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Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities

Final

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September 2005

Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities

U.S. EPA, OFFICE OF SOLID WASTE

U.S. ENVIRONMENTAL PROTECTION AGENCY

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DISCLAIMER

This document provides guidance to EPA regional and state RCRA hazardous waste programs, as well as to facilities subject to RCRA requirements and the general public. More specifically, this guidance document conveys how EPA generally intends to exercise its discretion in implementing RCRA statutory and regulatory provisions concerning combustion facilities subject to RCRA. EPA designed this guidance to explain and clarify national policy on issues related to EPA's obligation to ensure that operating permits granted to combustion facilities contain conditions necessary to protect human health and the environment.

The statutory provisions and EPA regulations discussed in this handbook contain legally binding requirements. This guidance itself does not substitute for those provisions, nor is it a regulation itself. Thus, this guidance does not impose legally binding requirements on EPA, states, or the regulated community, and may not apply to a particular situation based on the specific circumstances of the combustion facility. EPA and state regulators base their permitting decisions on the statute and regulations as applied to the specific combustion facility and retain their discretion to use approaches on a case-by-case basis that differ from those recommended in this guidance where appropriate. Therefore, interested parties are free to raise questions and concerns about the substance of this guidance document and the appropriateness of the application of recommendations to a particular situation. Because this guidance is not a regulation, EPA and state regulators will consider such questions and concerns when implementing the recommendations (for example, during the comment period provided on draft combustion permits). Whether the recommendations in this Handbook are appropriate in a given situation will depend on facility-specific circumstances.

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LIST OF ACRONYMS

:g	Microgram
:m	Micrometer
ACGIH	American Conference of Governmental Industrial Hygienists
ADD	Average daily dose
AEFA	Average emission factor approach
AEGL	Acute inhalation exposure guidelines
AERMOD	American Meteorological Society/EPA Regulatory Model
Ah	Aryl hydrocarbon
AHH	Aryl hydrocarbon hydroxylase
AIEC	Acute inhalation exposure criteria
AIHA	American Industrial Hygiene Association
APCD	Air pollution control device
APCS	Air pollution control system
ARE	Acute reference exposure
Acute REL	Acute reference exposure level
ASTM	American Society for Testing and Materials
atm	Atmosphere
ATSDR	Agency for Toxic Substances and Disease Registry
AWFCO	Automatic waste feed cutoff
BaP	Benzo(a)pyrene
BAF	Bioaccumulation factor
BBS	Bulletin board service
BCF	Bioconcentration factor
BEHP	Bis(2-ethylhexyl) phthalate
BIF	Boiler and industrial furnace
BPIP	Building profile input program check
BSAF	Sediment bioaccumulation factor
Btu	British thermal unit
BW	Body weight
CAA	Clean Air Act
CALPUFF	California Puff Model
CARB	California Air Resources Board
CAS	Chemical Abstracts Service
CFR	Code of Federal Regulations
CKD	Cement kiln dust
CLP	Contract Laboratory Program
cm	Centimeters
COPC	Compound of potential concern
CRQL	Contract required quantitation limit
CSV	Unspeciated chromatographical semivolatiles
CWA	Clean Water Act

LIST OF ACRONYMS (contd.)

DEHP	Diethylhexylphthalate
dL	Decaliter
DNA	Deoxyribonucleic acid
DNOP	Di(n)octyl phthalate
DOE	Department of Energy
DRE	Destruction and removal efficiency
DW	Dry weight of soil or plant/animal tissue
EPACA	U.S. Environmental Protection Agency Correlation Approach
EQL	Estimated quantitation limit
ERPG	Emergency response planning guidelines
ESP	Electrostatic precipitator
FW	Fresh weight (or whole/wet weight) of plant or animal tissue
g	Grams
GAQM	Guideline to Air Quality Models
GC	Gas chromatography
GEP	Good engineering practice
GRAV	Unspeciated gravimetric compounds
H3TD	Hierarchy of Human Health Toxicity Data
HEAST	Health Effects Assessment Summary Tables
HI	Hazard index
HQ	Hazard quotient
IARC	International Agency for Research on Cancer
IDL	Instrument detection limit
IEU/BK	Integrated exposure uptake/biokinetic
IPM	Insoluble polystyrene microspheres
IUPAC	International Union of Pure and Applied Chemistry
IRIS	Integrated Risk Information System
ISC-PRIME	Industrial Source Complex-Plume Rise Model Enhancements
ISCSTDFT	Industrial Source Complex Short Term Draft
ISCST3	Industrial Source Complex Short Term 3
K	Kelvin
kg	Kilogram
LADD	Lifetime average daily dose
L	Liter
lb	Pound
LCD	Local climatological data annual summary with comparative data
m	Meters
MACT	Maximum achievable control technology

LIST OF ACRONYMS (contd.)

MDL	Method detection limit
MEHP	Monoethylhexyl phthalate
mg	Milligram
Mg	Megagram
MIR	Maximum individual risk
MJ	Megajoule
mL	Milliliter
MPRM	Meteorological processor for regulatory models
MPTER	Air quality model for multiple point source gaussian dispersion algorithm with terrain adjustments
MRL	Minimum risk level
NCDC	National Climatic Data Center
NC DEHNR	North Carolina Department of Environment, Health, and Natural Resources
NCEA	National Center for Environmental Assessment
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NIOSH	National Institute of Occupational Safety and Health
NRC	Nuclear Regulatory Commission
NRC COT	National Research Council Committee on Toxicology
NTP	National Toxicology Program
NWS	National Weather Service
OAQPS	Office of Air Quality Planning and Standards
OEHHA	Office of Environmental Health Hazard Assessment
ORD	Office of Research and Development
OSHA	U.S. Occupational Safety and Health Administration
OSW	Office of Solid Waste
OSWER	Office of Solid Waste and Emergency Response
PAH	Polynuclear aromatic hydrocarbon
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo(p)dioxin
PCDF	Polychlorinated dibenzofuran
PCRAMMET	Personal computer version of the meteorological preprocessor for the old RAM program
PDF	Probability density function
pg	Picogram
PIC	Product of incomplete combustion
PM	Particulate matter
PMD	Portable monitoring device
PM10	Particulate matter less than 10 micrometers in diameter
POHC	Principal organic hazardous constituent
ppb	Parts per billion
ppm	Parts per million
ppmv	Parts per million by volume
ppt	Parts per trillion

LIST OF ACRONYMS (contd.)

PQL	Practical quantitation limit
PU	Polyurethane
QA	Quality assurance
QAPjP	Quality assurance project plan
QC	Quality control
RCRA	Resource Conservation and Recovery Act
RfC	Reference concentration
RfD	Reference dose
RME	Reasonable maximum exposure
RPF	Relative potency factor
RTDM	Rough terrain diffusion model
RTDMDEP	Rough terrain diffusion model deposition
s	Second
SAMSON	Solar and Meteorological Surface Observational Network
SCAPA	Subcommittee on Consequence Assessment and Protective Actions
SCRAM	Support Center for Regulatory Air Models
SF	Slope factor
SLERA	Screening level ecological risk assessment
SOCMI	Synthetic Organic Chemical Manufacturing Industries
SQL	Sample quantitation limit
SRA	Screening ranges approach
SVOC	Semivolatile organic compound
SW-846	U.S. Environmental Protection Agency Test Methods for Evaluating Solid Waste
TCDD	Tetrachlorodibenzo(p)dioxin
TDA	Toluenediamine
TDI	Toluene diisocyanate
TEELs	Temporary emergency exposure limits
TEF	Toxicity equivalent factor
TEQ	Toxicity equivalent quotient
TG	Terrain grid
TIC	Tentatively identified compound
TLV	Threshold limit value
TOC	Total organic carbon
TOE	Total organic emissions
TSD	Treatment, storage, and disposal
TTN	Technology transfer network
TWA	Time-weighted average
U/BK	Uptake/biokinetic
USCA	Unit-Specific Correlation Approach
USDA	U.S. Department of Agriculture

LIST OF ACRONYMS (contd.)

U.S. EPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
USLE	Universal soil loss equation
UTM	Universal transverse mercator
VOC	Volatile organic compound
WHO	World Health Organization

INDEXED LIST OF VARIABLES

Variable	Units	Definition	**[Sections]**/ Equations
γ	unitless	Empirical constant	used to generate *Rp*
λ_z	unitless	Dimensionless viscous sublayer thickness	5-41B; 5-42B; B-4-20; B-4-21
μ_a	g/cm-s	Viscosity of air	5-42B; B-4-21
μ_w	g/cm-s	Viscosity of water corresponding to water temperature	5-41B; B-4-20
ρ_a	g/cm^3 or g/m^3	Density of air	5-18; 5-41-B; 5-42B; B-2-8; B-3-8; B-4-20; B-4-21
ρ_s	kg/L	Bed sediment density	used to generate θ_{bs}
ρ_{soil}	g/cm^3	Solids particle density	5-7a; B-1-6; B-2-6; B-3-6; B-4-6
ρ_w	g/cm^3	Density of water corresponding to water temperature	5-41B; B-4-20
θ	unitless	Temperature correction factor	5-40; B-4-19
θ_{bs}	unitless	Bed sediment porosity	5-36B; 5-37; 5-47; B-4-16; B-4-25
θ_{sw}	mL water/cm^3 soil	Soil volumetric water content	**[5.2.4.4]**; 5-4; 5-5A; 5-7A; 5-7C; 5-32; 5-33; B-1-3; B-1-4; B-1-5; B-1-6; B-2-3; B-2-4; B-2-5; B-2-6; B-3-3; B-3-4; B-3-5; B-3-6; B-4-3; B-4-4; B-4-5; B-4-6; B-4-10; B-4-11
θ_v	cm^3/cm^3	Soil void fraction	5-7B; 5-7C
a	unitless	Empirical intercept coefficient	5-34; B-4-14
A_{beef}	mg COPC/kg FW tissue	Concentration of COPC in beef	**[5.4.4]**; 5-22; B-3-10
$A_{chicken}$	mg COPC/kg FW tissue	Concentration of COPC in chicken meat	**[5.6.1]**; 5-26; B-3-14
ADD	mg COPC/kg BW-day	Average daily dose	6-1
ADD_{infant}	pg COPC/kg BW infant/day	Average daily dose for infant exposed to contaminated breast milk	C-3-2
ADD_{mat}	pg COPC/kg BW mother/day	Average daily dose (mother)	
AEF	kg/hr-source	Applicable average emission factor for the equipment type	
A_{egg}	mg COPC/kg FW tissue	Concentration of COPC in eggs	**[5.6.1]**; C-1-3
Ah	m^2	Area planted	used to estimate Yp
Ahi	m^2	Area planted to *i*th crop	see *Ah*
A_I	m^2	Impervious watershed area receiving COPC deposition	5-31; 5-32; 5-33; 5-36C; B-4-9; B-4-10

INDEXED LIST OF VARIABLES (contd.)

