)
h	=	nozzle height (cm)	180 (McKone 1987)
v_t	=	terminal velocity (cm/sec)	400 (McKone 1987)

To calculate the dimensionless overall mass transfer coefficient (N), the overall mass transfer coefficient (K_{ol}) is computed using Equation 4-37.

$$K_{ol} = \beta \cdot \left(\frac{2.5}{D_w^{2/3}} + \frac{1}{D_a^{2/3} H'} \right)^{-1} \qquad \textbf{(4-37)}$$

where:

			Default
K_{ol}	=	Overall mass transfer coefficient (cm/sec)	Calculated
$\hat{a}$	=	Proportionality constant (cm/sec)$^{-1/3}$	216 (McKone 1987)
D_w	=	Diffusion coefficient in water (cm^2/sec)	Chem-specific (Appendix A)
D_a	=	Diffusion coefficient in air (cm^2/sec)	Chem-specific (Appendix A)
H'	=	Dimensionless Henry's law constant (H × 41)	Chem-specific (Appendix A)

The fraction of the gas phase saturation (f_{sat}) is unique to each compartment, thereby determining a unique fraction emitted (f_{em}) for each compartment. The fraction emitted from the bathroom or household water use is a function of the input transfer efficiency (or maximum fraction emitted) and the driving force for mass transfer. For the bathroom, the fraction emitted is calculated as follows:

$$f_{em,b} = \left(1 - \frac{y_b}{H' \cdot C_{gw}} \right) (\varepsilon_{transfer} \cdot f_{max,b}) \qquad \textbf{(4-38)}$$

where

			Default
$f_{em,b}$	=	fraction of constituent emitted from bathroom water use (unitless)	Calculated
y_b	=	gas phase constituent concentration in the bathroom (mg/L)	Equation 4-32

H'	=	Dimensionless Henry's law constant	Chem-specific (Appendix A)
C_{gw}	=	waste constituent concentration in the receptor well (mg/L)	Equation 2-6
$\text{å}_{transfer}$	=	transfer efficiency for non-shower water use (unitless)	Equation 4-40
$f_{max,b}$	=	maximum transfer efficiency for bathroom water use (unitless)	0.5 (McKone, 1987)

The fraction emitted from the household water use is calculated as follows:

$$f_{em,h} = \left(1 - \frac{y_h}{H' \cdot C_{gw}}\right)(\varepsilon_{transfer} \cdot f_{max,h}) \tag{4-39}$$

where

			Default
$f_{em,\,h}$	=	fraction of constituent emitted from bathroom water use (unitless)	Calculated
y_h	=	gas phase constituent concentration in the house (mg/L)	Equation 4-33
H'	=	Dimensionless Henry's law constant	41 × Henry's Law Constant
C_{gw}	=	waste constituent concentration in the receptor well (mg/L)	Equation 2-6
$\text{å}_{transfer}$	=	transfer efficiency for non-shower water use	Equation 4-40
$f_{max,\,h}$	=	maximum transfer efficiency for household water use (unitless)	0.66 (McKone, 1987)

The transfer efficiency is calculated using the following equation based on Little (1992).

$$\varepsilon_{transfer} = 2.76E{+}6\left(\frac{2.5}{(D_l \times 10^{-4})^{0.667}} + \frac{70}{(D_a \times 10^{-4})^{0.667} \times H'}\right)^{-1} \tag{4-40}$$

where

			Default
$\text{å}_{transfer}$	=	transfer efficiency for non-shower water use	Calculated
D_l	=	diffusion coefficient in water (cm^2/s)	chem-specific (Appendix A-1)
D_a	=	diffusion coefficient in air (cm^2/s)	chem-specific (Appendix A-1)
H'	=	Dimensionless Henry's law constant ($H' = 41 \times$ Henry's Law Constant)	chem-specific (Appendix A-1)

There is a theoretical maximum air concentration that can occur in a compartment. The DRAS calculates that theoretical maximum concentration, $C_{air\text{-}max}$, for each compartment using the Ideal Gas Law. The DRAS calculates $C_{air\text{-}i}$ for all three compartments and compares the compartment concentration to the theoretical maximum. If $C_{air\text{-}i}$ for a compartment is larger than $C_{air\text{-}max}$, $C_{air\text{-}i}$ for the compartment is set equal to $C_{air\text{-}max}$.

The $C_{air\text{-}i}$ or $C_{air\text{-}max}$ for each compartment are totaled in Equation 4-27 and the sum is used in Equation 4-26 to calculate the inhalation risk for each constituent. Then the risk for each waste constituent is summed using Equation 4-2 to obtain the total pathway-specific risk associated with shower inhalation of volatiles from groundwater during showering.

4.3.2 Surface Water Exposure Pathways

The surface water exposure pathways addressed by the DRAS include ingestion of surface water and ingestion of fish. An adult receptor is assured for both pathways. The pathway delisting levels and risks for these pathways are calculated using a method described in EPA Headquarters delisting docket materials (U.S. EPA 1993a, 1993b). The method uses the USLE to calculate the rate of erosion of a petitioned waste from a landfill to a surface water body. The receiving water body is assumed to be a stream. The amount of eroded waste delivered in runoff to the stream is calculated using a sediment delivery ratio, and the volume of surface water into which runoff occurs is determined by estimating the size of the stream into which the eroded waste is likely to be transported.

For the purposes of delisting, the stream is assumed to be a second-order or fifth-order stream (U.S. EPA 1993a). A second-order stream is assumed to be the smallest stream capable of supporting fish, whereas a fifth-order stream is assumed to be the smallest stream capable of serving as a community water supply. For the surface water ingestion pathway, drinking of unfiltered, untreated water is assumed. Clean Water Act regulations restrict the potential for such exposure, making this a very conservative exposure pathway. Delisting decisions based on this exposure pathway should reviewed carefully. The surface water ingestion analyses are based on the assumption that a chemical is transported to a fifth-order stream; the fish ingestion analyses are based on the assumption that the chemical is transported to a second-order stream.

Sections 4.3.2.1 and 4.3.2.2 describe the equations and default parameters used to calculate the pathway delisting levels and risks associated with the ingestion of surface water and ingestion of fish exposure pathways.

4.3.2.1 Ingestion of Surface Water

This section describes the computation of the total concentration delisting levels and risks associated with exposure of adults to carcinogenic waste constituents via ingestion of surface water. Section 4.3.2.1.1 presents the equations used to compute pathway delisting levels for carcinogenic waste constituents for standard multi year delisting petitions. Section 4.3.2.1.2 presents the equations and methods used to compute constituent-specific and pathway-specific risks for one-time delistings.

4.3.2.1.1 Delisting Level Analysis

Pathway total concentration delisting levels for carcinogens associated with ingestion of surface water ($C_{dl\text{-}water}$) are calculated using the user specified target risk level (default 1 x 10^{-5}). To calculate $C_{dl\text{-}water}$, the maximum allowable concentration of a waste constituent in surface water (C_{sw}) must first be calculated. The DRAS then back-calculates $C_{dl\text{-}water}$ using C_{sw}. Equation 4-41 is used to calculate C_{sw}.

$$C_{sw} = \frac{TR \cdot AT \cdot 365\ days/year}{EF \cdot IFW_{adj} \cdot CSF_{oral}} \tag{4-41}$$

where:

		Default
C_{sw} =	maximum allowable concentration of constituent in surface water (mg/L)	calculated
TR =	individual target risk level (unitless)	User specified (default 1 x 10^{-5})
EF =	exposure frequency (days/year)	350 (U.S. EPA 1998b)
AT =	averaging time (years)	75 (U.S. EPA 1997b)
IFW_{adj} =	water ingestion factor, age-adjusted ([L-yr]/[kg-day])	1.07 (Equation 3-4)
CSF_{oral} =	oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)

After is calculated, C_{sw}, the waste constituent's pathway total concentration delisting level, $C_{dl\text{-}water}$, for the surface water ingestion pathway is computed as shown in Equation 4-42.

$$C_{dl-water} = \frac{C_{sw}}{C_{5th-stream}} \tag{4-42}$$

where:

		Default
$C_{dl\text{-}water}$ =	pathway total concentration delisting level for ingestion of surface water (mg/kg)	calculated
C_{sw} =	maximum allowable concentration of constituent in surface water (mg/L)	Equation 4-41
$C_{5th\text{-}stream}$ =	concentration of eroded waste and soil in fifth-order stream (kg/L)	Equation 4-43

$C_{5th\text{-}stream}$ is calculated as shown in Equation 4-43.

$$C_{5th-stream} = \frac{A_w}{Q_{5th}} \cdot A \tag{4-43}$$

where:

		Default
$C_{5th\text{-}stream}$ =	concentration of eroded waste and soil in fifth-order stream (kg/L)	calculated
A_w =	rate of soil and waste erosion from landfill (kg/[acre-year])	Equation 2-51
Q_{5th} =	volumetric flow in fifth-order stream (L/year)	3.4 x 10^{11} (U.S. EPA, 1993a)
A =	area of waste management unit (acres)	Equation 2-12 or 2-13

If the total concentration of a given constituent in the waste exceeds its pathway total concentration delisting level, $C_{dl\text{-}water}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented in Section 4.3.2.1.2.

4.3.2.1.2 Risk Analysis

This section describes how the DRAS computes risks associated with the surface water ingestion exposure pathway. The DRAS computes each carcinogenic waste constituent's risk for surface water ingestion pathway as shown in Equation 4-44.

$$Risk = \frac{C_{sw} \cdot IFW_{adj} \cdot EF \cdot CSF_{oral}}{AT \cdot 365 days/year} \tag{4-44}$$

where:

		Default
$Risk$ =	risk for surface water ingestion for waste constituent (unitless)	calculated
C_{sw} =	concentration of constituent in surface water (mg/L)	Equation 4-45
IFW_{adj} =	water ingestion factor, age-adjusted ([L-year]/[kg-day])	1.07 (Equation 3-4)
EF =	exposure frequency (days/year)	350 (U.S. EPA 1997b)
CSF_{oral} =	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)
AT =	averaging time (years)	75 (U.S. EPA 1997b)

The concentration of a constituent in surface water, C_{sw}, is computed using Equation 4-45.

$$C_{sw} = C_{total\ waste} \cdot \frac{A_w}{Q_{stream}} \cdot A \qquad \textbf{(4-45)}$$

where:

			Default
C_{sw}	=	concentration of waste constituent in surface water (mg/L)	calculated
$C_{total\ waste}$	=	total concentration of constituent in waste (mg/kg)	waste-specific
A_w	=	rate of waste erosion from landfill (kg/[acre-year])	Equation 2-49
Q_{stream}	=	volumetric flow of stream (L/year)	(U.S. EPA 1994a)
		- for fifth-order stream	3.4×10^{11}
		- for second-order stream	3.3×10^{9}
A	=	area of waste management unit (acres)	Equation 2-12 or 2-13

Using Equation 4-44, the DRAS calculates the risks from ingestion of surface water for each constituent and then, using Equation 4-2, sums the individual constituent risks to determine the total risk for the surface water ingestion pathway.

4.3.2.2 Ingestion of Fish

This section describes computation of total concentration delisting levels and of risks associated with adult ingestion of fish that are exposed to petitioned carcinogenic waste constituents that erode from a waste disposal unit and are transported to surface water. The bioaccumulation factor (*BAF*) values used are for a trophic level 4 (carnivorous) fish (U.S. EPA 1995e).

Section 4.3.2.2.1 presents the equations used to compute carcinogenic waste constituent pathway delisting levels for standard multi year delistings. Section 4.3.2.2.2 presents the equations and methods used to compute constituent-and pathway-specific risks for one-time delistings.

4.3.2.2.1 Delisting Level Analysis

This section describes the equations used to calculate the pathway total concentration delisting levels for fish ingestion ($C_{dl\text{-}fish}$) for carcinogens. To perform these calculations, the DRAS must first calculate the maximum allowable fish tissue concentration such that the target risk level is not exceeded (C_{fish}). Knowing the allowable fish tissue concentration (C_{fish}), we can backcalculate to determine the maximum allowable total surface water concentration (C_{sw}). Using C_{sw}, the maximum allowable concentration of the dissolved phase of the constituent in the waste (C_{dw}) is then calculated. Finally, C_{dw} is used to calculate $C_{dl\text{-}fish}$. The first step, then, is to calculate C_{fish} for the carcinogenic constituent using Equation 4-46.

$$C_{fish} = \frac{TR \cdot AT \cdot 365 days/year \cdot BW}{EF \cdot ED \cdot CR_{fish} \cdot CSF_{oral}} \qquad \textbf{(4-46)}$$

where:

			Default
C_{fish}	=	maximum allowable concentration of constituent in fish tissue (mg/kg)	calculated
TR	=	target risk level (unitless)	user specified (1 x 10^{-5} default)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
CR_{fish}	=	fish consumption rate (kg/day)	0.02 (U.S. EPA 1997b)
CSF_{oral}	=	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)

Now, the maximum allowable concentration of the constituent in the total waste is determined, C_{sw}, using Equation 4-47 and the appropriate factor, either the *BCF* or the *BAF*, as discussed in Section 2.3.4.2, Concentration of Waste Constituent in Fish (C_{fish}).

$$C_{dw} = \frac{C_{fish}}{BAF \vee BCF} \qquad \textbf{(4-47)}$$

where:

			Default
C_{dw}	=	maximum allowable dissolved phase water concentration (mg/L)	calculated
C_{fish}	=	maximum allowable concentration of constituent in fish tissue (mg/kg)	Equation 4-43
BAF	=	fish bioaccumulation factor (L/kg) or	chem-specific (Appendix A-1)
BCF	=	fish concentration factor (L/kg)	chem-specific (Appendix A-1)

Using Equation 4-48, the maximum allowable concentration of the waste concentration in water column in the second order stream (C_{sw}) is calculated:

$$C_{sw} = C_{dw}\ [1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}] \qquad \textbf{(4-48)}$$

where:

		Default
C_{dw}	= Dissolved phase water concentration (mg/L)	Calculated
C_{sw}	= Waste concentration in water column (mg/L)	Equation 4-47
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)	

Finally, C_{sw} is used in Equation 4-49 to back calculate the delisting level ($C_{dl\text{-}fish}$) for the fish ingestion pathway.

$$C_{dl-fish} = \frac{C_{sw} \cdot Q_{2ndstream}}{A \cdot A_w} \tag{4-49}$$

where:

		Default
$C_{dl\text{-}fish}$	= pathway total concentration delisting level for fish ingestion (mg/kg)	calculated
C_{sw}	= Waste concentration in water column (mg/L)	Equation 4-48
A_w	= rate of waste erosion from landfill (kg/[acre-year])	Equation 2-51
$Q_{2ndstream}$	= volumetric flow of second-order stream (L/year)	$3.3\text{x}10^9$ (U.S.EPA 1994a)
A	= area of waste management unit (acres)	Equation 2-12 or 2-13

If the total concentration of a given waste constituent exceeds its total concentration delisting level, $C_{dl\text{-}fish}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented in Section 4.3.2.2.2.

4.3.2.2.2 Risk Analysis

In the risk analysis, the DRAS program computes the risk associated with the fish ingestion exposure pathway. Each carcinogenic waste constituent's risk for the fish is computed using Equation 4-50.

$$Risk = \frac{C_{fish} \cdot F_c \cdot CR_{fish} \cdot ED \cdot EF \cdot CSF_{oral}}{BW \cdot AT \cdot 365 \ days/year} \tag{4-50}$$

where:

Default

$Risk$	=	risk from fish ingestion for constituent (unitless)	calculated
C_{fish}	=	concentration of constituent in fish tissue (mg/kg)	Equation 4-51
F_c	=	fraction contaminated (unitless)	1.0 (U.S. EPA 1997b)
CR_{fish}	=	fish consumption rate (kg/day)	0.02 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
CSF_{oral}	=	constituent oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)

C_{fish} is calculated as shown in Equation 4-48.

$$C_{fish} = C_{dw} \cdot BAF \vee BCF \qquad \textbf{(4-51)}$$

where:

			Default
C_{fish}	=	concentration of constituent in fish tissue (mg/kg)	calculated
C_{dw}	=	concentration of dissolved phase of constituent in waste water (mg/L)	Equation 4-48
BAF	=	fish bioaccumulation factor (L/kg) or	chem-specific (Appendix A-1)
BCF	=	fish bioconcentration factor (L/kg)	chem-specific (Appendix A-1)

First the DRAS calculates the constituent-specific risk for fish ingestion using the total waste constituent concentration. Then the program sums all the constituent-specific risks for the fish ingestion pathway using Equation 4-2 to determine the total risk for the fish ingestion pathway.