Variable	Units	Definition	**[Sections]**/ Equations
A_L	m^2	Total watershed area receiving COPC deposition	5-32; 5-33; 5-34; 5-36C; 5-43; B-4-10; B-4-11;B-4-14; B-4-22
A_{milk}	mg COPC/kg FW tissue	Concentration of COPC in milk	**[5.4.5]**; 5-24; B-3-11
A_{pork}	mg COPC/kg FW tissue	Concentration of COPC in pork	**[5.5.1]**; 5-25; B-3-12
AT	days	Averaging time	**[6.5]**; 6-1; C-1-7; C-1-8; C-3-1
A_W	m^2	Water body surface area	**[4.1.2]**; 5-29; 5-30; 5-35; 5-36C; 5-43; B-4-8; B-4-12; B-4-15; B-4-22
b	unitless	Empirical slope coefficient	5-34; B-4-14
Ba_{beef}	day/kg FW tissue	Biotransfer factor for beef	5-22; **[A2.5.1]**; A-2-16; B-3-10
$Ba_{chicken}$	day/kg FW tissue	Biotransfer factor for chicken	5-26; **[A2.5.3]**; B-3-14
Ba_{egg}	day/kg FW tissue	Biotransfer factor for eggs	5-26; **[A2.5.3]**; A-2-18; B-3-13
BAF_{fish}	L/kg FW tissue	Bioaccumulation factor for fish	5-49; **[A2.5.4]**; B-4-27
Ba_{milk}	day/kg FW tissue	Biotransfer factor for milk	5-24; **[A2.5.1]**; A-2-17; B-3-11
Ba_{pork}	day/kg FW tissue	Biotransfer factor for pork	5-25; **[A2.5.2]**; B-3-12
BCF_{fish}	unitless (mg COPC/kg FW tissue)/(mg COPC/kg dissolved water)	Bioconcentration factor for fish	5-48; **[A2.5.4]**; B-4-26
BD	g soil/cm^3 soil	Soil bulk density	**[5.2.4.2]**; 5-4; 5-5A; 5-7A; 5-11; 5-32; 5-33; B-1-1; B-1-3; B-1-4; B-1-5; B-1-6; B-2-1; B-2-3; B-2-4; B-2-5; B-2-6; B-3-1; B-3-3; B-3-4; B-3-5; B-3-6; B-4-1; B-4-3; B-4-4; B-4-5; B-4-6; B-4-10; B-4-11
Br_{ag}	unitless	Plant-soil bioconcentration factor for aboveground produce	5-20A; **[A2.4.3]**; A-2-14A; B-2-9
Br_{forage}	unitless (µg COPC/g DW plant)/(µg COPC/g soil)	Plant-soil bioconcentration factor for forage	5-20A; **[A2.4.3]**; A-2-14B; B-3-9

INDEXED LIST OF VARIABLES (contd.)

Variable	Units	Definition	**[Sections]**/ Equations
Br_{grain}	unitless (µg COPC/g DW plant)/(µg COPC/g soil)	Plant-soil bioconcentration factor for COPC in grain	5-20A; **[A2.4.3]**; A-2-14B; B-3-9
$Br_{rootveg}$	unitless (µg COPC/g FW plant)/(µg COPC/g soil)	Plant-soil bioconcentration factor for COPC in belowground produce	5-20B; **[A2.4.2]**; A-2-13; B-2-10
Bs	unitless	Soil bioavailability factor	**[5.4.4.6]**; 5-22; 5-24; 5-25; 5-26; B-3-10; B-3-11; B-3-12; B-3-13; B-3-14
$BSAF$	unitless	Biota-to-sediment accumulation factor	**[5.75]**; 5-50; **[A2.5.4.3]**; B-4-28
Bv_{ag}	unitless (mg COPC/kg lipid tissue)/(mg COPC/kg sediment)	COPC air-to-plant biotransfer factor for aboveground produce (µg COPC/g DW plant)/(µg COPC/g air)—unitless	5-18; **[A2.4.4]**; A-2-15A&B; B-2-8
$Bv_{forage/silage}$	unitless (µg COPC/g DW plant)/(µg COPC/g air)	Air-to-plant biotransfer factor for forage and silage	5-18; **[A2.4.4]**; A-2-15A&B; B-3-8
C	unitless	USLE cover management factor	5-33A; B-4-13
C_a	µg/m^3	Total COPC air concentration	**[6.1]**; 7-1; 7-5; B-5-1; C-2-1; C-2-2; C-3-1
C_{acute}		Acute air concentration (µg/m^3)	7-9; B-6-1; C-4-1
$Cancer\ Risk_i$	unitless	Individual lifetime risk through indirect exposure to COPC carcinogen *i*	7-3; C-1-7
$Cancer\ Risk_{inh(i)}$	unitless	Individual lifetime cancer risk through direct inhalation of COPC carcinogen *i*	C-2-1
C_{BS}	g sediment/cm^3 water	Bed sediment concentration (or sediment bulk density)	5-36A; 5-37; 5-43; 5-47; B-4-16; B-4-22; B-4-25
C_d	unitless	Drag coefficient	5-41B; 5-42B; B-4-20;
C_{dw}	mg COPC/L water	Dissolved phase water concentration	**[5.7.4.9]**; 5-46; 5-48; 5-49; B-4-24; B-4-26; B-4-27; C-1-5
C_{fish}	mg COPC/kg FW tissue	Concentration of COPC in fish	**[5.7.5]**; 5-48; 5-49; 5-50; B-4-28; C-1-4
Chv	µg-s/g-m^3	Unitized hourly air concentration from vapor phase	B-6-1
Chp	µg-s/g-m^3	Unitized hourly air concentration from particle phase	B-6-1
$Chpb$	µg-s/g-m^3	Unitized hourly air concentration from particle-bound phase	

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Variable	Units	Definition	**[Sections]**/ Equations
Cs	mg COPC/kg soil	Average soil concentration over exposure duration	**[5.2.1]**; 5-1C&D; 5-20A&B; 5-22; 5-24; 5-25; 5-26; 5-32; 5-33; B-1-1; B-2-1; B-2-9; B-2-10; B-3-1; B-3-9; B-3-10; B-3-11; B-3-12; B-3-13; B-3-14; B-4-1; B-4-10; B-4-11
C_{sb}	mg COPC/kg sediment	Concentration sorbed to bed sediment	**[5.7.4.10]**; 5-47; 5-50; B-4-25; B-4-28
CSF	$(mg/kg\text{-}day)^{-1}$	Cancer slope factor	7-2; **[A2.6.2]**; C-1-7
Cs_{tD}	mg COPC/kg soil	Soil concentration at time *tD*	**[5.2.1]**; 5-1E; B-1-1; B-2-1; B-3-1; B-4-1; C-3-1
C_{wctot}	mg COPC/L water column	Total COPC concentration in water column	**[5.7.4.8]**; 5-45; 5-46; B-4-23; B-4-24
C_{wtot}	g COPC/m^3 water body (or mg/L)	Total water body COPC concentration including water column and bed sediment	**[5.7.4]**; 5-35; 5-45; 5-47; B-4-15; B-4-23; B-4-25
Cyp	µg-s/g-m^3	Unitized yearly average air concentration from particle phase	**[3.8.3.2]**; B-5-1
Cyv	µg-s/g-m^3	Unitized yearly average air concentration from vapor phase	**[3.8.3.1]**; 5-18; B-2-8; B-3-8; B-5-1
$Cywv$	µg-s/g-m^3	Unitized yearly (water body and watershed) average air concentration from vapor phase	**[3.8.3.1]**; 5-30; B-4-12
D_a	cm^2/s	Diffusivity of COPC in air	5-7A; 5-42B; **[A2.3.5]**; A-2-2A; B-1-6; B-2-6; B-3-6; B-4-6; B-4-21
d_{bs}	m	Depth of upper benthic sediment layer	5-35; 5-36A; 5-43; 5-45; 5-47; B-4-15; B-4-16; B-4-18; B-4-22; B-4-23; B-4-25
D_{mean}		Mean particle size density for a particular filter cut size	3-1
Ds	mg COPC/kg soil-yr	Deposition term	**[5.2.3]**; 5-1C, D&E; 5-36C; B-1-1; B-2-1; B-3-1; B-4-1
d_{wc}	m	Depth of water column	**[4.1.2]**; 5-35; 5-36A; 5-45; 5-47; B-4-15; B-4-16; B-4-18; B-4-23; B-4-25
D_w	cm^2/s	Diffusivity of COPC in water	5-41A&B; **[A2.3.5]**; A-2-2B; B-4-20

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Variable	Units	Definition	**[Sections]**/ Equations
$Dydp$	s/m^2-yr	Unitized yearly average dry deposition from particle phase	**[3.8.3.2]**; 5-11; 5-14; B-1-1; B-2-1; B-2-7; B-3-1; B-3-7
$Dydv$	s/m^2-yr	Unitized yearly average dry deposition from vapor phase	**[3.8.3.2]**; 5-11; 5-14; B-1-1; B-2-1; B-2-7; B-3-1; B-3-7
$Dytwp$	s/m^2-yr	Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase	**[3.8.3.2]**; 5-29; 5-31; B-4-1; B-4-8; B-4-9
$Dywp$	s/m^2-yr	Unitized yearly average wet deposition from particle phase	**[3.8.3.2]**; 5-11; 5-14; B-1-1; B-2-1; B-2-7; B-3-1; B-3-7
$Dywv$	s/m^2-yr	Unitized yearly average wet deposition from vapor phase	**[3.8.3.2]**; 5-11; 5-14; B-1-1; B-2-1; B-2-7; B-3-1; B-3-7
$Dytwv$	s/m^2-yr	Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase	**[3.8.3.2]**; 5-29; 5-31; B-4-1; B-4-8; B-4-9
d_z	m	Total water body depth	5-36A; 5-39; 5-41A; B-4-16; B-4-18; B-4-20
ED	yr	Exposure duration	6-1; C-1-7; C-1-8; C-3-1; C-3-2
EF	days/yr	Exposure frequency	6-1; C-1-7; C-1-8; C-3-1
ER	unitless	Soil enrichment ratio	5-33; B-1-3; B-2-3; B-3-3; B-4-3; B-4-11
E_v	cm/yr	Average annual evapotranspiration	5-5A; B-1-5; B-2-5; B-3-5; B-4-5
f_{bs}	unitless	Fraction of total water body COPC concentration in benthic sediment	**[5.7.4.1]**; 5-36B; 5-38; 5-47; B-4-16; B-4-17; B-4-25
F_i	unitless	Fraction of plant type *i* grown on contaminated soil and eaten by the animal	5-22; 5-24; 5-25; 5-26; B-3-10; B-3-11; B-3-12; B-3-13B-3-14
f_{lipid}	unitless	Fish lipid content	5-50; B-4-28
Fw	unitless	Fraction of COPC wet deposition that adheres to plant surfaces	5-14; B-2-7; B-3-7
f_{wc}	unitless	Fraction of total water body COPC concentration in the water column	**[5.7.4.1]**; 5-35; 5-36A; 5-38; 5-45; B-4-15; B-4-16; B-4-17; B-4-23

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Variable	Units	Definition	**[Sections]**/ Equations
F_v	unitless	Fraction of COPC air concentration in vapor phase	**[3.2]**; 5-11; 5-14; 5-18; 5-29; 5-30; 5-31; B-1-1; B-2-1; B-2-7; B-2-8; B-3-1; B-3-7; B-3-8; B-4-1; B-4-8; B-4-9; B-4-12; B-5-1; B-6-1
H	atm-m^3/mol	Henry's Law constant	5-7A; 5-30; 5-40; **[A2.3.4]**; A-2-1; B-1-6; B-2-6; B-3-6; B-4-6; B-4-12; B-4-19
HI	unitless	Hazard index	7-6; 7-7; C-1-11
HI_j	unitless	Hazard index for exposure pathway *j*	C-1-10
HQ	unitless	Hazard quotient	7-5; C-1-8
HQ_i	unitless	Hazard quotient for COPC *i*	7-6
$\text{HQ}_{\text{inh(i)}}$	unitless	Hazard quotient for direct inhalation of COPC	C-2-2; C-2-4
I	cm/yr	Average annual irrigation	5-5A; B-1-5; B-2-5; B-3-5; B-4-5
I_i	mg/day	Daily intake of COPC (*i*) from animal tissue	**[6.2.2]**; C-1-3
k	unitless	von Karman's constant	5-41B; 5-42B; B-4-20; B-4-21
K	ton/acre	USLE erodibility factor	5-33A; B-4-13
k_b	yr^{-1}	Benthic burial rate constant	**[5.7.4.7]**; 5-38; 5-43; 5-44; B-4-17
Kd_{bs}	cm^3 water/g bottom sediment	Bed sediment/sediment pore water partition coefficient	5-36B; 5-47; **[A2.3.8]**; A-2-8C; B-4-16; B-4-25;
Kd_{ij}	unitless	Partition coefficient for COPC *i* associated with sorbing material *j*	
Kd_s	cm^3 water/g soil	Soil-water partition coefficient	5-4; 5-5A; 5-7A; 5-20B; 5-32; 5-33; **[A2.3.8]**; A-2-8A; B-1-3; B-1-4; B-1-5; B-1-6; B-2-3; B-2-4; B-2-5; B-2-6; B-2-10; B-3-3; B-3-4; B-3-5; B-3-6; B-4-3; B-4-4; B-4-5; B-4-6; B-4-10; B-4-11
Kd_{sw}	L water/kg suspended sediment	Suspended sediments/surface water partition coefficient	5-36A; 5-39; 5-46; **[A2.3.8]**; A-2-8B; B-4-16; B-4-18; B-4-24
K_G	m/y	Gas phase transfer coefficient	**[5.7.4.6]**; 5-40; 5-42A&B; B-4-19; B-4-21

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Variable	Units	Definition	**[Sections]**/ Equations
K_L	m/yr	Liquid phase transfer coefficient	**[5.7.4.5]**; 5-40; 5-41A&B; B-4-19; B-4-20
K_{oc}	mL water/g soil	Soil organic carbon-water partition coefficient	**[A2.3.7]**; A-2-4; A-2-5; A-2-6; A-2-7;
K_{ow}	unitless (mg COPC/L octanol)/(mg COPC/L octanol)	Octanol-water partition coefficient	**[A2.3.6]**; A-2-4; A-2-5; A-2-6; A-2-7; A-2-12A&B; A-2-14A&B; A-2-15A; A-2-16; A-2-17; A-2-19
kp	yr^{-1}	Plant surface loss coefficient	**[5.3.1.2]**; 5-14; B-2-7; B-3-7
ks	yr^{-1}	COPC soil loss constant due to all processes	**[5.2.2]**; 5-1C, D&E; B-1-1; B-1-2; B-2-1; B-2-2; B-3-1; B-3-2; B-4-1; B-4-2;
kse	yr^{-1}	COPC loss constant due to soil erosion	**[5.2.2.2]**; 5-2A; B-1-2; B-1-3; B-2-2; B-2-3; B-3-2; B-3-3; B-4-2; B-4-3
ksg	yr^{-1}	COPC loss constant due to biotic and abiotic degradation	**[5.2.2.1]**; 5-2A; **[A2.3.9]**; A-2-9; B-1-2; ; B-2-2; B-3-2; B-4-2
ksl	yr^{-1}	COPC loss constant due to leaching	**[5.2.2.4]**; 5-2A; 5-5A; B-1-2; B-1-5; B-2-2; B-2-5; B-3-2; B-3-5; B-4-2; B-4-5
ksr	yr^{-1}	COPC loss constant due to surface runoff	**[5.2.2.3]**; 5-2A; 5-4; B-1-2; B-1-4; B-2-2; B-2-4; B-3-2; B-3-4; B-4-2; B-4-4
ksv	yr^{-1}	COPC loss constant due to volatilization	**[5.2.2.5]**; 5-2A; 5-7A; B-1-2; B-1-6; B-2-2; B-2-6; B-3-2; B-3-6; B-4-2; B-4-6
k_v	yr^{-1}	Water column volatilization rate constant	**[5.7.4.3]**; 5-38; 5-39; B-4-17; B-4-18
K_v	m/yr	Overall COPC transfer rate coefficient	**[5.7.4.4]**; 5-30; 5-39; 5-40; B-4-12; B-4-18; B-4-19
k_{wt}	yr^{-1}	Overall total water body dissipation rate constant	**[5.7.4.2]**; 5-35; 5-38; B-4-15;B-4-17
L	m	Monin-Obukhov Length	**[3.5.1]**
$LADD$	mg COPC/kg BW-day	Lifetime average daily dose	7-2

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Variable	Units	Definition	**[Sections]**/ Equations
L_{DEP}	g/yr	Total (wet and dry) particle phase and vapor phase COPC direct deposition load to water body	**[5.7.1.1]**; 5-28; 5-29; B-4-7; B-4-8
L_{dif}	g/yr	Vapor phase COPC diffusion load to water body	**[5.7.1.2]**; 5-28; 5-30; B-4-7; B-4-12
leak rate	kg/hr	Emission rate from the individual item of equipment	**[2.2.6.1]**
L_E	g/yr	Soil erosion load	**[5.7.1.5]**; 5-28; 5-33; B-4-7; B-4-11
L_R	g/yr	Runoff load from pervious surfaces	**[5.7.1.4]**; 5-28; 5-32; B-4-7; B-4-10
L_{RI}	g/yr	Runoff load from impervious surfaces	**[5.7.1.3]**; 5-28; 5-31; B-4-7; B-4-9
L_T	g/yr	Total COPC load to the water body including deposition, runoff, and erosion	**[5.7.1]**; 5-28; B-4-7; B-4-15
LS	unitless	USLE length-slope factor	5-33A; B-4-13
M_{skin}	g	Mass of a thin (skin) layer of below ground vegetable	5-19
$M_{vegetable}$	g	Mass of the entire vegetable	5-19
MF	unitless	Metabolism factor	**[5.4.4.7]**; 5-22; 5-24; 5-25; B-3-10; B-3-11; B-3-12
MW	g/mole	Molecular weight	**[A2.3.1]**; A-2-1
OC_{sed}	unitless	Fraction of organic carbon in bottom sediment	5-50; B-4-28
$p°_L$	atm	Liquid phase vapor pressure of chemical	A-2-11
$p°_S$	atm	Solid phase vapor pressure of chemical	A-2-11
P	cm/yr	Average annual precipitation	5-5A; B-1-5; B-2-5; B-3-5; B-4-5
PF	unitless	USLE supporting practice factor	5-33A; B-4-13
Pd	mg COPC/kg DW	Aboveground exposed produce concentration due to direct (wet and dry) deposition onto plant surfaces	**[5.3.1]**; 5-14; 5-23; B-2-7; B-3-7; C-1-2
P_i	mg/kg DW	Total COPC concentration in plant type *i* ingested by the animal	**[5.4.4.3]**; 5-22; 5-23; 5-24; 5-25; 5-26; 5-27; B-3-10; B-3-11; B-3-12; B-3-13; B-3-14
Pr	mg COPC/kg DW	Aboveground exposed and protected produce concentration due to root uptake	**[5.3.3]**; 5-20A&B; 5-23; 5-27; B-2-9; B-3-9; C-1-2
Pr_{bg}	mg COPC/kg DW	Belowground produce concentration due to root uptake	B-2-10; C-1-2
Pv	mg COPC/kg DW	Concentration of COPC in plant due to air-to-plant transfer	**[5.3.2]**; 5-18; 5-23; B-2-8; B-3-8; C-1-2

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Variable	Units	Definition	**[Sections]**/ Equations
Q	g/s	COPC emission rate	5-11; 5-14; 5-18; 5-29; 5-30; 5-31; B-1-1; B-2-1; B-2-7; B-2-8; B-3-1; B-3-7; B-3-8; B-4-1; B-4-8; B-4-9; B-4-12; B-5-1; B-6-1
Q_i	g/s	Emission rate of COPC *(i)*	
$Q_{i(adj)}$	g/s	Adjusted emission rate of COPC *(i)*	
$Qcp_{i(adj)}$	g/s	Adjusted emission rate of Table A-1 carcinogenic COPC *(i)*	
Qcp_i	g/s	Emission rate of Table A-1 carcinogenic COPC *(i)*	
Q_f	W/m^2	Anthropogenic heat flux	**[3.5.7]**
Qp_i	kg DW/day	Quantity of plant type *i* ingested by the animal each day	**[5.4.4.2]**; 5-22; 5-24; 5-25; 5-26; B-3-10; B-3-11; B-3-12; B-3-13; B-3-14
Qs	kg/day	Quantity of soil ingested by the animal each day	
Q_*	W/m^2	Net radiation absorbed	**[3.5.8]**
r	unitless	Interception fraction—the fraction of material in rain intercepted by vegetation and initially retained	
R	atm-m^3/mol-K	Universal gas constant	5-7A; 5-30; 5-40; A-2-11; B-1-6; B-2-6; B-3-6; B-4-6; B-4-12; B-4-19
RCF	(µg COPC/g DW plant)/(µg COPC/mL soil water)	Root concentration factor	5-20B; **[A2.4.1]**; A-2-12A&B; A-2-13; B-2-10
RO	cm/yr	Average annual surface runoff from pervious surfaces	5-4; 5-5A; 5-32; B-1-4; B-1-5; B-2-4; B-2-5; B-3-4; B-3-5; B-4-4; B-4-5; B-4-10
REL		California EPA Air Toxics Hot Spots Program acute reference exposure levels	**[7.4.2]**
RF	yr^{-1}	USLE rainfall (or erosivity) factor	5-33A; B-4-13
RfC		Inhalation reference dose	7-5; **[A2.6.1]**; C-2-2
RfD	mg COPC/kg body weight/day	Oral reference dose	7-5; **[A2.6.1]**; C-1-8
Rp	unitless	Interception fraction of the edible portion of plant	**[5.3.11]**; 5-14; B-2-7; B-3-7
S	mg COPC/L water	Solubility of COPC in water	**[A2.3.3]**; A-2-1

INDEXED LIST OF VARIABLES (contd.)