4.3.3 Air Exposure Pathways

The DRAS includes an analysis of the human health risks associated with exposure of adults to carcinogens via three air exposure pathways: (1) inhalation of particulates from solid wastes disposed of in landfills, (2) inhalation of volatiles from solid wastes disposed of in landfills, and (3) inhalation of volatiles from liquid wastes disposed of in surface impoundments. The DRAS also includes analysis of the human health risks associated with exposure of children to carcinogens via air exposure by ingestion of soil contaminated by deposition of particulate matter from solid wastes disposed of in landfills. The delisting level for the air exposure pathways is expressed in terms of the total constituent concentration in the waste.

The risks associated with inhalation exposure to particulate and volatile matter can typically be determined only when inhalation HBNs are available. Inhalation cancer slopes derived from oral potency values are only used when specifically recommended by IRIS or when target organ endpoints are far from the respiratory system (U.S. EPA 2006).

Releases and transport of contaminated particulates from landfills and releases of volatiles from landfills and surface impoundments are evaluated using methods described in U.S. EPA Headquarters delisting dockets (U.S. EPA 1993b,1994a). Particulate emissions are determined by estimating respirable contaminated particulate emissions from wind erosion of landfill surfaces (U.S. EPA 1985a). The DRAS uses the AP-42 methodologies for calculating dust emissions resulting from on-site vehicular traffic and waste loading and unloading operations (U.S. EPA 1985b).

Emission of volatiles from landfills is evaluated with Shen's modification of Farmer's equation, which was developed by U.S. EPA's OAQPS (U.S. EPA 1984). The modified Farmer's equation (Equation 2-33), which was developed to estimate atmospheric emissions from covered landfills. Estimates of emissions of volatile organics from liquid wastes in surface impoundments use a methodology developed by the U.S. EPA OAQPS and known as SIMS (U.S. EPA 1990a, 1990b). The approach used by SIMS estimates (1) individual liquid- and gas-phase mass transfer coefficients for each pollutant, (2) equilibrium constants for each pollutant based on Henry's Law constants, (3) overall mass transfer coefficients for each pollutant, and (4) emissions based on the pollutant mass balance in the vicinity of the surface impoundment.

The DRAS calculates emission rates for particulates and volatiles and uses a Gaussian dispersion model to predict worst-case average chemical concentrations 1,000 feet downwind of a hypothetical land disposal facility, as documented in U.S. EPA's "Ambient Air Dispersion Model" (U.S. EPA 1985c). The equations and default parameters used to calculate the concentrations of respirable particulates and volatiles at the POE are presented in Sections 2.3.1 and 2.3.2, respectively.

The following sections describe the equations and default parameters used to calculate pathway delisting levels and assess risks associated with exposure to waste constituents via the air exposure pathways. Section 4.3.3.1 presents the equations used to compute pathway delisting levels for standard multi year delistings. Section 4.3.3.2 presents the equations used to compute constituent-and pathway-specific risks for one-time delistings.

4.3.3.1 Delisting Level Analysis

Before pathway delisting levels for carcinogens associated with the air inhalation exposure pathways can be computed, the constituent's maximum allowable respirable air concentration at the POE (C_{air}) must be calculated. C_{air} is calculated at the target risk level using Equation 4-52.

$$C_{air} = \frac{TR \cdot AT \cdot 365 \ days/year}{EF \cdot IFA_{adj} \cdot CSF} \qquad \textbf{(4-52)}$$

where:

			Default
C_{air}	=	constituent's maximum allowable respirable air concentration at POE (mg/m^3)	calculated
TR	=	target risk (unitless)	user specified (default 1 x 10^{-5})
AT	=	averaging time (years)	75 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
IFA_{adj}	=	inhalation factor, age-adjusted ([m^3-year]/[kg-day])	10.67 (Equation 3-3)
CSF	=	inhalation cancer slope factor (mg/kg-day)$^{-1}$	chem-specific (Appendix A-1)

C_{air} is used with the air transport equation (Equation 2-25) and the air emission equations (Equation 2-33 for volatiles and Equation 2-24 for particulates) to back-calculate the pathway total concentration delisting levels. The following sections describe the methods used to calculate delisting levels for the landfill particulates inhalation pathway ($C_{dl\text{-}air\text{-}p}$), the landfill volatiles inhalation pathway ($C_{dl\text{-}air\text{-}v}$), and the surface impoundment volatiles inhalation pathway (C_o). If the total concentration of a given carcinogenic waste constituent exceeds $C_{dl\text{-}air\text{-}p}$, or if the TCLP concentration of a constituent exceeds either $C_{dl\text{-}air\text{-}v}$ or C_o, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented in Section 4.3.3.2.

Landfill Particulate Delisting Levels

First the maximum allowable emission rate of the constituent, Q_p, is back-calculated from C_{air} (see equation 4-52) using Equation 4-53.

$$Q_p = \frac{C_{air} \cdot \Sigma_z \cdot U \cdot L_v}{2.03 \cdot F} \qquad \textbf{(4-53)}$$

where:

			Default
Q_p	=	maximum allowable emission rate of waste constituent particulates (mg/s)	calculated
C_{air}	=	constituent's maximum allowable air concentration at POE (mg/m^3)	Equation 4-52

$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
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 across waste unit toward the receptor (unitless)	0.15 (U.S. EPA 1994a)

Then the pathway waste concentration delisting level for respirable landfill particulates in air ($C_{dl\text{-}air\text{-}p}$) is calculated by setting the maximum allowable total emission rate for respirable particulates, Q_{p10}, as equal to Q_p and using Equation 4-54.

$$C_{dl-air-p} = \frac{Q_{p10}}{(E_{w10} + E_{v10} + E_{l10}) \cdot \frac{10^3\ mg}{1\ g} \cdot \frac{1\ hr}{3{,}600} \cdot \frac{10^{-6}\ kg}{1\ mg}} \qquad \textbf{(4-54)}$$

where:

			Default
$C_{dl\text{-}air\text{-}p}$	=	pathway total concentration delisting level for respirable landfill air particulates (mg/kg)	calculated
Q_{p10}	=	emission rate of waste constituent particulates up to 10 ì m (mg/s)	set equal to Q_p (Equation 4-53)
E_{w10}	=	wind erosion emissions rate of particulates up to 10 ì m (g/hr)	Equation 2-9
E_{v10}	=	vehicle travel emissions rate of particulates up to 10 ì m (g/hr)	Equation 2-19
E_{l10}	=	waste loading and unloading emissions rate of particulates up to 10 ì m (g/hr)	Equations 2-21 and 2-22

Landfill Volatile Delisting Levels

First the maximum allowable volatile emission flux of the constituent, E_i, from the landfill is calculated by setting the maximum allowable total emission rate of volatiles, Q_v, as equal to Q_p (calculated in Equation 4-53) and using Equation 4-55.

$$E_i = \frac{Q_v}{1000\ mg\ /\ g} \qquad \textbf{(4-55)}$$

where:

			Default
E_i	=	maximum allowable volatile emission flux of	calculated

		constituent (g/s)	
Q_v	=	maximum allowable constituent volatile emission rate (mg/s)	equal to Q_p (Equation 4-53)
1000 mg/1g conversion factor			

Then E_i is used to calculate the pathway waste constituent concentration delisting level for volatiles from a landfill ($C_{dl\text{-}air\text{-}v}$) using Shen's modification of Farmer's Landfill Volatilization Equation as presented in Equation 4-56 (adopted from EPA-450/3-84-020; *Evaluation and Selection of Models for Estimating Air Emissions from Hazardous Waste Treatment, Storage, and Disposal Facilities*).

$$C_{dl-air-v} = \frac{E_i \cdot d \cdot P_T^2 \cdot 1x10^6 \cdot 1x10^4}{D_a \cdot P_a^{10/3} \cdot A_{exposed} \cdot C_{si}} \tag{4-56}$$

where:

			Default
$C_{dl\text{-}air\text{-}v}$	=	pathway waste constituent concentration delisting level for volatiles from landfill (mg/kg)	calculated
E_i	=	maximum allowable volatile emission flux of constituent (g/s)	Equation 4-55
d	=	depth of soil cover (m)	0.1524 (U.S. EPA 1994a)
P_T	=	total sand porosity (unitless)	0.4 (U.S. EPA 1994a)
D_a	=	diffusivity of constituent in air (cm^2/s)	chem-specific (Appendix A-1)
P_a	=	air-filled sand porosity (unitless)	0.4 (U.S. EPA 1994a)
$A_{exposed}$	=	surface area (m^2)	Equation 2-15
C_{si}	=	saturation vapor concentration of *i* in landfill	Equation 4-57
$1x10^6$	=	conversion from kilograms to milligrams	
$1x10^4$	=	conversion from m^2 to cm^2	

Finally, the saturation vapor concentration (C_{si}) of waste constituent *i* in the landfill is calculated using Equation 4-57.

$$C_{si} = \frac{V_p \cdot MW_i}{R \cdot T} \tag{4-57}$$

where:

			Default
C_{si}	=	saturation vapor concentration of *i* in landfill (g/m^3)	Calculated
MW	=	molecular weight of waste constituent *i*	chem-specific (Appendix A-1)
R	=	universal gas constant (atm-m^3/mole- °K)	8.21×10^{-5}
T	=	standard temperature (°K)	298
V_p	=	Vapor Pressure (atm)	chem-specific (Appendix A-1)

Surface Impoundment Volatile Delisting Levels

The pathway leachate concentration delisting level for volatiles from waste disposed of in a surface impoundment is calculated by setting the maximum allowable total emission rate of volatiles, Q_v, as equal to Q_p (calculated in Equation 4-53) and using Equation 4-58.

$$C_{dl-air-si} = \frac{Q_v \cdot t_f \cdot \frac{86{,}400\ s}{1\ day}}{V_{si} \cdot \frac{10^3 L}{1\ m^3} \cdot \exp\left[\frac{-K \cdot A \cdot \frac{4{,}046.8\ m^2}{1\ acre} \cdot t_r \cdot \frac{86{,}400\ s}{1\ day}}{V_{si}}\right]} \qquad \textbf{(4-58)}$$

where:

			Default
$C_{di-air-si}$	=	pathway leachate concentration delisting level for volatiles from surface impoundment (mg/L)	calculated
Q_v	=	maximum allowable constituent volatile emission rate (mg/s)	set equal to Q_p (Equation 4-53)
t_f	=	time for constituent concentration to reach 1 % of $C_{dl-air-si}$ (days)	Equation 4-59
V_{si}	=	volume of liquid in surface impoundment (m^3)	unit-specific (user-provided)
K	=	overall mass transfer coefficient (m/s)	Equation 4-60
A	=	area of waste management unit (acres)	Equation 2-12 or 2-13
t_r	=	retention time of liquid in surface impoundment (days)	Equation 2-40

The time for the constituent concentration to reach one percent of $C_{di-air-si}$, t_f, is computed using Equation 4-59.

$$t_f = \frac{Ln\ (0.01) \cdot V_{si}}{(K \cdot A \cdot \frac{4{,}046.8\ m^2}{1\ acre})} \cdot \frac{1\ day}{86{,}400\ s} \qquad \textbf{(4-59)}$$

where:

			Default
t_f	=	time for constituent concentration to reach 1 percent of $C_{dl-air-si}$ (days)	calculated
V_{si}	=	volume of liquid in surface impoundment (m^3)	unit-specific (user provided)
K	=	overall mass transfer coefficient (m/s)	Equation 4-60
A	=	area of waste management unit (acres)	Equation 2-12 or 2-13

The overall mass transfer coefficient, K, is calculated using Equation 4-60.

$$K = \frac{1}{(\frac{1}{K_l} + \frac{1}{K_g \cdot K_{eq}})} \qquad \textbf{(4-60)}$$

where:

			Default
K	=	overall water to air mass transfer coefficient (m/s)	calculated
K_l	=	liquid-phase mass transfer coefficient (m/s)	Equation 4-61
K_g	=	gas-phase mass transfer coefficient (m/s)	Equation 4-62
K_{eq}	=	equilibrium constant (unitless)	Equation 4-63

K_l is calculated using Equation(4-61).

$$K_l = 2.611\,x\,10^{-7} \cdot U_{10}^{\;2} \cdot \left[\frac{D_w}{D_{ether}}\right]^{2/3} \qquad \textbf{(4-61)}$$

where:

			Default
K_l	=	liquid-phase mass transfer coefficient (m/s)	calculated
U_{10}	=	wind speed (m/s)	5.73 (U.S. EPA 1994a)
D_w	=	diffusion coefficient in water (cm^2/s)	chem-specific (Appendix A-1)
D_{ether}	=	diffusion coefficient of ether (cm^2/s)	8.56 x 10^{-6} (U.S. EPA 1994a)

K_g is calculated using Equation (4-62).

$$K_g = 4.82\,x\,10^{-3} \cdot U_{10}^{\;0.78} \cdot S_{cg}^{\;-0.67} \cdot d_e^{\;-0.11} \qquad \textbf{(4-62)}$$

where:

			Default
K_g	=	gas phase mass transfer coefficient (m/s)	calculated
U_{10}	=	wind speed (m/s)	5.73 (U.S. EPA 1994a)
S_{cg}	=	Schmidt number on gas side (unitless)	Equation 4-64
d_e	=	effective diameter of surface impoundment (m)	unit-specific (user-provided)

K_{eq} is calculated using Equation 4-63.

$$K_{eq} = \frac{H}{R \cdot T} \qquad \textbf{(4-63)}$$

where:

			Default
K_{eq}	=	equilibrium constant (unitless)	calculated
H	=	Henry's Law constant (atm-m^3/mol)	chem-specific (Appendix A-1)
R	=	universal gas law constant (atm-m^3/mol-K)	8.21 x 10^{-5}
T	=	standard temperature (K)	298

S_{cg} is calculated using Equation 4-64.

$$S_{cg} = \frac{\mu_a}{\rho_a \cdot D_a} \tag{4-64}$$

where:

			Default
S_{cg}	=	Schmidt number on gas side (unitless)	calculated
ì$_a$	=	viscosity of air (g/[cm-sec])	0.000181 (U.S. EPA 1990a)
ñ$_a$	=	density of air (g/cm^3)	1.2 x 10^{-3} (U.S. EPA 1990a)
D_a	=	diffusivity of constituent in air (cm^2/s)	chem-specific (Appendix A-1)

4.3.3.2 Risk Analysis

Risk as a function of the effective air concentration of a particulate or volatile carcinogenic constituent (C_{air}) is calculated for each of the three air exposure pathways as shown in Equation 4-65.

$$Risk = \frac{C_{avg} \cdot IFA_{adj} \cdot EF \cdot CSF}{AT \cdot 365 days/year} \tag{4-65}$$

where:

			Default
$Risk$	=	risk for waste constituent (unitless)	calculated
C_{avg}	=	constituent's air concentration at POE (mg/m^3)	
		Particulates	Equation 4-66
		Landfill volatiles	Equation 4-67
		Surface Impoundment volatiles	Equation 4-71
IFA_{adj}	=	inhalation factor, age-adjusted ([m^3-year]/[kg-day])	10.67 (Equation 3-3)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
CSF	=	constituent inhalation cancer slope factor	chem-specific (Appendix A-1)

		(mg/kg-day)$^{-1}$	
AT	=	averaging time (years)	75 (U.S. EPA 1997b)

C_{avg} at the POE is determined separately for each of the three air exposure pathways using the air emission and transport equations derived in Section 2.3 and presented below for each pathway. Using Equation 4-65, the DRAS calculates the risks for receptors at the POE for the three air exposure pathways. The program then sums the individual constituent risks using Equation 4-2 to determine the total risk for each of the air inhalation pathways.