Variable	Units	Definition	**[Sections]**/ Equations
SD	unitless	Sediment delivery ratio	**[5.7.3]**; 5-33; 5-34; 5-36C; 5-43; B-1-3; B-2-3; B-3-3; B-4-3
$)Sf$	unitless	Entropy of fusion [$)S_f/R$ = 6.79]	A-2-11
SF	$(\text{mg/kg-day})^{-1}$	Slope factor	
S_T	cm^2/cm^3 air	Whitby's average surface area of particulates (aerosols)	A-2-11
T_a	K	Ambient air temperature	**[3.4.2]**; 5-7A; A-2-11; B-1-6; B-2-6; B-3-6; B-4-6
T_1	yr	Time period at the beginning of combustion	5-1D; B-1-1; B-2-1; B-3-1; B-4-1
T_2	yr	Length of exposure duration	5-1C&D; B-1-1; B-2-1; B-3-1; B-4-1
tD	yr	Time period over which deposition occurs (time period of combustion)	5-1C, D&E; B-1-1; B-2-1; B-3-1; B-4-1
T_m	K	Melting point of chemical	**[A2.3.2]**
Tp	yr	Length of plant exposure to deposition per harvest of edible portion of plant	**[5.3.1.3]**; 5-14; 5-16; 5-21; B-2-7; B-3-7
tp_i	yr	Length of plant's exposure to deposition per harvest of the edible portion of the *i* th plant group	5-13
Total Cancer Risk	unitless	Individual lifetime cancer risk through indirect exposure to all COPC carcinogens	7-3; 7-4; C-1-9
Total Cancer $Risk_{inh}$	unitless)	Total individual lifetime cancer risk through direct inhalation of all COPC carcinogens	C-2-3
TSS	mg/L	Total suspended solids concentration	5-36A; 5-36C; 5-39; 5-43; 5-46; B-4-16; B-4-18; B-4-22; B-4-24
T_{wk}	K	Water body temperature	5-30; 5-40; B-4-12; B-4-19;
$t_{1/2}$	days	Half-time of COPC	5-15
u	m/s	Current velocity	5-41A; B-4-20
URF	$:\text{g/m}^3$	Unit risk factor	7-1; C-2-1
Vf_x	m^3/yr	Average volumetric flow rate through water body	5-35; 5-36C; 5-43; B-4-15; B-4-22;
VG_{ag}	unitless	Empirical correction factor for aboveground produce (forage and silage)	**[5.3.2.1; 5.4.2.1]**; 5-18; B-2-8; B-3-8
$VG_{rootveg}$	unitless	Empirical correction factor for below ground produce	5-19; 5-20B; B-2-10
Vp	atm	Vapor pressure of COPC	**[A2.3.3]**; A-2-1
W	m/s	Average annual wind speed	**[3.4.1]**; 5-41B; 5-42B; B-4-20; B-4-21
W_b	m/yr	Rate of burial	5-44

INDEXED LIST OF VARIABLES (contd.)

Variable	Units	Definition	**[Sections]**/ Equations
X_e	kg/m^2-yr	Unit soil loss	**[5.7.2]**; 5-33; 5-33A; 5-36C; 5-43; B-1-3; B-2-3; B-3-3; B-4-3; B-4-11; B-4-13; B-4-22
Yh	kg DW	Dry harvest yield	
Yh_i	kg DW	Harvest yield of *i*th crop	
Yp	kg DW/m^2	Yield or standing crop biomass of edible portion of plant (productivity)	**[5.3.1.4; 5.4.1.4]**; 5-14; B-2-7; B-3-7
Yp_i	kg DW/m^2	Yield or standing crop biomass of the edible portion of the plant (productivity)	5-13
Z_s	cm	Soil mixing zone depth	**[5.2.4.1]**; 5-4; 5-5A; 5-7A&B; 5-11; B-1-1; B-1-3; B-1-4; B-1-5; B-1-6; B-2-1; B-2-3; B-2-4; B-2-5; B-2-6; B-3-1; B-3-3; B-3-4; B-3-5; B-3-6; B-4-1; B-4-3; B-4-4; B-4-5; B-4-6
0.01	kg cm^2/mg-m^2	Units conversion factor	
10^{-6}	g/µg	Units conversion factor	
10^{-6}	kg/mg	Units conversion factor	
0.31536	m-g-s/cm-µg-yr	Units conversion factor	
365	days/yr	Units conversion factor	
907.18	kg/ton	Units conversion factor	
0.1	g-kg/cm^2-m^2	Units conversion factor	
0.001	g/mg	Units conversion factor	
100	mg-cm^2/kg-cm^2	Units conversion factor	
1000	mg/g	Units conversion factor	
4047	m^2/acre	Units conversion factor	
1×10^3	g/kg	Units conversion factor	
3.1536×10^7	s/yr	Units conversion factor	

Chapter 1: INTRODUCTION

What's Covered in Chapter 1:

1. 1 Objective and Document Organization

1. 2 Background

1. 3 Using this Document

1. 4 Primary Reference Documents

1. 5 Risk Nomenclature

1.1 OBJECTIVE AND DOCUMENT ORGANIZATION

The U.S. Environmental Protection Agency's ("U.S. EPA" or "the Agency") Office of Solid Waste (OSW) has developed an approach for conducting multi-pathway, site-specific human health risk assessments on Resource Conservation and Recovery Act (RCRA) hazardous waste combustors. The approach, also known as the Human Health Risk Assessment Protocol ("HHRAP" or "protocol") can be used where the permitting authority determines such risk assessments are necessary. The HHRAP replaces an earlier Peer Review Draft published in July 1998.

> ***PLEASE NOTE***: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.
>
> The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

Our primary objective in developing the protocol was to offer a user-friendly approach to performing site-specific combustion risk assessments. We wanted to develop a guidance document that would:

- be useful to a diverse group of users: risk assessors, permit writers, risk managers, and community relations personnel;

- provide a logical method for doing risk assessments of facilities that burn hazardous waste;
- completely explain the reason for each recommended procedure and parameter value;
- provide a comprehensive enough collection of default input parameters to conduct a risk assessment. The collection would also be flexible enough to accommodate regional or site-specific information; and finally
- make sufficient tools available to produce transparent, defensible, and realistic results. When coupled with an ecological assessment (U.S. EPA 1999a), these tools would provide critical information often needed by risk managers when faced with the decision of permitting a hazardous waste combustion facility.

The HHRAP brings together information from other risk assessment guidance and method documents prepared by U.S. EPA and state environmental agencies. It also contains the latest advancements in risk assessment science, as well as experience EPA has gained through conducting and reviewing combustion risk assessments. This version of the protocol also addresses the comments put forward by the public and external scientific peer reviewers regarding earlier drafts of the HHRAP.

The first volume of the HHRAP contains the main body of the document, providing a detailed explanation of a risk assessment approach that we recommend you consider when conducting a risk assessment. The second volume contains the appendices, including:

Appendix A: a collection of chemical-specific information which might be of interest -

- A-1 - a compilation (from various EPA sources) of compounds of potential interest;
- A-2 - Compound-specific parameter value information. Details the sources of, or equations used to calculate, parameter values used in fate & transport-, biotransfer-, exposure-, and toxicity-related equations. Parameter values themselves are found in a companion database to the HHRAP, which is also available for download.

Appendix B: Equations and recommended default variable values for estimating media concentrations; and

Appendix C: Equations and recommended default variable values for estimating potential cancer risk and non-cancer health effects.

> ***Please Note***: Although these guidelines address many types of situations encountered in the field, they cannot encompass every potential situation. You should ensure that the recommendations in this guidance are appropriate for the facility, based on site-specific information and/or circumstances.

This protocol is a "snapshot" of current risk assessment science, and we encourage you to evaluate updates and alternatives to the recommended parameters (e.g., toxicological benchmarks; exposure factors) when they become available. If you use alternative values, however, keep in mind how changes in one parameter may affect other parameter values and/or calculations. We may revise the protocol in the future if any of the following become available:

- new research in risk assessment science and/or the combustion field;
- new information gathered while conducting site-specific risk assessments; and
- new initiatives introduced by the Agency.

These types of changes are inevitable in this evolving and highly technical field.

1.2 BACKGROUND[1]

Hazardous waste combustors are required to meet RCRA national performance standards and obtain a permit under 40 CFR Part 264, Subpart O; Part 266, Subpart H; and Part 270.[2] In addition, Section 3005(c)(3) of RCRA [codified at 270.32(b)(2)] requires that each permit contain the terms and conditions that the permitting authority considers necessary to protect human health and the environment. This is commonly referred to as the "omnibus authority." The omnibus authority gives the Agency both

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[1] This section summarizes the historical context of regulatory authority and associated policy regarding hazardous waste combustion site-specific risk assessment under the RCRA program. This discussion is not intended to update, revise or articulate new guidance or policy. Nor is it intended to update, revise or provide any new interpretations of any statutory or regulatory authority, including those relevant to the RCRA 3005(c)(3) "omnibus authority". In addition, it is not intended to reopen for consideration any statutory or regulatory interpretations of other related guidance documents, or the MACT rule (64 FR 52828).

[2] When combustion sources demonstrate compliance with the Part 63, Subpart EEE MACT standards, they may request that certain RCRA permit conditions (e.g., those based on the national performance standards) be removed from their RCRA permit, because they are no longer applicable, through a class 1 permit modification request with prior agency approval. In some cases, RCRA performance standards may be retained in the RCRA permit if they are more stringent than the relevant MACT standard.

the authority and the responsibility to set up site-specific RCRA permit conditions as necessary to "be protective of human health and the environment." These permit conditions are intended to supplement, not replace those conditions that are already required under the national performance standards [See Federal Register 1999 (MACT Rule)].

The RCRA national performance standards for incinerators were published in 1981 (40 CFR Part 264, Subpart O) and for boilers and industrial furnaces in 1991 (40 CFR Part 266, Subpart H). Since then, however, new information on indirect exposure pathways and non-dioxin products of incomplete combustion (PICs) suggests that these standards may not fully address all potentially significant risks.