Landfill Particulates

The average air concentration of a waste constituent at the POE (C_{avg}) derived from the particulate released from the landfill is calculated using Equation 4-66.

$$C_{avg} = \frac{2.03 \cdot Q_{p10}}{\Sigma_z \cdot U \cdot L_v} \cdot F \qquad \textbf{(4-66)}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	calculated
Q_{p10}	=	emission rate of waste constituent particulates up to 10 ì m (mg/s)	Equation 2-24
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 ft (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows across waste unit 0.15 toward receptor (unitless)	(U.S. EPA 1994a)

Landfill Volatiles

For the to landfill volatiles inhalation pathway, C_{avg} for landfill volatile emissions is calculated using Equation 4-67.

$$C_{avg} = \frac{2.03 \cdot Q_v}{\Sigma_z \cdot U \cdot L_v} \cdot F \qquad \textbf{(4-67)}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at	calculated

		POE (mg/m^3)	
Q_v	=	constituent volatile emission rate (mg/s)	Equation 4-68
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows across waste unit 0.15 toward receptor (unitless)	(U.S. EPA 1994a)

For volatile emissions from a covered landfill, the emission rate, Q$_v$, is calculated using Equation 4-68.

$$Q_v = E_i \cdot \frac{1{,}000\,mg}{1\,g} \qquad \textbf{(4-68)}$$

where:

			Default
Q_v	=	constituent volatile emission rate (mg/s)	calculated
E_i	=	emission flux of constituent (g/s)	Equation 4-69
1000mg/1g		conversion factor to mg	

The emission flux of a volatile waste constituent released from the landfill, E_i, is calculated using Equation 4-69. Equation 4-69 is essentially Shen's modification of Farmer's landfill volatilization equation (U.S. EPA 1984). Farmer's equation was developed by U.S. EPA's Office of Air Quality Planning and Standards (OAQPS) to estimate the rate of emission of volatiles from a covered landfill (U.S. EPA 1984).

$$E_i = \frac{C_{waste} \cdot D_a \cdot P_a^{10/3} \cdot A_{exposed} \cdot C_{si}}{d \cdot P_T^2 \cdot 1x10^6 \cdot 1x10^4} \qquad \textbf{(4-69)}$$

where:

			Default
E_i	=	maximum allowable volatile emission flux of constituent (g/s)	calculated
C_{waste}	=	waste concentration of constituent (mg/kg)	waste-specific (user-provided)
d	=	depth of soil cover (m)	0.1524 (U.S. EPA 1994a)
P_T	=	total sand porosity (unitless)	0.4 (U.S. EPA 1994a)
D_a	=	diffusivity of constituent in air (cm^2/s)	chem-specific (Appendix A-1)
P_a	=	air-filled sand porosity (unitless)	0.4 (U.S. EPA 1994a)
$A_{exposed}$	=	surface area (m^2)	Equation 2-15
C_{si}	=	saturation vapor concentration of i in landfill	Equation 4-70
$1x10^6$	=	conversion from kilograms to milligrams	
$1x10^4$	=	conversion from m^2 to cm^2	

And the saturation vapor concentration (C_{si}) of waste constituent i in the landfill is calculated using Equation 4-70.

$$C_{si} = \frac{p_i \cdot MW_i}{R \cdot T} \tag{4-70}$$

where:

			Default
C_{si}	=	saturation vapor concentration of i in landfill (g/m^3)	Calculated
p_i	=	vapor pressure of waste constituent i (atm)	chem-specific (Appendix A-1)
MW_i	=	molecular weight of waste constituent i	chem-specific (Appendix A-1)
R	=	universal gas constant (atm-m^3/mol- K)	8.21 x 10^{-5}
T	=	standard temperature (K)	298

Surface Impoundment Volatiles

For the surface impoundment volatiles inhalation pathway, C_{avg} for use in Equation 4-65 is calculated using Equation 4-71.

$$C_{avg} = \frac{2.03 \cdot Q_v}{\Sigma_z \cdot U \cdot L_v} \cdot F \tag{4-71}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	calculated
Q_v	=	constituent volatile emission rate (mg/s)	Equation 4-72
F	=	frequency wind blows from sector of interest	0.15 (U.S. EPA 1994a)
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27

Q_v for volatile emissions from a surface impoundment is computed using Equation 4-72.

$$Q_v = C_{waste} \cdot \frac{V_{si} \cdot \frac{10^3 L}{1\ m^3} \cdot \exp\left[\frac{-K \cdot A \cdot \frac{4{,}046.8\ m^2}{acre} \cdot t_r \cdot \frac{86{,}400\ s}{1\ day}}{V_{si}}\right]}{tf \cdot \frac{86{,}400\ s}{1\ day}} \tag{4-72}$$

where:

Default

Q_v	=	volatile emission rate of constituent i from the surface impoundment (mg/s)	calculated
C_{waste}	=	liquid waste concentration of constituent (mg/L)	waste-specific (user-provided)
V_{si}	=	volume of liquid in surface impoundment (m^3)	unit-specific (user-provided)
K	=	overall mass transfer coefficient (m/s)	Equation 4-60
A	=	area of waste management unit (acres)	Equation 2-12 or 2-13
t_r	=	retention time for liquid in surface impoundment (days)	Equation 2-40
t_f	=	time for constituent concentration to reach 1 percent of C_o (days)	Equation 4-59

4.3.4 Soil Exposure Pathway

This section presents equations used to calculate pathway total concentration delisting levels and risks associated with exposure of children to soils contaminated with carcinogenic waste constituents at a POE 1,000 feet from a waste disposal unit. The soil contamination is assumed to be the result of deposition of waste particulates from the air. The equations used to compute the delisting levels and risks are based on a method described in U.S. EPA Headquarters delisting docket materials (U.S. EPA 1993a, 1993b).

Section 4.3.4.1 presents the equations used to compute pathway total concentration delisting levels for standard multi year delistings. Section 4.3.4.2 presents the equations and methods used to compute constituent-and pathway-specific risks for one-time delistings.

4.3.4.1 Delisting Level Analysis

To calculate the pathway total concentration delisting levels for exposure to carcinogenic waste constituents via ingestion of soil ,$C_{dl\text{-}soil}$, the maximum allowable concentration of each constituent in soil at the POE, C_{soil}, must first be computed. C_{soil} is the maximum concentration of the waste constituent in soil that does not cause the risk from soil ingestion to exceed the target level. C_{soil} is calculated using Equation 4-73.

$$C_{soil} = \frac{TR \cdot AT \cdot 365 days/year}{EF \cdot IFS_{adj} \cdot CSF_{oral}} \cdot 1x10^6 mg/kg \tag{4-73}$$

where:

			Default
C_{soil}	=	maximum allowable concentration of constituent in soil at POE (mg/kg)	calculated
TR	=	target risk level (unitless)	user-specified (default 1 x 10^{-5})

AT	=	averaging time (years)	75 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
IFS_{adj}	=	soil ingestion factor ([mg/year]/[kg/day])	113 (Equation 3-2)
CSF_{oral}	=	constituent oral cancer slope factor (mg/kg-day)$^{-1}$	chem-specific (Appendix A-1)

Using C_{soil}, the DRAS back-calculates the maximum allowable constituent concentration in air at the POE, C_{avg}. Assumptions include a constituent accumulation period in the soil of 1 year and a depth of mixing of 1 cm C_{avg} is calculated as shown in Equation 4-74.

$$C_{avg} = \frac{\rho_b \cdot t \cdot C_{soil} \cdot \frac{10^3\ mg}{1\ g} \cdot \frac{10^6\ cm^3}{1\ m^3} \cdot \frac{1\ kg}{10^6\ mg}}{v_d \cdot \frac{31{,}536{,}000\ s}{1\ year}} \cdot 1/yr \qquad \textbf{(4-74)}$$

where:

			Default
C_{avg}	=	maximum allowable concentration of constituent in air at POE (mg/m^3)	calculated
C_{soil}	=	maximum allowable concentration of constituent in soil at POE (mg/kg)	Equation 4-73
ñ$_b$	=	soil bulk density (g/cm^3)	1.45 (Brady 1984)
t	=	soil thickness from which particles can be ingested (m)	0.01 (U.S. EPA 1994a)
v_d	=	deposition velocity (m/s)	0.03 (U.S. DOE, 1984)

Before the pathway delisting level can be calculated, it is necessary to compute the annual average emission rate of particles up to 30 ì m in size from the disposal unit as shown in Equation 4-75.

$$E_{T30} = E_{w30} + E_{v30} + E_{l30} \qquad \textbf{(4-75)}$$

where:

			Default
E_{T30}	=	annual average emission rate of particles up to 30 ì m (g/hr)	calculated
E_{w30}	=	wind erosion emission rate of particulates up to 30 ì m (g/hr)	Equation 2-16
E_{v30}	=	vehicle travel emissions rate of particulates up to 30 ì m (g/hr)	Equation 2-20
E_{l30}	=	loading and unloading emission rate of particles up to30 ì m (g/hr)	Equation 2-23

Finally, once C_{air} and E_{T30} are known, the total concentration delisting level is computed using Equation 4-76.

$$C_{dl-soil} = \frac{C_{avg} \cdot \Sigma_z \cdot U \cdot L_v \cdot \dfrac{1{,}000\ g}{1\ kg}}{2.03 \cdot E_{T30} \cdot F \cdot \dfrac{1\ hr}{3{,}600\ s}} \qquad \textbf{(4-76)}$$

where:

			Default
$C_{dl\text{-}soil}$	=	pathway total concentration delisting level for soil ingestion (mg/kg)	calculated
C_{avg}	=	maximum allowable concentration of constituent in air at POE (mg/m^3)	Equation 4-74
Σ_z	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1986a)
L_v	=	distance virtual point to compliance point located 1,000 feet (0.3048 km) downwind (m)	Equation 2-27
E_{T30}	=	annual average emission rate of particles up to 30 ì m (g/hr)	Equation 4-75 (also, see Section 2.3.1.3)
F	=	frequency that wind blows across waste unit 0.15 toward receptor (unitless)	(U.S. EPA 1994a)

If the total concentration for a given carcinogenic waste constituent exceeds its pathway total concentration delisting level, $C_{dl\text{-}soil}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of risk analysis calculations presented Section 4.3.4.2.

4.3.4.2 Risk Analysis

In the risk analysis, the DRAS computes the risk associated with the soil ingestion exposure pathway. The risk for each carcinogenic waste constituent associated with soil ingestion at the POE is calculated. Children have higher daily soil intake rates when comparing their contact rates and lower body weights to those of adults (U. S. EPA 1997b). However, the child's exposure duration is considerably shorter than the adult's. For carcinogens, exposure is averaged over a lifetime (75 years; U. S. EPA 1997b). Exposure to contaminants via soil ingestion is calculated as a weighted average of exposures for 6 years as a child and as an adult. The total exposure is still averaged over a lifetime. For the soil ingestion exposure scenario, the constituent-specific cancer risk from ingestion of soil is calculated using Equation 4-77.

$$Risk = \frac{C_{soil} \cdot IFS_{adj} \cdot EF \cdot CSF_{oral}}{AT \cdot 365 days/year \cdot 1x10^{6} mg/kg} \qquad \textbf{(4-77)}$$

where:

			Default
$Risk$	=	risk for waste constituent (unitless)	Calculated
C_{soil}	=	conc. of constituent in soil at POE (mg/kg)	Equation 4-78
IFS_{adj}	=	soil ingestion factor ([mg/year]/[kg/day])	113 (Equation 3-2)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
CSF_{oral}	=	oral cancer slope factor $(mg/kg\text{-}day)^{-1}$	chem-specific (Appendix A-1)
AT	=	averaging time (years)	75 (U.S. EPA 1997b)

C_{soil} is calculated based on the constituent's rate of deposition onto the soil. Assuming that at any given time the soil is burdened with 1 year's accumulation of constituent from air deposition and that the mixing depth of the constituent into the soil is 1 cm, Equation 4-78 is used to calculate C_{soil}.

$$C_{soil} = \frac{q_d \cdot \frac{31{,}536{,}000\ s}{1\ year}}{\rho_b \cdot t \cdot \frac{10^6\ cm^3}{1\ m^3} \cdot \frac{1\ kg}{10^3\ g}} \qquad \textbf{(4-78)}$$

where:

			Default
C_{soil}	=	conc. of constituent in soil at the POE (mg/kg)	calculated
q_d	=	constituent deposition rate $(mg/m^2\text{-}s)$	Equation 4-79
$\tilde{n}_b$	=	soil bulk density (g/cm^3)	1.45 (Brady 1984)
t	=	soil thickness of particles that can be ingested (m)	0.01 (U.S. EPA 1994a)

The constituent deposition rate is determined using the deposition velocity and the constituent concentration in air, as shown in Equation 4-79.

$$q_d = v_d \cdot C_{avg} \qquad \textbf{(4-79)}$$

where:

			Default
q_d	=	constituent deposition rate $(mg/m^2\text{-}s)$	calculated
v_d	=	deposition velocity (m/s)	0.03 (U.S. DOE, 1984)
C_{avg}	=	downwind concentration of waste constituent at	Equation 4-80

POE (mg/m^3)

C_{avg} is calculated using Equation 4-80.

$$C_{avg} = \frac{2.03 \cdot Q_{p30}}{\Sigma_z \cdot U \cdot L_v} \cdot F \tag{4-80}$$

where:

			Default
C_{avg}	=	downwind concentration of waste constituent at POE (mg/m^3)	calculated
Q_{p30}	=	emission rate of waste constituent particulates up to 30 ì m (mg/s)	Equation 2-30
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows across waste unit 0.15 toward receptor (unitless)	(U.S. EPA 1994a)

Using Equation 4-77, the DRAS calculates the risk associated with ingestion of soil for each constituent. Then the program sums the constituent-specific risks using Equation 4-2 to determine the total risk for the soil ingestion pathway.

4.4 QUANTITATIVE ESTIMATION OF POTENTIAL NONCANCER EFFECTS

This section presents the equations used to calculate pathway delisting levels and aggregate hazards associated with exposures to noncarcinogenic chemicals via the groundwater, surface water, air, and soil exposure pathways. Calculation of delisting levels and risks associated with exposures to carcinogenic chemicals is described in Section 4.3. The equations used below to compute pathway delisting levels and aggregate hazards are adapted from the "HWIR Technical Support Document" (U.S. EPA 1995b) and U.S. EPA OSW delisting dockets (U.S. EPA 1993a, 1993c, 1994a). The general hazard equations presented in Section 4.2.2 are implemented for specific constituents-and exposure pathways. The constituent and pathway-specific $HQ_{c,p}$, is computed for each constituent using an expanded form of Equation 4-4. Then the $HQ_{c,p}$ is summed across all constituents to calculate the HI_p, for each exposure pathway, as shown in Equation 4-5. The DRAS sums all the HI_p values for the groundwater exposure pathways and the surface exposure pathways separately to assist the user in identifying the most important exposure pathways. Finally, the aggregate HI_{cum} is calculated by summing the HI_p across all the groundwater and surface exposure pathways, as shown in Equation 4-6. Constituent-and pathway-specific subscripts are omitted in subsequent equations for simplicity of presentation; the context indicates whether the value being calculated is constituent-or pathway-specific.

For exposures to noncarcinogens, the DRAS evaluates the following groundwater pathways: ingestion of groundwater, dermal exposure to groundwater during bathing (adult and child), and inhalation exposure to volatile groundwater components during showering. The equations used to compute the associated with groundwater exposures are discussed in Section 4.4.1. For surface water, the DRAS evaluates hazards associated with the following exposure pathways: ingestion of drinking water and ingestion of fish from surface water bodies contaminated by runoff from landfills; the relevant equations are discussed in Section 4.4.2. For air exposure pathways, the DRAS evaluates hazards associated with inhalation of particulates and volatiles from solid wastes disposed of in landfills and inhalation of volatiles from liquid wastes disposed of in surface impoundments; the relevant equations are discussed in Section 4.4.3. Finally, the DRAS evaluates hazards associated with a child receptor’s ingestion of soil contaminated by air deposition of particulates eroded from a landfill. The relevant equations are discussed in Section 4.4.4.