For example the RCRA national standards were based on estimates of risk only from ***direct*** exposure to (i.e. inhaling) stack emissions. New information suggests risks from ***indirect*** exposures (e.g. ingesting contaminated soil, food, or water) are also important (Federal Register 1999). Bioaccumulation tends to concentrate some chemicals as they migrate through the environment. These higher concentrations can lead to exposures and risks of concern. For example, Fradkin et al. (1988) linked elevated levels of chemical pollutants in soils, lake sediments, and cow's milk to atmospheric transport and deposition of pollutants from combustion sources. Also, the *1994 Draft Health Reassessment of Dioxin-Like Compounds* (U.S. EPA 1994a), and the 1997 *Mercury Study Report to Congress* (U.S. EPA 1997c), indicate that indirect exposure pathways can lead to significant risks.

Indirect exposure pathways weren't directly taken into account by the 1981 and 1991 hazardous waste combustion standards. The regulations also didn't take into account the uncertainty surrounding the types and quantities of non-dioxin products of incomplete combustion (non-dioxin PICs) or any potential risks posed by these compounds.

To address these concerns, the Agency issued the *Hazardous Waste Minimization and Combustion Strategy* in 1994. This strategy recommended conducting a site-specific risk assessment for each combustion facility seeking a RCRA permit. Permitting authorities could use the results of an assessment to determine, on a case-by-case basis, if a combustor operating in accordance with the performance standards is protective of human health and the environment. If the permitting authority finds that the combustor operating in accordance with the performance standards is not protective of human health and the environment, the permitting authority would invoke the "omnibus authority" and either add additional conditions to the RCRA permit, or deny the RCRA permit. The permitting authority must explain the reasons for any additional permit conditions in the administrative record of the facility (Federal Register 1999).

In 1999 the Agency updated its earlier site-specific risk assessment recommendation, to account for the Phase 1 Maximum Achievable Control Technology (MACT) standards for hazardous waste incinerators, cement kilns, and lightweight aggregate kilns (see 64 FR 52828). While the Phase 1 MACT standards provide additional protection, we recognize that there may continue to be circumstances for which site-specific risk assessments are appropriate. For example, a site-specific risk assessment might be appropriate if there is reason to believe that operating in accordance with Phase 1 MACT standards alone may not be protective of human health and the environment. So, in the MACT standards rulemaking, we recommend that the permitting authority evaluate the need for a site-specific risk assessment on a case-by-case basis. For hazardous waste combustors not subject to the Phase 1 MACT standards, such as boilers and industrial furnaces, we continue to recommend that site-specific risk assessments generally be conducted as part of the RCRA permitting process (see Federal Register 1999).

As part of the September 2005 rule finalizing MACT standards for hazardous waste-burning incinerators, cement kilns, lightweight aggregate kilns, boilers, and industrial furnaces, we maintain virtually the same the site-specific risk assessment policy recommendation as conveyed in the 1999 final rule preamble (see previous paragraph)[3]. That policy, which establishes that the need for an SSRA should be determined on a case-by-case basis, now applies equally to both Phase 1 and Phase 2 sources.

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[3] The standards for Phase 1 sources (incinerators, cement kilns, and lightweight aggregate kilns) are referred to as Replacement Standards. The Replacement Standards replace the February 13, 2002 Interim Standards that were developed in response to a court decision to vacate challenged portions of the 1999 Phase 1 MACT final standards. Thus, the 2005 final rule establishes MACT standards for both Phase 1 and Phase 2 (boilers and industrial furnaces) sources.

In addition, the 2005 final rule codifies additional regulatory language that provides the authority for SSRAs. Although a comparative risk analysis conducted for the 2005 final rule concluded that the MACT standards for both Phase 1 and Phase 2 sources are generally protective, there may be instances where we cannot be assured that emissions from each source will be protective of human health and the environment. Because we believe that SSRAs are likely to continue to be necessary at some facilities (i.e., mainly those that have not previously conducted an SSRA), we have codified language in §§270.10(l) and 270.32(b)(3) that explicitly provides for the permit authority to require SSRAs on a case-by-case basis and add conditions to RCRA permits based on SSRA results, respectively. The language also reminds permit authorities that the determination that the MACT standards may not be sufficiently protective is to be based only on factors relevant to the potential risk from the hazardous waste combustion unit at the site. Additionally, guiding factors have been identified for permitting authorities to consider in determining whether the MACT will be sufficiently protective at an individual site. In summary, the 2005 final rule only modifies the statutory authority under which we implement the SSRA policy, while maintaining the same SSRA policy from a substantive standpoint.

1.3 USING THIS DOCUMENT

This document contains our generally recommended approach for conducting multi-pathway, site-specific human health risk assessments of RCRA hazardous waste combustors. This document *does not* provide recommendations on how to:

- ***Determine if a site-specific risk assessment should be performed***;

  You can find U.S. EPA's most recent recommendations for when, or if, a site-specific risk assessment should be performed, in documentation of the MACT rule, published on September 30, 1999 (Federal Register 1999).

- ***Conduct stack emissions testing for a site-specific assessment***;

  A separate guidance document entitled *Risk Burn Guidance for Hazardous Waste Combustion Facilities* EPA 530-R-01-001, July 2001 (U.S. EPA 2001c) contains approaches for collecting emissions data to support site-specific risk assessments. This document is on the U.S. EPA OSW website at: http://www.epa.gov/epaoswer/hazwaste/combust/pdfs/burn.pdf.

- ***Develop a site-specific ecological risk assessment***;

  We'd previously published our recommendations for conducting screening level ecological combustion risk assessments in a separate, companion document to the HHRAP. This companion document, the *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (SLERAP) Peer Review Draft (U.S. EPA 1999a), is currently undergoing substantial revision. Until revisions are complete, we can't recommend using the SLERAP.

- ***Use risk assessment results in risk management decisions, such as setting RCRA permit conditions***.

  Because this protocol is a technical risk ***assessment*** tool, it does not discuss risk ***management*** issues, such as how risk managers are to use the provided information (including uncertainty information), the potential for cumulative risks, or target risk levels. U.S. EPA's generally recommended risk and hazard targets can be found in *Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities* (U.S. EPA 1994f). Additionally, EPA Region 6's region-specific risk target recommendations, *Region 6 Risk Management Addendum* (U.S. EPA 1998b), are available on their website at: www.epa.gov/earth1r6/6pd/rcra_c/protocol/r6add.pdf.

  Please Note: the ultimate decision for how to incorporate risk assessment estimates in risk management decisions rests with the permitting authority.

For your convenience, the HHRAP many of the recommendations found in the above-referenced documents. However, unless we say so explicitly, the HHRAP does not intend to update, revise, or replace any of the information contained in the above-referenced documents.

The HHRAP ***does*** update and replace the following guidance documents:

- U.S. EPA, *Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes,* April 15, 1994 Draft and the October 4, 1994 Errata;
- U.S. EPA, *Protocol For Screening Level Human Health Risk Assessment at Hazardous Waste Combustion Facilities - Volumes 1 & 2,* Internal Review Draft, EPA-R6-096-002 February 28, 1997;
- U.S. EPA, *Human Health Risk Assessment Protocol,* Draft Interim Final, April 1998 (CD-Rom version);

- U.S. EPA, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* - *Volume 1-3*, Peer Review Draft, EPA 530-D-98-001A, B & C, July 1998 and the August 2, 1999, Errata.

We anticipate that risk assessments will be completed for new and interim-status facilities, where necessary, when they apply for their RCRA permit. The process we recommend evaluates risks to receptors posed by potential emissions from RCRA-regulated units. We encourage you to use existing and site-specific information throughout the risk assessment process in order to properly evaluate actual regulated operations for any particular combustor. We generally recommend conservative default assumptions only when they will provide confidence that ensuing permit limits will be health protective.

Throughout the HHRAP we offer parameter values for you to consider. These values are based on a number of elements, such as the best science available and professional judgement. Since this is a national level guidance, the recommended values typically reflect national average conditions. The values will be more appropriate for some sites, and less so for others. For example, the type of waterbody near a facility (i.e. lake, river, wetland) may affect the methylation rate of mercury in the waterbody, or the type of fish consumed may affect percent lipid content used in the assessment. So, a value that is reasonable for one facility may be over (or under) protective at a different facility.

In all cases, though, we give the reason for the suggested value. We encourage you to consider our reasoning, in deciding if a more representative estimate of a site-specific value (or range of values) is available and appropriate. If you use values other than those we recommend, you should explore how, or if, those changes may affect other parameter values and calculations used in the assessment. As with values recommended in this guidance, using values other than those recommended here should always be clearly identified and discussed in the risk plan and/or risk assessment (as appropriate). This will ensure clarity and transparency of the final risk assessment results.