4.4.1 Groundwater Exposure Pathways

For noncarcinogens, the DRAS evaluates exposure to groundwater using the same pathways that it uses for carcinogens: ingestion of groundwater, dermal exposure to groundwater during bathing, and inhalation of volatiles during showering with groundwater. The receptor is assumed to be an adult for the ingestion and showering pathways; both adult and child receptors are considered for bathing pathway. The following sections address calculation of constituent-and pathway-specific *HQ*s for each constituent and groundwater pathway.

4.4.1.1 Adult Ingestion of Groundwater

The DRAS computes leachate concentration delisting levels and hazards associated with exposure of adults to noncarcinogenic waste constituents via the groundwater ingestion pathway. Section 4.4.1.1.1 presents the equations used to compute pathway delisting levels for standard multi year delistings. Section 4.4.1.1.2 presents the equations and methods used to compute the constituent-specific HQ and the pathway-specific HI for one-time delistings.

4.4.1.1.1 Delisting Level Analysis

Pathway leachate concentration delisting levels for noncarcinogens associated with ingestion of groundwater, $C_{dl\text{-}ingest,}$ are determined based on a default target hazard quotient (THQ) of 0.1 in order to account for HQ from other pathways and/or chemicals. To calculate $C_{dl\text{-}ingest}$, the DRAS first computes the maximum allowable groundwater concentration for the ingestion pathway for the waste constituent $C_{gw\text{-}ingest}$ using Equation 4-81.

$$C_{gw-ingest} = \frac{THQ \cdot RfD \cdot BW \cdot AT \cdot 365 days/year}{CR \cdot EF \cdot ED} \tag{4-81}$$

where:

			Default
$C_{gw\text{-}ingest}$	=	maximum allowable constituent concentration in groundwater for ingestion (mg/L)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default)
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
CR	=	water consumption rate (L/day)	2 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)

Once $C_{gw\text{-}ingest}$ is calculated, $C_{dl\text{-}ingest}$ can be computed. $C_{gw\text{-}ingest}$ is multiplied by the waste constituent's DAF_{va} to generate $C_{dl\text{-}ingest}$ as shown in Equation 4-82.

$$C_{dl-ingest} = DAF_{va} \cdot C_{gw-ingest} \tag{4-82}$$

where:

			Default
$C_{dl\text{-}ingest}$	=	pathway leachate concentration delisting level for groundwater ingestion (mg/L)	calculated
DAF_{va}	=	waste volume-adjusted dilution attenuation factor (unitless)	equation 2-5
$C_{gw\text{-}ingest}$	=	maximum allowable constituent concentration in groundwater for ingestion (mg/L)	equation 4-81

If the TCLP concentration for a given waste constituent exceeds its $C_{dl\text{-}ingest}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.1.1.2.

4.4.1.1.2 Hazard Analysis

In the hazard analysis, the DRAS computes the hazard associated with the groundwater ingestion exposure pathway. The DRAS calculates the noncarcinogenic waste constituent-specific *HQ* for ingestion of groundwater using Equation 4-83.

$$HQ = \frac{CR \cdot C_{gw} \cdot EF \cdot ED}{BW \cdot RfD \cdot AT \cdot 365 days/year} \tag{4-83}$$

where:

			Default
HQ	=	constituent-specific hazard quotient for ingestion of groundwater (unitless)	calculated
CR	=	water consumption rate (L/day)	2 (U.S. EPA 1997b)
C_{gw}	=	waste constituent concentration in groundwater (mg/L)	Equation 2-6
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)

The *HQ*s for all noncarcinogenic constituents in the petitioned waste are summed using Equation 4-5 to determine the HI_p for the groundwater ingestion pathway.

4.4.1.2 Adult and Child Dermal Contact with Groundwater During Bathing

The DRAS computes of leachate concentration delisting levels and hazards associated with adult and child exposures during bathing through dermal absorption of noncarcinogenic constituents in groundwater. Section 4.4.1.2.1 presents the equations used to compute pathway delisting levels for standard multi year delistings. Section 4.4.1.2.2 presents the equations and methods used to compute constituent-specific *HQs* and pathway-specific *HIs* for one-time delistings.

4.4.1.2.1 Delisting Level Analysis

Pathway leachate concentration delisting levels for noncarcinogens associated with adult and child dermal exposures to groundwater during bathing, $C_{dl\text{-}dermal,}$ are based on the target hazard quotient. The DRAS computes delisting levels for the adult and child receptors after separately determining their respective exposures. The DRAS then selects the delisting level that corresponds to the more sensitive receptor. $C_{dl\text{-}dermal}$ is calculated as the product of the waste constituent's maximum allowable groundwater concentration for dermal exposure during bathing, $C_{gw\text{-}dermal}$, and the waste constituent DAF_{va} generated by the EPACMTP model (see Section 2.2). Equation 4-84 shows this calculation.

$$C_{dl-dermal} = DAF_{va} \cdot C_{gw-dermal} \tag{4-84}$$

where:

		Default
$C_{dl\text{-}dermal}$ =	pathway leachate concentration delisting level for groundwater dermal contact (mg/L)	calculated
DAF_{va} =	waste volume-adjusted dilution attenuation factor (unitless)	Equation 2-6
$C_{gw\text{-}dermal}$ =	maximum allowable constituent concentration in groundwater for dermal exposure (mg/L)	Equation 4-12, 4-13, or 4-14

For organic waste constituents, $C_{gw\text{-}dermal}$ is computed using one of two methods, depending on whether the duration of the bath t_{event} is greater or less than the time required for the flux of the chemical through the skin to reach steady-state (t^*, a chemical-specific constant). For inorganic waste constituents, only one method is used. The maximum allowable constituent concentration in groundwater is computed using Equation 4-12 for organic constituents when t_{event} is less than t^*, Equation 4-13 for organic constituents when t_{event} is greater than or equal to t^*, and Equation 4-14 for inorganic constituents.

The dose of the waste constituent absorbed during bathing per unit area per event (DA_{event}) is used in each equation to calculate $C_{gw\text{-}dermal}$. DA_{event} is calculated separately for adults and children using Equation 4-85; the appropriate default parameters are used for the adult and child receptors.

$$DA_{event} = \frac{AT \cdot DAD \cdot BW \cdot 365 days/year}{EV \cdot EF \cdot ED \cdot A_{skin}} \tag{4-85}$$

where:

		Default
DA_{event} =	dose absorbed per event (mg/cm^2-event)	calculated
AT =	averaging time (years)	
	-adult	30 (U.S. EPA 1997b)
	-child	6 (U.S. EPA 1997b)
DAD =	dermally absorbed dose (mg/kg-day)	Equation 4-86
BW =	body weight (kg)	

		- adult	72 (U.S. EPA 1997b)
		- child	15 (U.S. EPA 1997b)
EV	=	event frequency (events/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	
		-adult	30 (U.S. EPA 1997b)
		-child	6 (U.S. EPA 1997b)
A_{skin}	=	exposed skin surface area (cm^2)	
		- adult	20,000 (U.S. EPA 1997b)
		- child	7,900 (U.S. EPA 1997b)

The *DAD* for dermal contact with groundwater needed in Equation 4-85 is calculated using Equation 4-86. *DAD* is the product of the *THQ* and the *RfD* for a given noncarcinogenic waste constituent.

$$DAD = THQ \cdot RfD \qquad \textbf{(4-86)}$$

where:

			Default
DAD	=	dermally absorbed dose (mg/kg-day)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)

$C_{dl\text{-}dermal}$ is calculated using Equation 4-84 for each noncarcinogenic waste constituent in the petitioned waste. If the TCLP concentration of a given waste constituent exceeds $C_{dl\text{-}dermal}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.1.2.2.

4.4.1.2.2 Hazard Analysis

In the hazard analysis, the DRAS computes the hazard associated with the dermal contact with groundwater during bathing exposure pathway. The DAD for exposure to a noncarcinogenic waste constituent during bathing is computed using Equation 4-87.

$$DAD = \frac{DA_{event} \cdot EV \cdot EF \cdot ED \cdot A_{skin}}{AT \cdot 365 days/year \cdot BW} \qquad \textbf{(4-87)}$$

where:

			Default
DAD	=	dermally absorbed dose (mg/kg-day)	calculated
DA_{event}	=	dose absorbed per event (mg/cm^2-event)	Equation 4-19, 4-20, or 4-21

EV	=	event frequency (events/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	
		- adult	30 (U.S. EPA 1997b)
		- child	6 (U.S. EPA 1997b)
A_{skin}	=	exposed skin surface area (cm^2)	
		- adult	20,000 (U.S. EPA 1997b)
		- child	7,900 (U.S. EPA 1997b)
AT	=	averaging time (years)	
		- adult	30 (U.S. EPA 1997b)
		- child	6 (U.S. EPA 1997b)
BW	=	body weight (kg)	
		- adult	72 (U.S. EPA 1997b)
		- child	15 (U.S. EPA 1997b)

DA_{event} is computed using Equation 4-19 for organic constituents when t_{event} is less than t^*, Equation 4-20 for organic constituents when t_{event} is greater than or equal to t^*, and Equation 4-21 for inorganic constituents. All three equations use the groundwater well concentration, C_{gw}, which is equal to the TCLP concentration of the constituent divided by the DAF_{va} (see Equation 2-6).

The constituent-specific HQ for the dermal contact with groundwater during bathing exposure pathway is computed as shown in Equation 4-88 (U.S. EPA 1992b).

$$HQ = \frac{DAD}{RfD} \tag{4-88}$$

where:

			Default
HQ	=	constituent-specific hazard quotient for dermal contact with groundwater during bathing (unitless)	calculated
DAD	=	dermally absorbed dose (mg/kg-day)	Equation 4-87
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)

Using Equation 4-88, the DRAS calculates the constituent-specific HQs for groundwater exposure via dermal absorption during bathing and, using Equation 4-5, sums the HQs to determine the total HI_p for the groundwater dermal absorption pathway.

4.4.1.3 Adult Inhalation of Volatiles from Groundwater During Showering

The DRAS computes leachate concentration delisting levels and hazards associated with exposure of adults to noncarcinogenic volatile constituents groundwater during showering. Exposure of children is not evaluated for this pathway because the adult resident receptor scenario has been found to be protective of children (U.S. EPA 1997a).

The method used to determine waste constituent-specific exposures resulting from inhalation of constituents in tap water is based on information used to support the " Petroleum Refinery Listing Determination" (U.S. EPA 1997a). This project used a method developed by McKone (1987) to estimate a time-varying constituent concentration in three compartments: the shower, the bathroom, and the house.

The constraints associated with implementing the risk algorithms in a QuattroPro Version 6.0 spreadsheet precluded incorporating and solving time-varying shower equations because a complex system of macros and linked analyses would have been required. Therefore, U.S. EPA Region 6 uses simplified versions of the equations that compute a conservative constant concentration for each of the three compartments. The simplified equations are based on the steady-state model developed by McKone and Bogen (1992), which calculates a chemical concentration in air for each of the three compartments.

Section 4.4.1.3.1 presents the equations used to compute delisting levels for standard multi year delistings. Section 4.4.1.3.2 presents the equations used to compute the constituent-specific *HQ* and pathway-specific *HI* for one-time delistings.

4.4.1.3.1 Delisting Level Analysis

The pathway leachate concentration delisting levels for noncarcinogens associated with adult inhalation exposure to groundwater during showering, $C_{dl\text{-}inhale}$, are back-calculated from the maximum allowable groundwater concentration for the showering exposure pathway, $C_{gw\text{-}inhale}$. Specifically, $C_{dl\text{-}inhale}$ is calculated as the product of $C_{gw\text{-}inhale}$ and the constituent's DAF_{va} using Equation 4-89.

$$C_{dl-inhale} = DAF_{va} \cdot C_{gw-inhale} \tag{4-89}$$

where:

			<u>Default</u>
$C_{dl\text{-}inhale}$	=	pathway leachate concentration delisting level for shower inhalation (mg/L)	calculated
DAF_{va}	=	waste volume-adjusted dilution attenuation factor (unitless)	Equation 2-5
$C_{gw\text{-}inhale}$	=	maximum allowable constituent concentration in groundwater for inhalation (mg/L)	Equation 4-91

To calculate $C_{gw\text{-}inhale}$, one must first calculate the maximum allowable concentration volatilized into air, CET_{max}, during showering. For noncarcinogens, CET_{max}, is calculated by DRAS using the target hazard quotient and the constituent's inhalation RfD. Equation 4-90 shows how CET_{max}, is calculated.

$$CET_{\max} = \frac{THQ \cdot RfD \cdot BW \cdot AT \cdot 365 days/year}{IR \cdot EV \cdot EF \cdot ED} \qquad \textbf{(4-90)}$$

where:

			Default
CET	=	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-27)
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfD	=	constituent inhalation reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
IR	=	inhalation rate (m^3/day)	20 (U.S. EPA 1997b)
EV	=	event frequency (showers/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)

CET_{max}, is the maximum allowable constituent concentration in air that will not allow the acceptable hazard level to be exceeded. Once CET_{max}, is calculated for a given waste constituent, the DRAS calculates $C_{gw\text{-}inhale}$ based on the inhalation exposure in the shower, bathroom, and house compartments combined. The DRAS calculates $C_{gw\text{-}inhale}$ using Equation 4-91.

$$C_{gw-inhale} = \frac{CET_{\max}}{\sum_{comp} F_{GW,\ comp}\ ET_{comp}} \qquad \textbf{(4-91)}$$

where:

			Default
$C_{gw\text{-}inhale}$	=	maximum allowable constituent concentration in groundwater used for showering (mg/L)	calculated
CET_{max},	=	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-90)
$F_{GW,\ comp}$	=	fraction of average waste constituent concentration in the compartment over the groundwater waste constituent concentration based on a forward calculation	Equation 4-25
ET_{comp}	=	exposure time in compartment (days/shower)	(U.S. EPA 1997b)
		- shower	0.00792

- bathroom	0.0338
- house	0.625

If the TCLP concentration for a given waste constituent exceeds $C_{dl\text{-}inhale}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.1.2.3.

4.4.1.3.2 Hazard Analysis

In the hazard analysis, the DRAS computes the hazard associated with adult inhalation exposure to a noncarcinogenic waste constituent during showering with groundwater. To determine total inhalation exposure to volatilized waste constituents in the house during and after showering, the DRAS computes the air concentration of waste constituents in three compartments of the house; the shower, the bathroom and the house. The DRAS computes each constituent's *HQ* for this exposure pathway using Equation 4-92.

$$HQ = \frac{CET \cdot EV \cdot EF \cdot ED \cdot IR \cdot 1000L/m^3}{AT \cdot 365days/year \cdot RfD \cdot BW} \tag{4-92}$$

where:

			Default
HQ	=	hazard quotient for inhalation during showering for constituent (unitless)	calculated
CET	=	sum of products of maximum allowable constituent air concentration in specific compartment and corresponding exposure time (mg-days/m^3)	calculated (Equation 4-27)
EV	=	event frequency (showers/day)	1 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ET_{comp}	=	exposure time in compartment (days/shower)	(U.S. EPA 1997b)
		- shower	0.00792
		- bathroom	0.0338
		- house	0.625
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
IR	=	inhalation rate (m^3/day)	20 (U.S. EPA 1997b)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
RfD	=	constituent's inhalation reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
$1000L/m^3$		conversion factor for cubic meters to liters	

CET is calculated by summing the waste constituent concentration for each of the three compartments in the house using Equation 4-27 and the appropriate default parameters. There is a theoretical maximum air concentration that can occur in a compartment. The DRAS calculates that theoretical maximum concentration, $C_{air\text{-}max}$, for each compartment using the Ideal Gas Law. The DRAS calculates $C_{air\text{-}i}$ for all three compartments and compares the compartment concentration to the theoretical maximum. If $C_{air\text{-}i}$ exceeds $C_{air\text{-}max}$ for a given waste constituent, $C_{air\text{-}i}$ is reset to the value of $C_{air\text{-}max}$. Using Equation 4-92 the user calculates the constituent-specific HQs for groundwater exposure via inhalation during showering. The DRAS then sums these HQs using Equation 4-5 to determine the HI_p for the pathway.