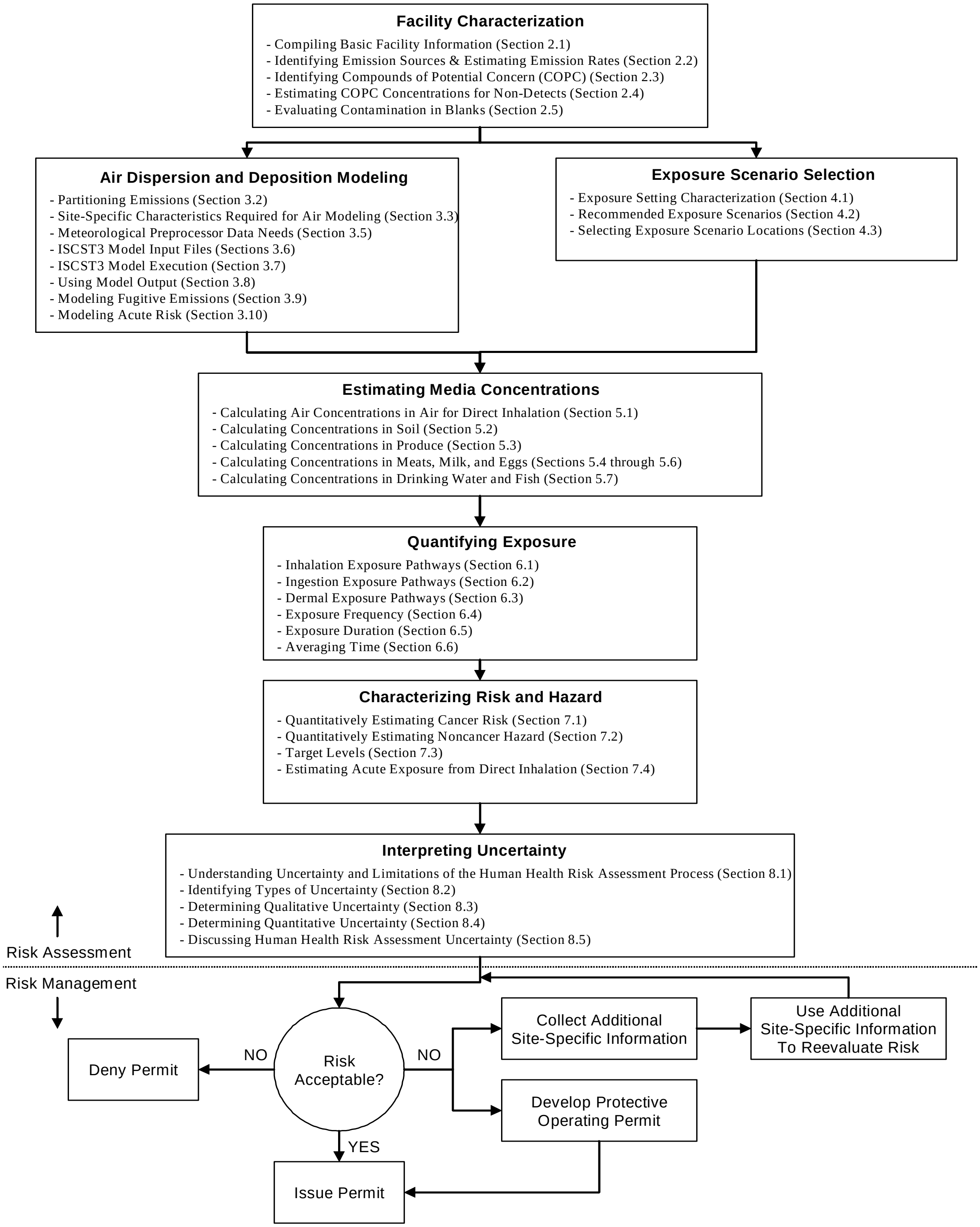
You would need considerable time, effort, and funding to investigate the representativeness of all the values (or ranges of values) available in the HHRAP. As a result, you might choose to use only readily available site-specific information in an initial assessment. You could then use the results of that assessment to determine where (or if) more site-specific risk information should be collected (see Figure 1-1). This allows you to use resources most efficiently and effectively, by focusing resources on areas that are considered "risk drivers", rather than areas that do not appreciably affect the risk outcome. For example, if the assessment shows that the primary pollutant and exposure pathway is mercury in fish, then you could target site-specific data gathering efforts on values related to mercury emissions, surface water concentrations and/or fish consumption. You would not have to spend resources collecting site-specific information that may not affect the final results of the assessment (for example, Manganese exposure through ingestion of produce).

You can also use the HHRAP as a screening tool. For example, a facility with a highly variable waste stream might choose to provide the permitting authority with historical data and assume that all compounds will be retained in the risk analysis. Or you might choose to use more conservative assumptions throughout, to make the assessment fit a more classic "screening level" approach. For example, you could choose *not* to initially investigate the actual land use surrounding a facility, but instead locate all the selected receptors at the area of greatest contaminant deposition. If estimates don't exceed the selected risk target, additional iterations of the assessment may not be necessary. Regardless, every risk assessment is limited by the quantity and quality of:

- Site-specific environmental data;
- Emission rate information; and
- Other assumptions made during the risk estimation process (e.g., fate and transport variables, exposure assumptions, and receptor characteristics).

These limitations and uncertainties are described extensively throughout the main document and the appendices, and are summarized in Chapter 8. You should generally make every effort to reduce limitations and uncertainties in the risk assessment process, since they can affect the confidence in the risk assessment results.

FIGURE 1-1
HUMAN HEALTH RISK ASSESSMENT PROCESS



The EPA Information Quality Guidelines (U.S. EPA 2002c) recommend ensuring the objectivity of information found in risk assessments by applying, to the extent practicable and consistent with Agency statutes and existing legislative regulations, the following adaptation of the quality principles found in the Safe Drinking Water Act (SDWA) Amendments of 1996:

> (A) The substance of the information is accurate, reliable and unbiased. This involves the use of:
>
> > (i) the best available science and supporting studies conducted in accordance with sound and objective scientific practices, including, when available, peer reviewed science and supporting studies; and
> >
> > (ii) data collected by accepted methods or best available methods (if the reliability of the method and the nature of the decision justifies the use of the data).
>
> (B) The presentation of information on human health, safety, consistent with the purpose of the information, is comprehensive, informative, and understandable. In a document made available to the public, EPA specifies:
>
> > (i) each population addressed by any estimate of applicable human health risk;
> >
> > (ii) the expected risk for the specific populations affected;
> >
> > (iii) each appropriate upper-bound or lower-bound estimate of risk;
> >
> > (iv) each significant uncertainty identified in the process of the assessment of risk; and
> >
> > (v) peer-reviewed studies known to the Administrator that support, are directly relevant to, or fail to support any estimate of risk and the methodology used to reconcile inconsistencies in the scientific data.

How risk results are viewed by the risk manager and other stakeholders is complex and can involve other factors besides those included in this document (e.g. public concern). Consequently, interpreting risk assessment results warrants careful consideration. Risk management decisions are beyond the scope of the HHRAP, and we don't provide any guidance on interpreting risk results. It should be noted, though, that identifying potentially unacceptable risks does not necessarily signify the end of the risk assessment process. You can view risk assessments as an iterative process[4], with a number of available options once risk estimates are produced. The iterative nature of the risk assessment/risk management interface is graphically represented in Figure 1-1, and the various available options are briefly described below:

---

[4] As stated in the U.S. EPA (2002c) "Risk assessments may be performed iteratively, with the first iteration employing protective (conservative) assumptions to identify possible risks. Only if potential risks are identified in a screening level assessment is it necessary to pursue a more refined, data-intensive risk assessment. The screening level assessments may not result in "central estimates" of risk or upper and lower-bounds of risks. Nevertheless, such assessments may be useful in making regulatory decisions..."

Example 1: If the initial risk estimates (coupled with any other related factors) indicate that risks are not expected to pose a concern to human health or the environment, the risk manager and/or permit writer will likely end the site-specific risk assessment process and the facility will likely receive a permit.

Example 2: If the initial risk estimates (coupled with any other related factors) indicate that the risks are at or above a level that may pose a risk to human health or the environment, then additional information might be added to the risk assessment (e.g. site-specific information that's more representative of the actual exposure settings). Additional iterations of the risk assessment could then be performed. This iterative process enables you to determine if the risks identified in the earlier assessment accurately represent the situation at a given combustion facility.

Example 3: If the initial risk estimates (coupled with any other related factors) or subsequent iterations (as detailed in Example 2 above), indicate potentially unacceptable risk, risk managers and/or permit writers might use the results of the risk assessment to propose revised or additional permit conditions (such as waste feed limits and/or process operating conditions) to lower the potential risk to acceptable levels. Another risk assessment could verify that the proposed permit conditions will enable the combustor to operate in a manner that's protective of human health and the environment.

In some situations, target risk levels might be selected and back-calculations conducted to determine what emission and/or waste feed rate would allow the facility to operate in a protective manor. In any case, the acceptable waste feed rate and other appropriate conditions could be incorporated into the RCRA permit.

Example 4: If the initial risk estimates or subsequent iterations (coupled with any other related factors) indicate potentially unacceptable risk, risk managers and/or permit writers might also choose, where appropriate, to deny the permit.

The HHRAP may also be useful when a facility or regulatory agency decides to perform a pre-trial burn risk assessment. A pre-trial burn risk assessment can evaluate pre-existing permit limits (e.g. regulatory limits such as MACT or BIF) to determine if more extensive or refined risk-based testing is necessary as part of the trial burn testing program. Also, the pre-trial burn risk assessment can test the parameters used in the initial trial burn sampling and analysis plan. Testing trial burn parameters minimizes trial burn contributions to risk assessment uncertainty, and avoids the expense of multiple trial burn iterations. For example, if the initial detection or quantitation limit for a specific compound (such as a dioxin, furan, or bioaccumulative metal) is too high during trial burn sampling and analysis, then the final risk estimate may be artificially inflated, especially for indirect exposure pathways. If trial burn sampling and analysis uses a ***lower*** detection or quantitation limit, the compound might be found not to add appreciably to the risk results. The pre-trial burn risk assessment can also determine whether modifications to the sampling collection (such as increased sample volumes) are needed to achieve lower detection or quantitation limits. Please see Chapter 2 for more detailed information on how risk assessments relate to trial burns.

1.4 PRIMARY REFERENCE DOCUMENTS

One of the main benefits of the HHRAP is that it assembles in one place more than a decade of research and experience regarding practices for conducting risk assessments of hazardous waste combustion facilities. This section describes, in chronological order, the ***primary*** guidance documents we used to prepare the HHRAP. Many other important reference materials were also needed to produce a document of this magnitude. We have listed *all* reference materials used in preparing this document in the Reference chapter.

Some of the documents we used were themselves developed over a period of several years, including revisions. In some cases, revisions to the original document address only specific issues rather than a complete revision of the original document. The following discussion lists and briefly describes each document. Overall, the guidance documents listed below reflect a continual refining and enhancing of the risk assessment method.

The following was the first U.S. EPA guidance document for conducting risk assessments at combustion units:

> **U.S. EPA. 1990e**. *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions, Interim Final*. Environmental Criteria and Assessment Office. ORD. EPA-600-90-003. January.

Referred to as the “IEM” document, EPA (1990e) outlines and explains a set of general procedures for conducting risk assessments that includes both the direct inhalation pathway and indirect food chain pathways. The IEM document was subsequently supplemented by the following:

U.S. EPA. 1993f. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions, Review Draft*. Office of Health and Environmental Assessment. ORD. EPA-600-AP-93-003. November 10.

Referred to as the "Addendum", EPA (1993f) outlines recommended revisions and added new exposure pathways to the previous U.S. EPA guidance (1990e), and has been used by the risk assessment community since its release.

In 1994, we issued several additional hazardous waste combustion risk assessment documents, including:

U.S. EPA. 1994f. *Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. OSWER. EPA-530-R-94-021. April.

This document (1994f) is made up of a series of four attachments, all issued around the same time frame (April/May 1994) as separate documents:

U.S. EPA. 1994g. *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. 1994h. Table 1, "Chemicals Recommended for Identification," and Table 2, "Chemicals for Potential Identification." *Attachment A, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 15.

U.S. EPA. 1994i. *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. 1994j. *Draft Guidance on Trial Burns. Attachment B, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. May 2

Combined, these four documents present a generally recommended procedure for sampling the combustor emissions, identifying the compounds of concern, conducting both a direct and indirect risk assessment, and implementing the results of the risk assessment for hazardous waste combustion facilities. We used the methodologies identified in both the ORD "IEM" and "Addendum" documents as the foundation of our hazardous waste combustion risk assessment methodology. The "IEM" and the "Addendum" were broader in scope than our document. We used many, but not all, of the methods, models and exposure scenarios that are described in the two ORD documents. Because the ORD documents contain much of the background information necessary to complete a risk assessment, that information was not repeated in our guidance documents. Shortly after the release of our documents, the trial burn portion and the risk assessment portion were further revised with the following releases:

> **U.S. EPA. 1994n**. *Draft Revision of Guidance on Trial Burns. Attachment B, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. OSWER. June 2.
>
> **U.S. EPA. 1994p**. *Errata, 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*. October 4.

As a follow-up to these documents, we prepared another draft guidance. The following was released for internal review, but never formally or officially released as a program-supported document:

> **U.S. EPA. 1994r**. *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. OSW. December 14.

In 1997, the state of North Carolina's Department of Environment, Health, and Natural Resources (DEHNR) developed the following guidance document for conducting risk assessments in their state:

> **NC DEHNR. 1997**. *North Carolina Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

The NC DEHNR document reiterates U.S. EPA procedures (1994r), with the addition of a tiered approach that can help the regulatory agency or facility to choose approaches that reflect the investment they want to make in conducting risk assessments. For instance, a small, on-site unit with limited waste stream variability may find the first tier assessment (worst-case) in the North Carolina protocol appropriate, whereas a larger facility with a diverse waste feed mixture may decide to complete a Tier 2 or Tier 3 assessment, which are progressively more site-specific.

In 1998, the ORD revised, updated and combined the "IEM" and the "Addendum" documents into one document, entitled:

> **U.S. EPA 1998**. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (U.S. EPA 1998c).

This document is referred to as the "MPE" document. It includes information which was gained from cross-Agency review, EPA's Science Advisory Board (SAB) and the public on the "IEM" and the "Addendum" documents. It also includes information from the draft dioxin reassessment "*Estimating Exposure to Dioxin-Like Compounds*" (U.S. EPA 1994a) and the "*Mercury Study Report to Congress*" (December 1997). As with the MPE's predecessor documents, it is considered the foundation of our hazardous waste combustion risk assessment methodology and is frequently referenced in the HHRAP.

In 1999 we released a technical document that detailed the risk assessment conducted to support the hazardous waste combustion Maximum Achievable Control Technology (MACT) standards:

> **RTI 1999**. *The Background Information Document to the Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Wastes* Final Report. EPA Contract Number 68-W6-0053.

To ensure consistency, we considered EPA(1999) throughout the development of the HHRAP.

Finally, in 2001 we updated and finalized the document entitled:

> **U.S. EPA 2001c**. *Risk Burn Guidance for Hazardous Waste Combustion Facilities,* July.

Referred to as the "risk burn guidance," EPA(2001c) was prepared by U.S. EPA Region 4 and U.S. EPA OSW. It details recommendations regarding stack emissions tests which may be performed at hazardous waste combustion facilities to support site-specific risk assessments.

As previously stated, our primary objective in developing the HHRAP was to suggest a user-friendly approach to performing site-specific combustion risk assessments. The HHRAP achieves this goal by offering a comprehensive set of tools. You will no longer need to search through a long list of guidance documents to find an appropriate method and/or value when conducting a site-specific risk assessment of a hazardous waste combustor. Instead, you have one self-contained document with the majority of all the available information needed to complete a risk assessment. With the HHRAP's extensive reference list, you also have the original source of a method and/or value. This simplifies the process of deciding if the reference is appropriate to use for your specific situation.

1.5 RISK NOMENCLATURE

Unless otherwise stated, the following definitions for risk-related terms are from the National Academy of Sciences 1983, *Risk Assessment in the Federal Government*, and used 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 chemicals of potential 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	The amount of a substance available for interaction with metabolic processes or biologically significant receptors after crossing the exchange boundary of an organism (U.S. EPA 1998c).
Exposure	The condition of a chemical contacting the exchange boundary of an organism (U.S. EPA 1998c).
Indirect Exposure	Resulting from contact of human and ecological receptors with soil, plants, or waterbodies on which emitted chemical has been deposited. For screening level purposes, indirect exposure includes ingestion of above ground fruits and vegetables, beef and milk, chicken and eggs, freshwater fish and soil.
Direct Exposure	Exposure via inhalation.

Chapter 2:
Characterizing Facility Emissions

What's Covered in Chapter 2:

2.1 Compiling Basic Facility Information

2.2 Identifying Emission Sources & Estimating Emission Rates

2.3 Identifying Compounds of Potential Concern (COPCs)

2.4 Estimating COPC Concentrations for Non-Detects

2.5 Evaluating Contamination In Blanks

This chapter provides guidance on characterizing the nature and magnitude of facility emissions. Characterizing includes (1) compiling basic facility information, (2) identifying emission sources, (3) estimating emission rates, (4) identifying COPCs, (5) estimating COPC concentrations for non-detects, and (6) evaluating contamination in blanks. You can consider the information listed in the highlighted box at the end of each section the minimum that we recommend to ensure a risk assessment is scientifically sound. However, you may want to consult up front the more detailed discussions found in each section. A more complete understanding of the relevant issues will make sure that all appropriate information is collected simultaneously. This will help minimize the time and effort expended collecting site-specific information.

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

2.1 COMPILING BASIC FACILITY INFORMATION

If you are a risk assessor, there is basic facility information you should consider while conducting the risk assessment, and include in the risk assessment report. Including this basic facility information in the report will enable reviewers to establish a contextual sense of how the facility relates to other facilities and other hazardous waste combustors. It's also very important to thoroughly understand (and document) any regulatory limits evaluated in the risk assessment, because the risk assessment report informs the setting of risk-based permit limits. For example, specific emissions data might not be collected for a particular unit where waste feeds are controlled in lieu of demonstrating compliance with an emissions limit under the regulations (e.g., Tier I under the BIF rule for certain metals, MTEC under the HWC MACT rule). For transparency, we therefore recommend clearly identifying the basis for the assumptions and/or data to be used in the risk assessment, along with the rationale for how the information will be used.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

C Description of physical setting

C Principal business and primary production processes

C Normal and maximum production rates

C Types of waste storage and treatment facilities

C Type and quantity of wastes stored and treated

C Process flow diagrams showing both mass and energy inputs and outputs

< Relevant information from an existing or proposed permit and/or compliance documents (e.g., Waste Analysis Plan or Feedstream Analysis Plan; Startup, Shutdown, and Malfunction Plan; Certifications of Compliance; etc.)

2.2 IDENTIFYING EMISSION SOURCES & ESTIMATING EMISSION RATES

Burning hazardous waste typically emits combustion by-products from a stack. In addition to emissions from a combustion stack, types of emissions associated with the combustion of hazardous waste may include (1) process upsets emissions, (2) accidental releases, (3) general RCRA fugitive emissions, and if the facility is a cement kiln (4) cement kiln dust (CKD) fugitive emissions. Each of these emission source types is defined below with regards to the context and scope of this guidance.

> ***Stack Emissions* -** Release of compounds or pollutants from a hazardous waste combustor into the ambient air while the unit is operated as intended and in compliance with a permit and/or regulation (for interim status).

> ***Process Upset Emissions* -** Release of compounds or pollutants from a hazardous waste combustor into the ambient air while the unit is not being operated as intended, or during periods of startup or shutdown. Upset emissions usually occur during events and times when the unit is not operating within the limits specified in a permit or regulation. Conditions within the combustion system during the process upset result in incomplete destruction of the wastes, or otherwise promote the formation and/or release of hazardous compounds from combustion stacks. Upset emissions are generally expected to be greater than stack emissions.
>
> ***Accidental Releases* -** an accidental release is defined in Section 112(r) of the Clean Air Act as an unanticipated emission of a regulated substance or other extremely hazardous substance into the ambient air from a stationary source. Accidental releases are typically associated with non-routine emissions from RCRA facilities, such as the failure of tanks or other material storage and handling equipment, complete failure of combustion and air pollution control systems (e.g. resulting from explosions, and fires), or transportation accidents.
>
> ***RCRA Fugitive Emissions* -** Release of compounds or pollutants into the ambient air from RCRA-regulated sources other than hazardous waste combustion stacks. RCRA fugitive emissions are typically associated with the release of pollutants from leaks in the combustion chamber (e.g., “puffs”); tanks, valves, flanges, and other material-handling equipment used in the storage and handling of RCRA hazardous wastes; residues from the combustion process such as ash or quench water; and other RCRA treatment, storage, or disposal units (e.g., landfills).
>
> ***CKD Fugitive Emissions* -** Release of pollutants into the ambient air caused by the handling, storage, and disposal of cement kiln dust.

We generally recommend that as applicable, all of these emission source types except accidental releases be addressed in the risk assessment. Accidental releases aren’t within the scope of this guidance. We generally recommend evaluating accidental releases per Section 112(r) of the CAA and current Agency guidance (U.S. EPA 1996f) or the *RMP Offsite Consequence Analysis Guidance*, dated May 24, 1996. Despite this general guidance, it is for the permitting authority to decide on a site-specific basis whether the risk assessment will consider accidental releases.

The following subsections contain guidance for estimating emissions of the source types to be included in the risk assessment. Guidance on air dispersion modeling of stack and fugitive emissions is presented in Chapter 3.

2.2.1 Estimating Stack Emission Rates for Existing Facilities

We generally consider it important to determine stack emission rates (in grams per second) of every COPC identified using the procedures outlined in Section 2.3. We anticipate that emission rates for existing facilities (i.e. already built and operational) will be based on direct stack measurements from regulatory performance tests, because permitting authorities generally require performance tests before granting a permit to burn hazardous wastes, or in order to demonstrate compliance with emission standards.

As mentioned in the Special Note "How This Document Relates to Trial Burns" and elsewhere, we suggest incorporating data collection for a risk assessment into the regulatory performance testing program whenever possible. This will optimize the use of both facility and permitting agency resources, and minimize expenditures associated with stack testing and subsequent data review, evaluation, and permitting. Incorporating risk assessment data collection into the performance test program can ensure that proper evaluation across test conditions is achieved for optimal data usability and versatility– both engineering considerations relating to unit operations flexibility, and streamlined data collection considerations for characterizing potential emissions.

Test Design: Performance vs. Health Criteria

Performance testing involves demonstrating compliance with emission standards or performance criteria under any operational circumstance. Facilities typically wish to maximize the flexibility of operating conditions allowed by their permit. This leads to performance tests designed to demonstrate compliance even while operating under extreme conditions (i.e., conditions that may only seldom occur or may only occur for brief durations over the course of a year).

Regulatory performance requirements are generally **national-scale**, technology-based **instantaneous** limits. Such performance requirements might not consider various **health impacts** to receptors located near **a particular facility**. Therefore, data needs to assess potential health risks are typically more comprehensive than those of a performance test. For example, a performance test will provide information to calculate destruction and removal efficiency (DRE) for a principal organic hazardous constituent (POHC) and perhaps measure total particulate matter, but will not typically quantify the various products of incomplete combustion (PICs) in the stack gases, nor establish particle size distribution for gases exiting the stack.