4.4.2 Surface Water Exposure Pathways

The surface water exposure pathways addressed by the DRAS include ingestion of surface water and ingestion of fish. An adult receptor is assumed for both pathways. The pathway delisting levels and hazards for these pathways are calculated using a method described in EPA Headquarters delisting docket materials (U.S. EPA 1993a, 1993b). The method uses the USLE to calculate the rate of erosion of a petitioned waste from a landfill to a surface water body. The receiving water body is assumed to be a stream. The amount of eroded waste delivered in runoff to the stream is calculated using a sediment delivery ratio, and the volume of surface water into which runoff occurs is determined by estimating the size of the stream into which the eroded waste is likely to be transported.

For the purposes of delisting, the stream is assumed to be a second-order or fifth-order stream (U.S. EPA 1993a). A second-order stream is assumed to be the smallest stream capable of supporting fish, whereas a fifth-order stream is assumed to be the smallest stream capable of serving as a community water supply. The surface water ingestion analyses are based on the assumption that a noncarcinogenic waste constituent is transported to a fifth-order stream; the fish ingestion analyses are based on the assumption that the chemical is transported to a second-order stream.

Sections 4.4.2.1 and 4.4.2.2 describe the equations and default parameters used to calculate the pathway delisting levels and hazard associated with the ingestion of surface water and ingestion of fish exposure pathways.

4.4.2.1 Ingestion of Surface Water

The DRAS computes total concentration delisting levels and hazards associated with exposure of adults to noncarcinogenic waste constituents via the surface water ingestion pathway. Section 4.4.2.1.1 presents the equations used to compute pathway total concentration delisting levels for standard multi year delistings. Section 4.4.2.1.2 presents the equations and methods used to compute constituent- and pathway-specific hazards for one-time delistings.

4.4.2.1.1 Delisting Level Analysis

Pathway total concentration delisting levels for noncarcinogens associated with ingestion of surface water, $C_{dl\text{-}water}$, are computed for the target hazard quotient.. To calculate $C_{dl\text{-}water}$, it is necessary to first calculate the maximum allowable concentration of the waste constituent in surface water, C_{sw}, using Equation 4-93. The DRAS then calculates $C_{dl\text{-}water}$ using C_{sw}.

$$C_{sw} = \frac{THQ \cdot RfD \cdot BW \cdot AT \cdot 365 days/year}{CR \cdot EF \cdot ED} \qquad \textbf{(4-93)}$$

where:

			Default
C_{sw}	=	maximum allowable concentration of constituent in surface water (mg/L)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfD	=	reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
CR	=	water consumption rate (L/day)	2.0 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)

Once C_{sw} is known, $C_{dl\text{-}water}$ is calculated using Equation 4-94.

$$C_{dl-water} = \frac{C_{sw}}{C_{5th-stream}} \qquad \textbf{(4-94)}$$

where:

			Default
$C_{dl\text{-}water}$	=	pathway total waste concentration delisting	calculated

		level for ingestion of surface water (mg/kg)	
C_{sw}	=	maximum allowable concentration of constituent in surface water (mg/L)	Equation 4-93
$C_{5th\text{-}stream}$	=	concentration of constituents in fifth-order stream (kg/L)	Equation 4-43

If the total concentration of a constituent in the waste exceeds its pathway total concentration delisting level, $C_{dl\text{-}water}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.2.1.2.

4.4.2.1.2 Hazard Analysis

In the hazard analysis, the DRAS computes hazards associated with the surface water ingestion pathway. The DRAS calculates each noncarcinogenic waste constituent's *HQ* for this pathway using Equation 4-95.

$$HQ = \frac{CR \cdot C_{sw} \cdot EF \cdot ED}{BW \cdot RfD \cdot AT \cdot 365 days/year} \qquad \textbf{(4-95)}$$

where:

			Default
HQ	=	constituent-specific hazard quotient for ingestion of surface water (unitless)	calculated
CR	=	water consumption rate (L/day)	2.0 (U.S. EPA 1997b)
C_{sw}	=	concentration of constituent in surface water (mg/L)	Equation 4-45
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
RfD	=	reference dose (mg/kg-day)	chem-specific (Appendix A-1)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)

Equation 4-45, which is used to compute C_{sw}, is based on the USLE and the sediment delivery ratio for overland transport of eroded waste to surface water as described in the delisting docket (U.S. EPA 1993a). Using Equation 4-95, the DRAS calculates the *HQ* associated with ingestion of surface water for each waste constituent. The program then sums the *HQ*s using Equation 4-5 to determine the total HI_p for the surface water ingestion pathway.

4.4.2.2 Ingestion of Fish

This section describes the computation of total pathway concentration pathway delisting levels and hazards associated with exposure of adults to noncarcinogenic waste constituents via ingestion of fish. Section 4.4.2.2.1 presents the equations used to compute constituent-specific total concentration delisting levels for standard multi year delistings. Section 4.4.2.2.2 presents the equations used to compute constituent-and pathway-specific hazards for one-time delistings. The BAF values used are for a trophic level 4 (carnivorous) fish (U.S. EPA 1995e).

4.4.2.2.1 Delisting Level Analysis

This section describes the equations used to calculate the pathway total concentration delisting levels for exposure to noncarcinogenic constituents through fish ingestion, $C_{dl\text{-}fish}$. The DRAS first calculates the maximum allowable constituent concentration in surface water that will not result in a constituent concentration in fish tissue in excess of the maximum edible concentration. The maximum allowable fish tissue concentration for a waste constituent, C_{fish}, is calculated at the target hazard quotient using Equation 4-96.

$$C_{fish} = \frac{THQ \cdot RfD \cdot BW \cdot AT \cdot 365 days/year}{CR_{fish} \cdot EF \cdot ED} \tag{4-96}$$

where:

			Default
C_{fish}	=	maximum allowable concentration of constituent in fish tissue (mg/kg)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfD	=	reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
CR_{fish}	=	fish consumption rate (kg/day)	0.02 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)

Once C_{fish} is calculated, the corresponding maximum allowable concentration of the of the constituent freely dissolved in surface water is calculated using Equation 4-97.

$$C_{dw} = \frac{C_{fish}}{BAF \vee BCF} \qquad \textbf{(4-97)}$$

where:

			Default
C_{dw}	=	maximum allowable dissolved phase water concentration in the second order stream (mg/kg)	calculated
C_{fish}	=	maximum allowable concentration of constituent in fish tissue (mg/kg)	Equation 4-96
BAF	=	bioaccumulation factor of waste constituent (L/kg)	chem-specific (Appendix A-1)
or			
BCF	=	bioconcentration factor of waste constituent (L/kg)	chem-specific (Appendix A-1)

Using Equation 4-98, the maximum allowable dissolved phase water in the second order stream is converted to the total concentration of the waste constituent in water column (C_{sw}):

$$C_{sw} = C_{dw}\ [1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}] \qquad \textbf{(4-98)}$$

where:

			Default
C_{dw}	=	Dissolved phase water concentration (mg/L)	Equation 4-97
C_{sw}	=	Waste concentration in water column (mg/L)	calculated
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 2-52
1×10^{-6}		Units conversion factor (kg/mg)	

Finally, C_{sw} is used in Equation 4-99 to back calculate the delisting level ($C_{dl\text{-}fish}$) for the fish ingestion pathway.

$$C_{dl\text{-}fish} = \frac{C_{sw} \cdot Q_{2ndstream}}{A \cdot A_w} \qquad \textbf{(4-99)}$$

where:

			Default
$C_{dl\text{-}fish}$	=	pathway total concentration delisting level for fish ingestion (mg/kg)	calculated

C_{sw}	=	maximum allowable concentration in water column of constituent in 2nd order stream (mg/L)	Equation 4-98
A_w	=	rate of waste ans soil erosion from landfill (kg/[acre-year])	Equation 2-51
$Q_{2ndstream}$	=	volumetric flow of second-order stream (L/year)	3.3×10^9 (U.S. EPA 1994a)
A	=	area of waste management unit (acres)	Equation 2-12 or -13

If the total concentration of a given waste constituent exceeds its pathway total concentration delisting level, $C_{dl\text{-}fish}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.2.2.2.

4.4.2.2.2 Hazard Analysis

In the hazard analysis, the DRAS computes the hazards associated with the fish ingestion exposure pathway. The DRAS computes each noncarcinogenic waste constituent's *HQ* for this pathway using Equation 4-100.

$$HQ = \frac{C_{fish} \cdot CR_{fish} \cdot EF \cdot ED}{BW \cdot RfD \cdot AT \cdot 365 days/year} \qquad \textbf{(4-100)}$$

where:

			Default
HQ	=	hazard quotient (unitless)	calculated
C_{fish}	=	concentration of constituent in fish tissue (mg/kg)	Equation 4-51
CR_{fish}	=	fish consumption rate (kg/day)	0.02 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
BW	=	body weight (kg)	72 (U.S. EPA 1997b)
RfD	=	reference dose (mg/kg-day)	chem-specific (Appendix A-1)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)

The concentration of the waste constituent in fish tissue, C_{fish}, is computed using Equation 4-51. This equation is based on an estimate of the dissolved fraction (C_{dw}) of the waste constituent concentration present in a second-order stream. The DRAS sums the *HQ*s for the noncarcinogenic waste constituents using Equation 4-5 to determine the total HI_p for the fish ingestion pathway.

4.4.3 Air Exposure Pathways

The DRAS includes an analysis of the human health hazards associated with exposures to noncarcinogens via three air exposure pathways: (1) inhalation of particulates from solid wastes disposed of in landfills, (2) inhalation of volatiles from solid wastes disposed of in landfills, and (3) inhalation of volatiles from liquid wastes disposed of in surface impoundments. An adult receptor is assumed for all three pathways. The delisting level for the inhalation of particulates pathway is expressed in terms of the total constituent concentration in the waste. Leachate concentration delisting levels are generated for the inhalation of volatiles pathways.

The risks associated with inhalation exposure to particulate and volatile matter can typically be determined only when inhalation HBNs are available. Inhalation reference concentrations derived from oral reference doses are only used when specifically recommended by IRIS or when target organ endpoints are far from the respiratory system (U.S. EPA 2006).

The methods used to evaluate the release and transport of noncarcinogenic particulates and volatiles from waste disposal units are the same as those used for carcinogens; only the assessment calculations differ. A discussion of the source of the methods is presented in Section 4.3.3.

The following sections describe the equations and default parameters used to calculate pathway delisting levels and assess hazards associated with exposure to waste constituents via the air exposure pathways. Section 4.4.3.1 presents the equations used to compute pathway delisting levels for standard multi year delistings. Section 4.4.3.2 presents the equations used to compute constituent-specific and pathway-specific hazards for one-time delistings.

4.4.3.1 Delisting Level Analysis

This section describes the equations used to calculate air exposure pathway delisting levels for noncarcinogenic waste constituents. Before the delisting levels can be calculated, each constituent's maximum allowable respirable air concentration at the POE, C_{air}, must be calculated. C_{air} is calculated at the target hazard quotient, as shown in Equation 4-101.

$$C_{air} = \frac{THQ \cdot RfC \cdot AT \cdot 365\ days/year}{ED \cdot EF} \qquad \textbf{(4-101)}$$

where:

			Default
C_{air}	=	constituent's maximum allowable respirable air concentration at POE (mg/m^3)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfC	=	constituent reference concentration (mg/m^3)	chem-specific (Appendix A-1)
AT	=	averaging time (years)	30 (U.S. EPA 1997b)
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)

C_{air} is used with the air transport equation (Equation 2-25) and the particulate and volatile air emission equations (Equation 2-24 for particulates, and Equation 2-33 for volatiles) to back-calculate the pathway total concentration and leachate concentration delisting levels, respectively. The following sections describe the methods used to calculate delisting levels for landfill particulates ($C_{dl\text{-}air\text{-}p}$), landfill volatiles ($C_{dl\text{-}air\text{-}v}$), and surface impoundment volatiles (C_o). If the total concentration of a given noncarcinogenic waste constituent exceeds $C_{dl\text{-}air\text{-}p}$, or the TCLP concentration of a constituent exceeds either $C_{dl\text{-}air\text{-}v}$ or C_o, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.3.3.2.

Landfill Particulate Delisting Levels

First the maximum allowable emission rate of the constituent, Q_p, is calculated using in C_{air} using Equation 4-102.

$$Q_p = \frac{C_{air} \cdot \Sigma_z \cdot U \cdot L_v}{2.03 \cdot F \cdot F_{inhal}} \qquad \textbf{(4-102)}$$

where:

			Default
Q_p	=	maximum allowable total constituent emission rate (mg/s)	calculated
C_{air}	=	constituent's maximum allowable air concentration at POE (mg/m^3)	Equation 4-101
$\sum_z$	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
F	=	frequency that wind blows across waste unit toward receptor (unitless)	0.15 (U.S. EPA 1994a)
F_{inhal}	=	fraction of particles inhaled	0.125 (U.S. EPA 1994a)

Then the pathway total concentration delisting level for respirable landfill in air particulates, $C_{dl\text{-}air\text{-}p}$, is calculated by setting the maximum allowable total emission rate for respirable particulates, Q_{p10}, as equal to Q_p and using Equation 4-103.

$$C_{dl-air-p} = \frac{Q_{p10}}{(E_{w10} + E_{v10} + E_{l10}) \cdot \frac{10^3\ mg}{1\ g} \cdot \frac{1\ hr}{3{,}600\ s} \cdot \frac{10^{-6}\ kg}{1\ mg}} \qquad \textbf{(4-103)}$$

where:

			Default
$C_{dl\text{-}air\text{-}p}$	=	pathway total concentration delisting level for respirable landfill particulates in air (mg/kg)	calculated
Q_{p10}	=	emission rate of waste constituent particulates up to 10 ì m (mg/s)	set equal to Q_p (Equation 4-102)
E_{w10}	=	wind erosion emission rate of particulates up to 10 ì m (g/hr)	Equation 2-9
E_{v10}	=	vehicle travel emission rate of particulates up to 10 ì m (g/hr)	Equation 2-19
E_{l10}	=	waste loading and unloading emission rate of particulates up to 10 ì m (g/hr)	Equations 2-21 and 2-22

Landfill Volatile Delisting Levels

First the maximum allowable volatile emission flux of the constituent, E_i, from the landfill is calculated by setting the maximum allowable total emission rate of volatiles, Q_v, as equal to Q_p (calculated in Equation 4-102) and using Equation 4-104.

$$E_i = \frac{Q_v}{1000\ mg\ /\ g} \qquad \textbf{(4-104)}$$

where:

			Default
E_i	=	maximum allowable volatile emission flux of constituent (g/s)	calculated
Q_v	=	maximum allowable constituent volatile emission rate (mg/s)	set equal to Q_p (Equation 4-102)
1000mg /g		conversion factor	

Then, E_i is used to calculate the pathway total delisting level concentration for volatiles from a landfill, $C_{dl\text{-}air\text{-}v}$, using Equation 4-105.

$$C_{dl-air-v} = \frac{E_i \cdot d \cdot P_T^2 \cdot 1x10^6 \cdot 1x10^4}{D_a \cdot P_a^{10/3} \cdot A_{exposed} \cdot C_{si}} \qquad \textbf{(4-105)}$$

where:

			Default
$C_{dl\text{-}air\text{-}v}$	=	pathway total waste concentration delisting level for volatiles from landfill (mg/kg)	calculated
E_i	=	maximum allowable volatile emission flux of constituent (g/s)	Equation 4-55
d	=	depth of soil cover (m)	0.1524 (U.S. EPA 1994a)
P_T	=	total sand porosity (unitless)	0.4 (U.S. EPA 1994a)
D_a	=	diffusivity of constituent in air (cm^2/s)	chem-specific (Appendix A-1)
P_a	=	air-filled sand porosity (unitless)	0.4 (U.S. EPA 1994a)
$A_{exposed}$	=	surface area (m^2)	Equation 2-15
C_{si}	=	saturation vapor concentration of *i* in landfill (g/m^3)	Equation 4-106
$1x10^6$	=	conversion from kilograms to milligrams	
$1x10^4$	=	conversion from m^2 to cm^2	

Finally, the saturation vapor concentration (C_{si}) of waste constituent *i* for volatiles from a landfill ($C_{dl\text{-}air\text{-}v}$) is calculated using Equation 4-106.

$$C_{si} = \frac{p_i \cdot MW_i}{R \cdot T} \qquad \textbf{(4-106)}$$

where:

			Default
C_{si}	=	saturation vapor concentration of *i* in landfill (g/m^3)	Equation 4-56
p_i	=	vapor pressure of waste constituent *i* (at)	chem-specific (Appendix A-1)
MW	=	molecular weight of waste constituent *i*	chem-specific (Appendix A-1)
R	=	universal gas constant (atm-m^3/mol- K)	8.21×10^{-5}
T	=	standard temperature (K)	298

Surface Impoundment Volatile Delisting Levels

The pathway leachate concentration delisting level for volatiles from waste disposed of in a surface impoundment is calculated by setting the maximum allowable total emission rate of volatiles, Q_v, as equal to Q_p (calculated in Equation 4-102) and using Equation 4-107.

$$C_{dl-air-si} = \frac{Q_{pv} \cdot t_f \cdot \frac{86{,}400\ s}{1\ day}}{V_{si} \cdot \frac{10^3 L}{1\ m^3} \cdot \exp\left[\frac{-K \cdot A \cdot \frac{4{,}046.8\ m^2}{1\ acre} \cdot t_r \cdot \frac{86{,}400\ s}{1\ day}}{V_{si}}\right]} \qquad \textbf{(4-107)}$$

where:

			Default
$C_{dl-air-si}$	=	pathway leachate concentration delisting level for volatiles from surface impoundment (mg/L)	calculated
Q_v	=	maximum allowable constituent volatile emission rate (mg/s)	set equal to Q_p (Equation 4-102)
t_f	=	time for constituent concentration to reach 1 percent of C_o (days)	Equation 4-59
V_{si}	=	volume of liquid in surface impoundment (m^3)	unit-specific (user- provided)
K	=	overall mass transfer coefficient (m/s)	Equation 4-60
A	=	area of waste management unit (acres)	Equation 2-12 or 2-13
t_r	=	retention time for liquid in surface impoundment (days)	Equation 2-40

4.3.3.2 Hazard Analysis

Hazard as a function of the effective air concentration of a particulate or volatile noncarcinogenic constituents, C_{air}, is calculated for each of the three air exposure pathways as shown in Equation 4-108.

$$HQ = \frac{C_{air} \cdot ED \cdot EF}{RfC \cdot AT \cdot 365 days/year} \qquad \textbf{(4-108)}$$

where:

			Default
HQ	=	hazard quotient for waste constituent (unitless)	calculated
C_{air}	=	effective concentration of constituent in air at POE (mg/m^3)	
		Particulates	Equation 4-66
		Landfill volatiles	Equation 4-68
		Surface impoundment volatiles	Equation 4-73
ED	=	exposure duration (years)	30 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
RfC	=	constituent reference concentration (mg/m^3)	Chem-specific (Appendix A-1)

AT = averaging time (years) 30 (U.S. EPA 1997b)

C_{air} at the POE is determined separately for each of the three air exposure pathways using the air emission and transport equations derived in Section 2.3 and presented in Section 4.3.3.2.

Using Equation 4-108, the DRAS calculates the waste constituent HQs for receptors at the POE for the three air exposure pathways. The program then sums the individual constituent HQs using Equation 4-5 to determine the HI_p for each of the air inhalation pathways.

4.4.4 Soil Exposure Pathway

This section presents equations used to calculate pathway total concentration delisting levels and hazards constituents associated with exposures of children to soils contaminated with noncarcinogenic waste constituents at a POE 1,000 feet from a waste disposal unit. The soil contamination is assumed to be the result deposition of waste particulates from the air. The equations used to compute the delisting levels and hazards are based on a method described in U.S. EPA Headquarters delisting docket materials (U.S. EPA 1993a, 1993b).

Section 4.4.4.1 presents the equations used to compute pathway total concentration delisting levels for standard multi year delistings. Section 4.4.4.2 presents the equations and methods used to compute constituent-specific and pathway- hazards for one-time delistings.

4.4.4.1 Delisting Level Analysis

This section describes computation of the pathway total concentration delisting level for exposure to noncarcinogens via ingestion of soil, $C_{dl\text{-}soil}$. First, the maximum allowable concentration of each constituent in soil at the POE, C_{soil}, is calculated. C_{soil} is the maximum concentration of the waste constituent in soil that does not cause the risk from soil ingestion to exceed the target hazard quotient. C_{soil} is calculated using Equation 4-109.

$$C_{soil} = \frac{THQ \cdot RfD \cdot BW_c \cdot AT \cdot 365 days/year}{CR_{soil} \cdot ED \cdot EF} \cdot 1x10^6 mg/kg \qquad \textbf{(4-109)}$$

where:

Default

C_{soil}	=	maximum allowable concentration of constituent in soil at the POE (mg/kg)	calculated
THQ	=	target hazard quotient (unitless)	user specified (0.1 default U.S. EPA 1997b)
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)
BW_c	=	body weight of child (kg)	15 (U.S. EPA 1997b)
AT	=	averaging time (years)	6 (U.S. EPA 1997b)
CR_{soil}	=	soil consumption rate (mg/day)	200 (U.S. EPA 1997b)
ED	=	exposure duration (years)	6 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)

Using C_{soil}, the DRAS back-calculates the maximum allowable constituent concentration in air at the POE, C_{air}, assuming a constituent accumulation period in the soil of 1 year. C_{air} is calculated, at a depth of mixing of 1 cm as shown in Equation 4-110.

$$C_{air} = \frac{\rho_b \cdot t \cdot C_{soil} \cdot \frac{10^3\ mg}{1\ g} \cdot \frac{10^6\ cm^3}{1\ m^3} \cdot \frac{1\ kg}{10^6\ mg}}{v_d \cdot \frac{31{,}536{,}000\ s}{1\ year}} \cdot 1/year \tag{4-110}$$

where:

			Default
C_{air}	=	maximum allowable concentration of constituent in air at POE (mg/m^3)	calculated
ñ$_b$	=	soil bulk density (g/cm^3)	1.45 (Brady 1984)
t	=	soil thickness from which particles can be ingested (m)	0.01 (U.S. EPA 1994a)
C_{soil}	=	maximum allowable concentration of constituent in soil at the POE (mg/kg)	Equation 4-109
v_d	=	deposition velocity (m/s)	0.03 (U.S. DOE, 1984)

Finally, once C_{air} is known, the total concentration delisting level is computed using Equation 4-111.

$$C_{dl-soil} = \frac{C_{air} \cdot \Sigma_z \cdot U \cdot L_v \cdot \frac{1{,}000\, g}{1\, kg}}{2.03 \cdot E_{T30} \cdot F \cdot \frac{1\, hr}{3{,}600\, s}} \tag{4-111}$$

where:

			Default
$C_{dl\text{-}soil}$	=	pathway total concentration delisting level for soil ingestion (mg/kg)	calculated

C_{air}	=	maximum allowable concentration of constituent in air at POE (mg/m^3)	Equation 4-110
Σ_z	=	vertical dispersion coefficient (m)	Equation 2-26
U	=	mean annual wind speed (m/s)	4 (U.S. EPA 1994a)
L_v	=	distance from virtual point to compliance point located 1,000 feet (304.8 m) downwind (m)	Equation 2-27
E_{T30}	=	annual average emission rate of particles up to 30 ì m (g/hr)	Equation 4-75 (also see Section 2.3.1.3)
F	=	frequency that wind blows across waste unit toward receptor (unitless)	0.15 (U.S. EPA 1994a)

If the total concentration for a given noncarcinogenic waste constituent exceeds its pathway total concentration delisting level, $C_{dl\text{-}soil}$, the waste may not qualify for a standard multi year delisting. However, the waste may qualify for a one-time delisting, depending on the results of hazard analysis calculations presented in Section 4.4.4.2.

4.4.4.2 Hazard Analysis

In the hazard analysis, the DRAS computes the hazard associated with the soil ingestion exposure pathway. The hazard for each noncarcinogenic waste constituent associated with soil ingestion at the POE is calculated. Only the child receptor is considered. The focus on children reflects the higher daily soil intake rates for children and their lower body weights. The constituent-specific hazard from ingestion of soil is calculated using Equation 4-112.

$$HQ = \frac{C_{soil} \cdot CR_{soil} \cdot ED \cdot EF}{BW_c \cdot RfD \cdot AT \cdot 365 days/year \cdot 1x10^6 mg/kg} \qquad \textbf{(4-112)}$$

where:

			Default
HQ	=	constituent hazard quotient (unitless)	calculated
C_{soil}	=	concentration of constituent in soil POE (mg/kg)	Equation 4-78
CR_{soil}	=	soil consumption rate (mg/day)	200 (U.S. EPA 1997b)
ED	=	exposure duration (years)	6 (U.S. EPA 1997b)
EF	=	exposure frequency (days/year)	350 (U.S. EPA 1997b)
BW_c	=	body weight of child (kg)	15 (U.S. EPA 1997b)
RfD	=	constituent reference dose (mg/kg-day)	chem-specific (Appendix A-1)
AT	=	averaging time (years)	6 (U.S. EPA 1997b)

C_{soil} is calculated based on the constituent's rate of deposition onto the soil. Assuming that at any given time the soil is burdened with 1 year's accumulation of constituent from air deposition and that the mixing depth of the constituent into the soil is 1 cm, Equation 4-78 is used to calculate C_{soil}. Then, using Equation 4-112, the DRAS calculates the HQ associated with ingestion of soil for each constituent. Finally this program sums the constituent-specific HQs using Equation 4-5 to determine the total HI_p for the soil ingestion pathway.

Chapter 5
Uncertainty

What's covered in Chapter 5:

- ♦ Uncertainty in the Delisting Risk-Based Process
- ♦ Types of Uncertainty
- ♦ Uncertainty in Sample

It is U.S. EPA policy that stakeholders in environmental issues be provided with sufficient information to allow them to independently assess environmental risks and the reasonableness of risk reduction actions (U.S. EPA 1995c). Furthermore, all risk characterizations should exhibit transparency, clarity, consistency, and reasonableness (U.S. EPA 1995c). To ensure that all risk assessments exhibit these qualities, the U.S. EPA Administrator has specified two requirements that must be met when characterizing risk: (1) addressing qualitative and quantitative features of the risk assessment and (2) identifying uncertainties as a measure of the confidence in the assessment. U.S. EPA Region 6 intends to meet these requirements in its risk assessments of wastes petitioned for RCRA exclusion (delisting) under 40 CFR 260.20 and 260.22. This chapter identifies and discusses the uncertainties associated with delisting risk assessments, including the uncertainty involved in the delisting risk-based process, types of uncertainty associated with risk characterization, and uncertainties associated with sample analyses used to characterize wastes petitioned for delisting.

Uncertainty is inherent in the delisting process and can be introduced into a risk assessment at each stage of the process outlined in this DTSD. For delisting, uncertainty can generally be classified in terms of sampling and nonsampling errors with regard to the following:

- Determining the total and leachable concentrations of waste constituents
- Estimating the release of pollutants from a waste management unit to the environment
- Predicting and transport of pollutants in a range of variable environments by processes that often are not completely understood or are too complex to quantify accurately

- Estimating the potential for adverse health effects in humans based on animal studies
- Estimating the probability of adverse effects on a human population that is highly variable in terms of genetic predisposition, age, activity level, and overall health.

Uncertainty is inherent in the process even if the most accurate data are used in the most sophisticated models. The methodology outlined in this document relies on a combination of probabilistic and point values—some conservative and some representing central tendencies. These values yield a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk. For example, to develop the waste-volume adjusted DAFs, U.S. EPA Region 6 runs the EPACMTP in Monte Carlo mode, selecting input parameter values from frequency distributions of each mode input parameter to perform thousands of simulations that result in one 90th percentile DAF value. This value is used in combination with point values of the ingestion, dermal absorption, and shower inhalation risk algorithms to derive a final point estimate of risk or a single delisting level. For this reason, while the degree of conservatism in risk estimates is not fully known; the values combine many conservative factors and are likely to overstate actual risk (Hattis and Burmaster 1994). This chapter discusses types of uncertainty, areas where uncertainty can be introduced into a risk assessment, and methods for qualitatively and quantitatively addressing uncertainty in risk assessments.

To describe the uncertainties associated with a delisting evaluation, this chapter discusses the following: (1) uncertainty inherent in the delisting evaluation process, (2) types of uncertainty, and (3) uncertainties specifically associated with sampling and analysis of petitioned waste.

5.1 UNCERTAINTY IN THE DELISTING RISK-BASED PROCESS

To qualify for exclusion from Subtitle C requirements, in addition to showing that a waste does not exceed target risks, a petitioner must demonstrate that a waste generated at a facility does not meet any of the criteria for which the wastes was listed (see 40 CFR 260.22(a) and associated background documents for the listed wastes). In addition, the HSWA of 1984 require that U.S. EPA consider any factors (including additional constituents) other than those for which the waste was listed if there is reason to believe that such additional factors could cause the waste to be hazardous. Accordingly, a petitioner also must (1) demonstrate that the waste does not exhibit any of the hazardous waste characteristics (ignitability, reactivity, corrosivity, and toxicity) and (2) present sufficient information for U.S. EPA to determine whether the waste contains other toxicants at hazardous levels (see 40 CFR 260.22(a), 42 [USC] 6921(f), and the background documents for the listed wastes). Although wastes that are "delisted" or excluded have been evaluated to determine whether they exhibit any characteristics of hazardous waste, a generator remains obligated under RCRA to determine whether its waste remains nonhazardous.

If U.S. EPA grants an exclusion for a petitioned waste, the delisted waste is no longer controlled under federal hazardous waste regulations. The petitioner must then manage and dispose of the waste in accordance with local and state requirements or specifications. In some cases, U.S. EPA grants an exclusion on the condition that a facility or waste meet certain requirements. For example, for a waste that is highly variable in composition, U.S. EPA often imposes postexclusion testing requirements that the petitioner must meet prior to waste disposal. Only those batches of waste that have met the verification testing conditions provided in the final exclusion can be managed as nonhazardous waste; batches that fail to meet the verification testing conditions must be managed as hazardous waste.

To demonstrate that wastes do not meet any of the criteria for which the wastes were listed, U.S. EPA Region 6 uses risk assessment algorithms for the disposal, release, and exposure scenarios (see Chapters 2, 3, and 4). The uncertainties inherent in the use of the algorithms and the uncertainties in characterizing the disposal, release and exposure scenarios are described in the following sections.

5.1.1 Use of Risk Assessment Algorithms for Delisting Evaluations

During a delisting determination for a petitioned waste, U.S. EPA uses fate and transport models and risk assessment algorithms to (1) predict the concentrations of hazardous constituents that may be released from the petitioned waste after disposal, (2) determine the means by which receptors would be exposed to the released constituents and (3) estimate the risks and hazards that such exposures would impose on the receptors. Appropriate fate and transport models are used to estimate the potential for leachable hazardous constituents to be released to the underlying aquifer and also to estimate the potential for release of waste constituents to air and surface water. Chapter 2 outlines the methodology used to generate waste constituent concentration values that receptors would be exposed to at a defined POE.

Specifically, U.S. EPA uses acceptable health-based levels for hazardous constituents of concern at a specified POE. These levels, also known as toxicity factors, include RfDs, RfCs, Unit Risk Factors (URFs), and CSFs. These health-based levels are applied in addition to U.S. EPA's acceptable risk range of 1×10^{-6} to 1×10^{-4} for known or suspected carcinogens and an HQ of less than 1.0 for noncarcinogens (U.S. EPA 1990g). To determine the allowable concentrations (or the delisting levels) for specific waste constituents in standard multiyear delistings, the target risk level for Region 6 is based on a 1×10^{-5} risk level for carcinogens an HQ of 0.1 for noncarcinogens. Other regions or delisting authorities may assign slightly different target risk levels. A waste constituent's delisting level is the maximum allowable total and leachate concentrations that a petitioned waste can contain at the petition-specified annual waste volume. In multiyear RCRA delistings, an order of magnitude safety factor is applied to carcinogens and noncarcinogens in order to account for possible multiple exposure pathways and multiple waste constituent exposures over time.

The delisting level for a multiyear delisting can be back-calculated by applying the acceptable health-based level at the target risk level. If the total or leachate concentration of a constituent in a petitioned waste is greater than the back-calculated delisting level then the petitioned waste is deemed to have failed the risk evaluation and is considered to have the potential to adversely affect human health and the environment. Conversely, if the waste constituent total and leachate concentrations are less than the corresponding maximum allowable levels then the waste remains a candidate for a standard multiyear delisting.

To estimate the total aggregate risk and hazard for a petitioned waste, U.S. EPA Region 6 uses the maximum waste volume, the maximum total waste constituent concentrations and the maximum leachate concentrations. The aggregate risk and hazard for a petitioned waste can be useful for a one-time delisting where establishing more conservative delisting levels is not necessary. Toxicological determination for the delisting can be performed at the lower end of the risk range for several reasons. First, because the future variability of additional batches of the wastes does not have to be accounted for, delisting levels need not be established. Second, the risk assessment is aggregate and assumes that the receptor is exposed to all waste constituents at the maximum concentrations via each exposure pathway; however, this is a very conservative assumption and is not likely to occur. Therefore, for a one-time delisting, if the computed aggregate risk and hazard for a petitioned waste do not exceed the cutoff risk level of 1×10^{-4} and HI of 1.0, the waste may qualify in Region 6 as a candidate for a RCRA delisting. Other Regions and delisting authorities may chose to set a different cutoff risk level for one-time delistings.

5.1.2 Disposal, Release, and Exposure Scenarios

Fate and transport models are used to determine the maximum allowable waste constituent concentrations and to compute aggregate risk and hazard. U.S. EPA's approach in using such models has been to represent a reasonable worst-case waste disposal scenario for the petitioned waste rather than to rely on site-specific factors. U.S. EPA believes that a reasonable worst-case scenario is appropriate when determining whether a waste should no longer be managed under RCRA Subtitle C. The use of a reasonable worst-case scenario results in conservative values for the point compliance concentrations and ensures that the waste, regulated as hazardous, will not pose a threat to human health or the environment if the petitioner chooses to dispose of the waste in accordance with Subtitle D requirements. Site-specific factors (for example, site hydrogeology) are not considered in the risk assessment because a delisted waste is no longer subject to hazardous waste control and may be disposed of in any Subtitle D landfill or surface impoundment. Therefore, conservative default parameters are used to predict reasonable worst-case scenarios for waste disposal in any Subtitle D landfill or surface impoundment.

5.2 TYPES OF UNCERTAINTY

Uncertainty can be classified as one of four types: (1) parameter uncertainty, (2) model uncertainty (that is, does the model accurately represent and simulate conditions that may exist at a waste disposal site?), (3) decision rule uncertainty and (4) variability (Finkel 1990).

Parameter Uncertainty

Parameter uncertainty arises when parameters used in equations cannot be measured precisely or accurately because of either (1) equipment limitations or (2) spatial or temporal variances between the quantities being measured. Random or sample errors are common sources of parameter uncertainty, especially for small sample sizes. However, it is more difficult to recognize nonrandom or systemic errors that result from sampling, the experimental design, or the choice of assumptions. Examples of parameter uncertainty include uncertainty in waste characterization data and uncertainty regarding the data and input parameters used in the release and exposure algorithms. Uncertainty associated with waste characterization data may arise from the uncertainty inherent in measurement of a waste volume petitioned for delisting, whether it is estimated or measured. Uncertainty also arises with regard to waste constituent data, especially if (1) chemicals are present in the waste that are not listed as chemicals of concern, (2) chemicals are present at less than analytical detection levels, or (3) the measured concentrations are not representative of the entire waste being petitioned for delisting. In addition, the leachable concentration of a chemical is estimated with the TCLP test, which is only a laboratory approximation of what will actually happen in a landfill scenario.

Model Uncertainty

Uncertainty is associated with models used in all phases of a risk assessment, including (1) animal models used as surrogates to test human health effects, (2) dose-response models used in extrapolations, and (3) computer models used to predict the fate and transport of chemicals in the environment. Use of rodents as surrogates for humans introduces uncertainty due to the considerable interspecies variability in sensitivity. In addition, computer models are simplifications of reality and may exclude some variables that influence predictions but cannot be included in models because of (1) increased model complexity or (2) lack of data. The risk assessor and modeler should consider the importance of excluded variables on a case-by-case basis because a specific variable may significantly affect uncertainty in some instances and not in others. A similar problem can occur when a model that is appropriate for use under average conditions is used to model conditions that significantly differ from the average. For many situations, choosing the correct model can be difficult because conflicting theories may appear to explain a specific phenomenon equally well.

The models used by U.S. EPA Region 6 that are described in this document were selected on the basis of scientific policy. The EPACMTP groundwater transport model was selected because it provides the information needed to conduct risk assessments and is considered by U.S. EPA to be a state-of-the-science model that has undergone review by independent peer review panels and the U.S. EPA Science Advisory Board. This model was developed in the context of setting national regulatory levels and as such attempts to account for the uncertainty that may be encountered during modeling of contaminant transport in groundwater nationwide. As with all models, the simplifying assumptions used to implement the EPACMTP can introduce uncertainty with regard to the models simulation of conditions at a specific sites.

Other fate and transport models recommended by this guidance may also introduce uncertainty. For example, the model used to estimate chemical concentrations in surface water bodies may be particularly conservative because it does not consider chemical decay or sorption processes. Downstream dilution of chemical concentrations in water and sediment is not considered in a delisting risk assessment; therefore, its results will likely be conservative for chemicals in surface water.

Decision Rule Uncertainly

The uncertainty associated with risk analysis influences many policy and risk management decisions. Decision rule uncertainty (that is, the uncertainty introduced in the determination of appropriate risk levels) is probably of greatest concern to risk managers. This type of uncertainty arises, for example, out of the need to balance different social concerns when determining an acceptable level of risk. Possibly the most important aspect of the risk assessment is the selection of constituents to be included in the analysis.

A second area of decision rule uncertainty involves use of standard U.S. EPA default values in the risk analysis. Standard default values used in most U.S. EPA risk assessments include inhalation and consumption rates, body weight, and lifespan. Inhalation and consumption rates are highly correlated to body weight for adults. Using a single-point estimate for these variables instead of a joint probability distribution ignores the variability that may influence the risk assessment results by a factor of as much as two or three.

A third area of decision rule uncertainty involves use of U.S. EPA-verified CSFs, RfDs, and RfCs. These health benchmarks are used as single-point estimates throughout the risk analysis, introducing both uncertainty and variability. However, U.S. EPA has developed a process for setting verified health benchmark values to be used in all U.S. EPA risk assessments. Exception for the dioxin toxicity equivalency methodology, all health benchmarks recommended for use in all analyses are verified by U.S. EPA work groups and are available in IRIS. The information in IRIS is now maintained on the U.S. EPA web site at "http://www.epa.gov/ngispgm3/iris/index.html/". This DTSD does not estimate the uncertainty associated with using U.S. EPA-verified health benchmarks or the dioxin toxicity equivalency methodology.

5.2.1 Variability

"Variability" is often used interchangeably with the term "uncertainty," but the two terms have specific differences. Variability may be tied to variations in physical and biological processes, and it cannot be reduced with additional research or information; however, it may be known with greater certainty (for example, the age distribution of a population may be known and represented by the mean age and its standard deviation). Uncertainty is a description of the imperfect knowledge of the true value of a particular variable or its real variability in an individual or group. In general, uncertainty can be reduced through additional information gathering or analysis activities (that is, with better data or better models); additional data will not change real variability, although it may be more accurately known (Hattis and Burmaster 1994).

5.3 UNCERTAINTY IN SAMPLE ANALYSES

Based on lessons learned from previous U.S. EPA OSW delisting decisions, U.S. EPA Region 6 recognizes that a significant amount of uncertainty may result from (1) the sample analysis method used to analyze a petitioned waste and (2) the use of a delisted material. The uncertainties associated with these two aspects of delisting risk characterizations are discussed in this section.

5.3.1 The Toxicity Characteristic Leaching Procedure (TCLP)

The TCLP is a test method designed to simulate the leaching that a waste will undergo when the waste is disposed of in a sanitary landfill. This test is designed to determine the mobility of both organic and inorganic analytes present in liquid, solid, and multiphasic wastes. When performed as prescribed in Appendix II of 40 CFR Part 261, Method 1311, an analysis of any liquid fraction of the TCLP extract will indicate whether a regulated waste constituent is present in the waste. The test method is used to determine if, even after accounting for dilution from the other fractions of the extract, the concentration of the waste constituent exceeds the regulatory level allowed in that media. If the TCLP test determines that the waste constituent concentration exceeds allowable levels, as indicated by a risk assessment, the petitioned waste is hazardous and may not be delisted under 40 CFR 260.20 and 260.22.

Limitations in the use of the TCLP have been identified, specifically in the Reynolds Metals Company (RMC) delisting (U.S. EPA 1991b). Problems associated with the RMC delisting petition resulted from a combination of phenomena not accounted for in the assumptions of the TCLP. As described in the November 15, 1996, “Hazardous Waste Characteristics Scoping Study,” the high alkalinity of some wastes may make the TCLP an inappropriate predictor of leachate composition. Very basic or high pH wastes contain enough hydroxyl ion to release certain other analytes to the leachate. In high pH ranges, certain waste constituents are quite soluble (that is, they have a low K_d), which makes consideration of the dilution factor very important.

Another potential problem highlighted by the RMC delisting petition involves the disposal scenario. The TCLP assumes that the waste will be commingled with 95 percent industrial waste in a municipal landfill. The TCLP, however, does not account for waste disposed of in a monofill. The TCLP assumptions are also inadequate with regard to degradation and dilution factors. For example, the TCLP uses a liquid to solid ratio of 20:1; however, the liquid to solid ratio in the RMC monofill is 0.09-0.15:1. This range of liquid to solid ratios cannot be simulated in bench-scale tests. Consequently, TCLP tests that use a liquid to solid ratio of 20:1 would dilute results for the RMC monofill. U.S. EPA Region 6 conducted limited experiments that varied the liquid to solid ratios and pH of the extraction media (U.S. EPA 1997). These limited experiments indicated that the pH and liquid to solid ratios significantly alter the TCLP leachate concentrations.

Based on results of analyses performed for the RMC delisting, it is reasonable to conclude that the TCLP is not always an adequate assessing a petitioned waste in terms of leaching considerations and associated risk. U. S. EPA’s regional offices continue to investigate the TCLP and acknowledge that further testing must be conducted U.S. EPA Region 6 intends to supplement the TCLP with additional tests for waste disposed of in a monofill. Supplemental testing may include the Synthetic Acid Precipitation Leaching Procedure, the Multiple Extraction Procedure, and column tests similar to those developed by RMC in order to predict the leaching potential of a monofill. Selection of an appropriate additional test should consider two primary goals: (1) the test should adequately predict the potential leachate concentrations of the waste, and (2) the test should approximate the volume of rainfall that the waste will be exposed to. Using a total solids concentration for a given waste constituent in place of leachate testing is not recommended.

5.3.2 Use of a Delisted Material

Under the U. S. EPA Region 6 Delisting Program, the EPACMTP has been adapted for use in predicting groundwater impacts for two Subtitle D disposal scenarios: (1) landfill and (2) surface impoundment. One assumption of the EPACMTP is that these disposal scenarios represent reasonable worst-case scenarios for waste that will be delisted. However, use of the waste for road bed material, fill material or other purposes has not been modeled in this delisting risk-based approach. In addition, the model does not evaluate waste disposal in a monofill, use of waste as fill material in a mine reclamation project or other disposal scenarios.

To avoid problems encountered in the RMC delisting or similar problems in future delisting decisions, U.S. EPA Region 6 has removed the discussion of beneficial use and recycling from the boilerplate language of the FR notice. A waste will be delisted only if it meets the criteria for which it was modeled. After an exclusion has been granted, if a facility finds an additional use for the waste or determines that the waste can be recycled and reused, the facility must submit appropriate information to U.S. EPA for evaluation before the waste can be delisted and used for the proposed purpose.

In addition, U.S. EPA Region 6 has adopted language from the Conversion Systems, Inc., delisting exclusion (U.S. EPA 1993a) that limits disposal of a delisted waste to the modeled landfill or surface impoundment. If a facility can provide a model for scenarios that are not addressed by the EPACMTP or the surface pathway modeling default parameters, U. S. EPA will evaluate that model to ensure that the appropriate pathways and assumptions have been considered before allowing the waste to be delisted.

References

Allison, J.D., D.S. Brown, and K.J. Novo-Gradac. 1991. *MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems: Version 3.0 User's Manual*. EPA/600/3-91/021. U.S. EPA Office of Research and Development, Environmental Research Laboratory, Athens, GA. March.

ATSDR. 1995. *Toxicological Profile for Polychlorinated Biphenyls (Update)*. Draft for Public Comment. Prepared by: Research Triangle Institute. Prepared for: U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. August.

Brady, N.C. 1984. *The Nature and Properties of Soils.* Ninth Edition. Collier MacMillan Publishers.

Carsel, Robert F., Rudolph S. Parrish, Russell L. Jones, James L. Hansen, and Richard L. Lamb. 1988. Characterizing the uncertainty of pesticide leaching in agricultural soils. *Journal of Contaminant Hydrology*, 2: 111-124.

Carsel, Robert F., and Rudolph S. Parrish, 1988. Developing joint probability distributions of soil water retention characteristics. *Water Resource Research*. 29:755-770.

Electric Power Research Institute, 1985. A review of Field Scale Physical Solute Transport Processes in Saturated and Unsaturated Porous Media. EPRI EA-4190, Project 2485-5, Palo Alto, California.

Farmer, W.J., M.S. Yang, and J. Letey. 1978. "Land Disposal of Hexachlorobenzene Wastes: Controlling Vapor Movement in Soils." In: *Land Disposal of Hazardous Wastes, Proceedings of the Fourth Annual Research Symposium*. Held at San Antonio, TX on March 6, 7, and 8, 1978. EPA-600/9-78-016. U.S. EPA Office of Research and Development, Municipal Environmental Research Laboratory, Cincinnati, OH. August.

Gelhar, Lynn W., 1985. Personal Communication.

Gelhar, Lynn W., Claire Welty, Kenneth R. Rehfeldt, 1992. A Critical review of data on field-scale dispersion in aquifers. *Water Resources Research*, 28(7), 1955-1974.

Hattis, D., and D.E. Burmaster. 1994. "Assessment of Variability and Uncertainty Distributions for Practical Risk Analyses." *Risk Analysis*, 14(5): 713-730.

Keup, L.E. 1985. "Flowing Water Resources." *Water Resources Bulletin*, 21(2): 291-296. April.

McKone, T.E. 1987. "Human Exposure to Volatile Organic Compounds in Household Tap Water: The Indoor Inhalation Pathway." *Environmental Science and Technology*, 21(12): 1194-1201.

McKone, T.E., and K.T. Bogen. 1992. "Uncertainties in Health-Risk Assessment: An Integrated Case Study Based on Tetrachloroethylene in California Groundwater." *Regulatory Toxicology and Pharmacology*, 15: 86-103.

Mills, W.B., J.D. Dean, D.B. Porcella, S.A. Gherini, R.J.M. Hudson, W.E. Frick, G.L. Rupp, and G.L. Bowie. 1982. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants, Part 1*. EPA-600/6-82-004a. U.S. EPA Office of Research and Development, Environmental Research Laboratory, Athens, GA. September.

National Research Council. 1983. *Risk Assessment in the Federal Government: Managing the Process*. National Academy Press, Washington, D.C.

Newell, Charles J., Loren P. Hopkins, and Philip B. Bedient, 1989. *Hydrogeologic Database for Ground Water Modeling*. API Publication No. 4476, American Petroleum Institute, Washington, DC.

Newell, Charles J., Loren P. Hopkins, and Philip B. Bedient, 1990. "Hydrogeologic Database for Ground Water Modeling." *Ground Water*, Vol. 28, No. 5, pp. 703 - 714.

North Carolina Department of Health, Environment, and Natural Resources (NC DHENR). 1997. *North Carolina Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. Prepared by: Research Triangle Institute, Center for Environmental Analysis, Research Triangle Park, NC. Prepared for: State of North Carolina, Division of Waste Management. January.

Shen, Thomas T. 1980. *Emission Estimation of Hazardous Organic Compounds from Waste Disposal Sites*. Presented at the 73rd Annual Meeting of the Air Pollution Control Association, Montreal, Quebec. June 22-27.

Suter II, G.W. and C.L. Tsao. 1996. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*. ES/ER/TM-96/R2. Prepared by: Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN. Prepared for: U.S. DOE, Office of Environmental Management. Oak Ridge National Laboratory, Oak Ridge, TN. June

U.S. Department of Energy. 1984. *Atmospheric Science and Power Production*. DOE/TIC-27601 (DE84005177). Office of Scientific and Technical Information, Technical Information Center, Oak Ridge, TN.

U.S. EPA.1984. *Evaluation and Selection of Models for Estimating Air Emissions from Hazardous Waste Treatment, Storage, and Disposal Facilities*. EPA-450/3-84-020. Office of Air Quality Planning and Standards, Research Triangle Park, NC. December.

U.S. EPA.1985a. *Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites*. EPA/600/8-85/002. Office of Research and Development, Office of Health and Environmental Assessment, Washington, D.C. February.

U.S. EPA.1985b. *Compilation of Air Pollutant Emission Factors, Volume I: Stationary Point and Area Sources*. Fourth Edition. EPA Publication AP-42. Office of Air and Radiation, Office of Air Quality Planning and Standards, Research Triangle Park, NC. September.

U.S. EPA.1985c. "Transport to the Air - Ambient Air Dispersion Model (AADM)." In: "Hazardous Waste Management System: Identification and Listing of Hazardous Waste; Proposed Exclusions and Proposed Organics Model." *Federal Register*, 50(229): 48963-48967. November 27.

U.S. EPA. 1986a. *Industrial Source Complex (ISC) Dispersion Model User's Guide - Second Edition Volume I*. EPA-450/4-86-005a. Office of Air and Radiation, Office of Air Quality Planning and Standards, Research Triangle Park, NC. June.

U.S. EPA. 1986b. *Superfund Exposure Assessment Manual*. Draft. OSWER Directive 9285.5-1. Submitted to: Office of Emergency and Remedial Response, Washington, D.C. January 14.

U.S. EPA. 1988a. *Background Document on Subsurface Fate and Transport Model*. Office of Solid Waste - OS-333.

U. S. EPA. 1988b. *National Survey of Solid Waste (Municipal) Landfill Facilities*. Draft. EPA/530-SW88-034. Office of Solid Waste and Emergency Response, Washington, D.C. September.

U.S. EPA. 1988c. *Drinking Water Criteria Document for Polychlorinated Biphenyls (PCBs)*. Final. PB89-199256. Prepared by: Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, OH. Prepared for: Office of Drinking Water. April.

U. S. EPA.1988d. *Interlaboratory Comparison of Methods 1310, 1311, and 1312 for Lead in Soil*. U.S. EPA Contract 68-01-7075. Research Triangle Institute, Research Triangle Park, NC. November.

U.S. EPA.1989a. *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A)*. EPA/540/1-89/002. Office of Emergency and Remedial Response, Washington, D.C. December.

U.S. EPA.1989b. *Exposure Factors Handbook*. EPA/600/8-89/043. Office of Health and Environmental Assessment, Exposure Assessment Group, Washington, D.C. July.

U.S. EPA. 1989c. "Interim Final Guidance for Soil Ingestion Rates;" from J. Winston Porter, OSWER Assistant Administrator; to Regional Administrators, Regions I-X. OSWER Directive 9850.4. Office of Solid Waste and Emergency Response. January 27.

U.S. EPA. 1990a. *Surface Impoundment Modeling System (SIMS) Version 2.0 User's Manual*. EPA-450/4-90-019a. Office of Air Quality Planning and Standards, Technical Support Division, Research Triangle Park, NC. September.

U.S. EPA. 1990b. *Background Document for the Surface Impoundment Modeling System (SIMS) Version 2.0*. EPA-450/4-90-019b. Office of Air Quality Planning and Standards, Technical Support Division, Research Triangle Park, NC. September.

U.S. EPA. 1990c. *Background Document for EPA's Composite Model for Landfills (EPACML)*. EPACML-S0002.A. Office of Solid Waste, Washington, D.C. February.

U.S. EPA.1990d. "Hazardous Waste Management System; Identification and Listing of Hazardous Waste; Toxicity Characteristics Revisions; Final Rule." *Federal Register*, 55(61): 11798-28895. March 29.

U.S. EPA. 1990e. "National Oil and Hazardous Substances Pollution Contingency Plan; Final Rule." *Federal Register*, 55(46). March 8.

U.S. EPA. 1990f. *Exposure Factors Handbook*. EPA/600/8-89/043. Office of Research and Development, Office of Health and Environmental Assessment. March.

U.S. EPA. 1990g. *Statistics of Aquifer Material Properties and Empirical pH-dependent Partitioning Relationships for As (III), As (V), Ba(II), Cd(II), Cr(VI), Cu(II), Hg(II), Ni(II), Pb(II), Sb(V), Se(IV), Tl(I), and Zn(II)*. Office of Research and Development, Athens GA; edited by N. T. Loux,, C. R. Chafin, and S. M. Hassan. March.

U.S. EPA. 1990h. *User's Manual for EPA's Composite Model for Landfills (EPACML)*. Office of Solid Waste, Washington, DC.

U.S. EPA. 1991a. *Risk Assessment Guidance for Superfund: Volume I - Human Health Evaluation Manual (Part B, Development of Risk-based Preliminary Remediation Goals)*. EPA/540/R-92/003; Publication 9285.7-01B. Office of Research and Development, Office of Emergency and Remedial Response, Washington, D.C. December.

U.S. EPA.1991b. "Hazardous Waste Management System; Identification and Listing of Hazardous Waste; Proposed Use of EPA's Composite Model for Landfills (EPACML) and Proposed Exclusion." *Federal Register*, 56(138): 32993-33012. July 18.

U.S. EPA.1991c. *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual, Supplemental Guidance, "Standard Default Exposure Factors"*. Interim Final. OSWER Directive: 9285.6-03. Office of Emergency and Remedial Response, Toxics Integration Branch, Washington, D.C. March 25.

U. S. EPA. 1991d. *MINTEQA2/PRODEFA2, A Geochemical Assessment Model for Environmental Systems: Version 3.0 User's Manual*. EPA/600/3-91/021. U.S. EPA Office of Research and Development, Environmental Research Laboratory, Athens, GA. March.

U.S. EPA. 1992a. "Guidelines for Exposure Assessment; Notice." *Federal Register*, 57(104): 22888-22938. May 29.

U.S. EPA. 1992b. *Dermal Exposure Assessment: Principles and Applications*. Interim Report. EPA/600/8-91/011B. Office of Research and Development, Office of Health and Environmental Assessment, Exposure Assessment Group, Washington, D.C. January.

U.S. EPA. 1992c. *Health Assessment for 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and Related Compounds. Chapters 1 to 8*. Workshop Review Draft. EPA/600/AP-92/001a through 001h. Office of Research and Development, Office of Health and Environmental Assessment, Washington, D.C. August.

U.S. EPA. 1992d. *Framework for Ecological Risk Assessment*. EPA/630/R-92/001. Risk Assessment Forum, Washington, D.C. February.

U.S. EPA.1993a. *Docket Report on Evaluation of Contaminant Releases to Surface Water Resulting from Conversion System's Petitioned Waste*. August 27.

U.S. EPA.1993b. *Parameter Values for the EPA's Composite Module for Landfills (EPACML) Used in Developing Nationwide Regulations: Toxicity Characteristic Rule*. Office of Solid Waste, Washington, D.C.

U.S. EPA.1993c. *Docket Report on Evaluation of Air Emissions Resulting from Conversion Systems, Inc.'s Petitioned Waste*. September 9.

U.S. EPA. 1993d. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. External Review Draft. EPA/600/AP-93/003. Office of Research and Development, Office of Health and Environmental Assessment, Exposure Assessment Group, Washington, D.C. November.

U.S. EPA.1993e. *Environmental Fate Constants for Organic Chemicals under Consideration for EPA's Hazardous Waste Identification Projects*. U. S. EPA, Environmental Research Laboratory, Office of Research and Development, Athens, GA. Edited by Kollig, Heinz.

U.S. EPA.1994a. *Docket Report on Evaluation of Contaminant Releases to Air from U.S. DOE Hanford's Petitioned Waste*. May 27.

U.S. EPA. 1994b. *Health Assessment Document for 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and Related Compounds, Volume III of III*. External Review Draft. EPA/600/BP-92/001c. Office of Research and Development, Office of Health and Environmental Assessment, Washington, D.C. August.

U.S. EPA. 1994c. *Estimating Exposure to Dioxin-Like Compounds. Volume II: Properties, Sources, Occurrence, and Background Exposures*. External Review Draft. EPA/600/6-88/005Cb. Office of Research and Development, Office of Health and Environmental Assessment, Exposure Assessment Group, Washington, D.C. June.

U.S. EPA.1994d. *Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 15.

U.S. EPA.1994e. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response, Office of Solid Waste. December 14.

U.S. EPA 1994f. "Ecological Screening Values".In: *Ecological Risk Assessment Bulletin No. 2*. Waste Management Division, U.S. EPA Region IV, Atlanta, GA.

U.S. EPA.1994g. *Draft Revision, Implementation Guidance for Conducting Indirect Exposure Analysis at RCRA Combustion Units. Attachment, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 22.

U.S. EPA.1994h. Memorandum. "Subject: Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities." OSWER Directive 9355.4-12. From Elliot P. Laws, Assistant Administrator. To Regional Administrators Regions I - X. July 14.

U.S. EPA. 1995a. "Hazardous Waste Management System: Identification and Listing of Hazardous Waste: Hazardous Waste Identification Rule (HWIR)." *Federal Register*, 60(245): 66344-66469. December 21.

U.S. EPA. 1995b. *Technical Support Document for the Hazardous Waste Identification Rule: Risk Assessment for Human and Ecological Receptors*. Office of Solid Waste, Washington, D.C. August.

U.S. EPA. 1996a. *Application of EPACMTP to Region 6 Delisting Program: Development of Waste Volume-Specific Dilution Attenuation Factors*. August.

U.S. EPA. 1996b. *EPACMTP Background Document*. Office of Solid Waste. Washington, D.C. September.

U.S. EPA. 1996c. *EPACMTP Background Document for the Finite Source Methodology for Chemicals with Transformation Products and Implementation of the HWIR*. Office of Solid Waste. Washington, D.C. September.

U.S. EPA.1996d. *Background Document for EPACMTP: Metals Transport in the Subsurface, Volume 1: Methodology*. Office of Solid Waste. Washington, D.C. August.

U.S. EPA.1996e. *The Region 6 RCRA Delisting Program Guidance Manual for the Petitioner*. U.S. EPA Region 6, Dallas, TX. March 21.

U.S. EPA. 1996f. *Soil Screening Guidance: User's Guide*. Second Edition. EPA/540/R-96/018. Publication 9355.4-23. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Washington, D.C. July.

U.S. EPA. 1996g. *Integrated Risk Information System*. *Database on the Toxicology Information Network.*

U.S. EPA.1996h. *PCBs: Cancer Dose-Response Assessment and Application to Environmental Mixtures*. EPA/600/P-96/001F. Office of Research and Development, National Center for Environmental Assessment, Washington, D.C. September.

U.S. EPA.1996i. "Ecotox Thresholds." *ECO Update* 3(2):1-12. EPA/540/F-95/038. Publication 9345.0-12FSI. Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Washington, D.C. January.

U.S. EPA. 1997a. *Supplemental Background Document; NonGroundwater Pathway Risk Assessment; Petroleum Process Waste Listing Determination*. 68-W6-0053. Office of Solid Waste, Research Triangle Park, North Carolina. March 20.

U.S. EPA.1997b. *Exposure Factors Handbook, Volumes I, II, and III*. EPA/600/P-95/002Fabc. Office of Research and Development, Washington, D.C. August.

U.S. EPA. 1997c. *Docket Report on Evaluation of Contaminant Releases to Surface Water Resulting from Conversion System's Petitioned Waste*. July 28.

U.S. EPA.1997d. *Human Health Risk Assessment Protocol*. US EPA Region 6, Dallas, TX, and Office of Solid Waste, Washington, D.C.

U.S. EPA. 1997e. *EPA's Composite Model for Leachate Migration with Transformation Products, EPACMTP: User's Guide*. Office of Solid Waste, Washington, D.C.

U.S. EPA.1997f. *Health Effects Assessment Summary Tables, FY 1997 Update*. 9200.6-303 (97-1). EPA 540/R-97-036. Office of Research and Development, Office of Solid Waste and Emergency Response, Office of Emergency and Remedial Response, Washington, D.C. Prepared for National Center for Environmental Assessment. July.

U.S. EPA. 1997g. "Test Methods for Evaluating Solid Waste, Physical/Chemical Methods, EPA Publication SW-846." *Federal Register*, 62(114): 32452. Friday, June 13.

U. S. EPA. 1997h. *Analysis of EPA's Industrial Subtitle D Databases used in Groundwater Pathway Analysis of the Hazardous Waste Identification Rule (HWIR)*. Office of Solid Waste, Washington, DC. September.

U.S. EPA. 1998a. *The Region 6 RCRA Delisting Risk Assessment Software User's Manual*. U.S. EPA Region 6, Dallas, TX. October.

U.S. EPA. 1998b. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Volume One*. Peer Review Draft. EPA530-D-98-001A. Office of Solid Waste and Emergency Response. July.

U.S. EPA.1998c. "Hazardous Waste Management System; Identification and Listing of Hazardous Waste; Petroleum Refining Process Wastes; Land Disposal Restrictions for Newly Identified Wastes; and CERCLA Hazardous Substance Designation and Reportable Quantities." *Federal Register*, 63(151): 42110. August 6.

U.S. EPA. 1998d. "National Recommended Water Quality Criteria; Notice; Republication." *Federal Register*, 63(237): 68354-68364. December 10.

U.S. EPA.1998e. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities*. Draft Interim Final. April.

U.S. EPA. 1999. *EPA's Composite Model for Leachate Migration with Transformation Products (EPACMTP) Background Document for Metals; Volume 2: Sorption Isotherms*. Office of Solid Waste, Washington, DC. August.

U.S. EPA.2001. *Industrial Surface Impoundments in the United States.* EPA530-R-01-005. Office of Solid Waste and Emergency Response. Washington, DC. March.

U.S. EPA 2003a. *EPACMTP Technical Background Document*. EPA530-R-03-006. Office of Solid Waste and Emergency Response. Washington, DC. April.

U.S. EPA 2003b. *EPACMTP Parameters/ Data Background Document.* EPA530-R-03-003. Office of Solid Waste and Emergency Response. Washington, DC. April.

U.S. EPA2005. *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, (Final)*. EPA530-R-05-006. Office of Solid Waste and Emergency Response. Washington, DC. September.

U.S. EPA 2006. *Memorandum, Subject: Updates to DRAS Toxicity Values*, Mangino, M. Ph.D., et al, U.S. EPA Region 5 Delisting Program, Chicago, Illinois. February 3.

Visher, S.S. 1954. *Climatic Atlas of the United States*. Cambridge, MA. Harvard University Press, Cambridge, MA. Oxford University Press, London.

Westat. 1987. *Nationwide Survey of Industrial Subtitle D Landfills*.

Wischmeier, W.H. and D.D. Smith. 1978. *Predicting Rainfall-Erosion Losses from Cropland East of the Rocky Mountains*. USDA Handbook, No. 282. Agricultural Research Service.