Situations have occurred in which some facilities assume that the only way to maximize permit flexibility and satisfy risk assessment data needs is to conduct separate tests. We encourage another interpretation. As mentioned elsewhere, we suggest modifying performance-based test protocols to include risk assessment data requirements as appropriate to evaluate both acute and chronic exposures posed by facility-specific operations that may occur under the terms of the permit.

Experience has shown us that in order to evaluate both acute and chronic reasonable maximum exposure estimates, the potential emissions evaluated in the risk assessment need to be based on actual operating scenarios that may occur under the terms of the permit. We acknowledge that proper design of a regulatory test program that includes risk assessment data collection is challenging for facilities that burn highly variable waste and/or have multiple operating conditions. At the same time, inadvertently omitting potential risk drivers from the risk assessment due to improper or oversimplified test design is to be avoided. Therefore, if feed streams differ between the various test conditions in the regulatory test program, we generally recommend providing appropriate rationale for this difference in the test plan, and discussing in the risk assessment report any impacts the differences may have had upon the risk analysis.

In some cases, a facility may elect to focus data collection efforts for a risk assessment into one test condition in the regulatory testing program that demonstrates "normal operations" using feedstreams considered "worst-case" from an operational and potential emissions standpoint.

> ***Please Note***: We suggest such a test condition only for facilities that can identify and substantiate via historical operating records the typical (or day-to-day) operating mode for the hazardous waste combustion unit and ancillary equipment, regardless of the type of waste fed.

If combining multiple and variable feedstreams into a single "worst-case" feed, the mixture needs to represent those actual waste matrices and constituents that are the most difficult to burn (ensuring that the combustion unit is fairly challenged). A "worst-case" feed also needs to contain the most toxic substances managed in the unit, on a mass basis proportional to that fed at any time (ensuring that all potentially toxic emissions are quantified). Multiple and/or highly variable feedstreams that require different extremes of a wide operating envelope, or feedstreams that are combined on a disproportional mass basis to that fed at any time, may not be representative of actual operations. We're concerned that they might even result in emission estimates that are not sufficiently conservative for the risk assessment. Please see the Risk Burn Guidance for Hazardous Waste Combustion Facilities (U.S. EPA 2001c) for further information on designing a testing program that integrates risk assessment data collection into the regulatory testing of hazardous waste combustors.

<u>Test Conditions' Relation to Permit Conditions</u>

The feed and operating conditions demonstrated during the testing will define an operating envelope that not only establishes the working assumptions for the risk assessment, but also the final permit terms. If a facility collects emissions data for the risk assessment under "normal" operating conditions, additional permit limitations may be appropriate to ensure that conditions represented as normal during the test are, in fact, normal over the long-term operation of the facility.

We generally recommend including the risk assessor in the early planning efforts of the test program development, so that the test program can be more effectively streamlined while meeting multiple and complex data collection goals. At least three valid runs at steady state are needed to characterize a test condition. Since steady state conditions are typically outlined in the test plan for each test condition and verified by field observations during the test program, resulting emission rates for any particular COPC are likely to be fairly consistent between runs for each test condition. Test reports document any abnormalities in steady state operations for a particular run or test condition. If any one run experiences significant issues that may impact the data quality or comparability with the other runs, the facility and permitting authority might decide during the test to discontinue the problem run and initiate a new run to ensure a valid test condition. Test reports also document this occurrence and identify those valid runs of the test condition.

For risk assessments, we generally recommend using the maximum of the three emission rates identified for each COPC during a particular test condition, adjusted for process upsets. This approach is consistent with implementing a steady state test designed for collecting risk data where the combustor burns representative, yet worst-case or challenging feeds, at operating conditions that are allowable and/or typical under the permit. This approach will also allow consistency in refinements to the final risk analysis that may involve several risk evaluations for different operating scenarios (i.e., across the various test conditions) in order to afford operational flexibility while maintaining permit provisions that ensure protection of human health.

> ***Please Note***: The recommendation to use the highest of the three emission rates is an update to Section 8.1.2 of U.S. EPA (2001c).

An alternative to a regulatory performance test is the use of data "in lieu of" testing. Permitting authorities generally consider this type of data on a case-by-case basis. Prior to accepting such data as a surrogate for use in the risk assessment, we recommend evaluating the data from both an engineering perspective and a data usability perspective. To evaluate the similarities between combustors, consider the design and construction of the combustor and associated air pollution control devices, along with the basic operating conditions of the process equipment as tested (e.g., capacity, flow rates, supplemental fuels used, etc.) to ensure comparable emissions. Stack test measurements from a similar combustor are useful if the combustor burns similar waste feed(s) in terms of constituents, type of waste matrix, and amount of waste fed on a mass basis. In addition, we recommend evaluating the methods used to quantify COPCs and associated detection limits achieved during the test, as well as verifying that the data quality documentation is acceptable for risk assessment purposes.

SPECIAL NOTE: HOW THIS DOCUMENT RELATES TO TRIAL BURNS

We believe that generating defensible emission rates for compounds of potential concern (COPCs) is one of the most important parts of the risk assessment process. This requires special consideration when planning a risk assessment. Therefore, we consider emissions testing, risk assessment planning, and implementation as interdependent aspects of the hazardous waste combustion site-specific risk assessment process.

As described elsewhere in this chapter, **traditional regulatory performance tests (e.g., RCRA trial burns designed solely to measure DRE) *do not* sufficiently characterize COPC emissions for performing site-specific risk assessments**. We therefore generally recommend that collecting emissions data for a site-specific risk assessment include a thorough understanding of the operating limits to be established in the regulatory permit and the possible emissions that may result under the permitted operations of the unit. Regardless of whether the emissions data for the risk assessment is collected in a separate test condition or in multiple test conditions that are part of a regulatory performance test program, we recommend that to the extent possible, the planning, regulatory agency review, and collecting of emissions data be conducted simultaneously, to ensure consistency in data evaluation across test conditions is achieved and actual operations allowable under the permit are appropriately evaluated. This approach also eliminates redundancy or the need to repeat activities and minimizes cost expenditures overall.

The guidance documents below relate to the RCRA hazardous waste combustion program. You may find the listed documents useful for developing and conducting trial burns:

> U.S. EPA. 1989f. *Handbook: Guidance on Setting Permit Conditions and Reporting Trial Burn Results. Volume II of the Hazardous Waste Incineration Guidance Series.* Office of Research and Development (ORD). EPA/625/6-89/019. January.
>
> U.S. EPA. 1989g. *Handbook: Hazardous Waste Incineration Measurement Guidance Manual. Volume III of the Hazardous Waste Incineration Guidance Series.* Office of Solid Waste and Emergency Response. EPA/625/6-89/021. June.
>
> U.S. EPA 1990i. *Handbook: Quality Assurance/Quality Control (QA/QC) Procedures for Hazardous Waste Incineration*. Office of Research and Development. EPA/625/6-89/023. January.
>
> U.S. EPA. 1992c. *Technical Implementation Document for EPA's Boiler and Industrial Furnace Regulations.* Office of Solid Waste and Emergency Response. EPA-530-R-92-011. March.
>
> U.S. EPA. 2001c. *Risk Burn Guidance for Hazardous Waste Combustion Facilities*. U.S. EPA Region 4 and OSW. EPA 530/R/01/001. July.
>
> Generic Trial Burn Plans and/or Quality Assurance Plans and Procedures (QAPP) developed by individual EPA regional offices or authorized states.

2.2.1.1 Additional Emissions Testing Considerations

COPC emission rates demonstrated in a traditional regulatory performance test (such as a RCRA trial burn) are expected to be greater than normal emission rates because a facility "challenges" its combustor during a trial burn. These challenges introduce a wide range of conditions for automatic waste feed cutoff (AWFCO) systems. Regulatory performance tests are usually conducted under two conditions:

1. a high-temperature test, in which the emission rate of metals is maximized, and
2. a low-temperature test, in which the ability of the combustor to destroy principal organic hazardous constituents (POHCs) in the waste feed is challenged.

The combination of high POHC feed rates and extreme operating conditions tested during a low-temperature trial burn typically produce higher PIC emission rates. However, this is not true in all cases. For example, the formation of Polychlorinated dibenzo(p)dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) doesn't depend on "POHC incinerability" low temperature conditions. PCDDs and PCDFs can be formed catalytically in the low-temperature regions of the combustion unit or APCS. We recommend basing the decision to test under low, high, or both temperature conditions on the characteristics of the facility as discussed in the preceding section, considering facility-specific unit operation information for the particular types of wastes burned in the combustion unit as well as the particular APCS.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- All stack sampling information (current and historical data relevant to the risk assessment) on emission rates from the combustor during normal or day-to-day operating conditions and/or regulatory performance test conditions.

- A description of the waste feed streams burned during the stack sampling. This includes chemical composition and physical properties, which demonstrate that the waste feeds are representative of worst-case wastes relative to producing emissions that may pose a potential risk concern.

- Description of the operating conditions under which each set of emission rate data being used was developed.

*** * * NOTICE * * ***

The permitting authority might not request a risk assessment for every possible metal or PIC from a combustor. This doesn't imply, however, that it will only ask for targeted sampling for COPCs during regulatory performance tests. Based on permitting experience and discussions with analytical laboratories, we maintain that complete target analyte list analyses conducted when using U.S. EPA standard sampling methods (e.g., 0010 or 0030) don't subject facilities to significant additional costs or burdens during the trial burn process. We recommend that stack emission samplers strive to collect as much information as possible to characterize the stack gases generated from the combustion of hazardous waste. Facilities should, then, generally expect that data collected for the risk assessment may include the following tests: Method 0010, Method 0030 or 0031 (as appropriate), total organic compounds (using the Guidance for Total Organics, including Method 0040), Method 0023A, Method 26A or 26 (as appropriate), the multiple metals train, and method for particle size distribution (e.g., CARB 501 or Method 5 Modified with analysis by Scanning Electron Microscopy [SEM]). The permitting authority might also determine that using other test methods is appropriate for the performance test, to address detection limit or other site-specific issues. See Table B.1-4 of the *Risk Burn Guidance for Hazardous Waste Combustion Facilities* (U.S. EPA 2001c) for a complete list of methods.

2.2.1.2 Estimating the Total Organic Emission (TOE) Rate

We recognize that despite all efforts, it might not be possible to identify all the compounds in the emissions from a facility. This data gap has the potential to underestimate risks and represents a non-conservative uncertainty. Organic compounds that can't be identified by laboratory analysis can't be defensibly treated as COPC's in the risk calculations. However, these compounds might still contribute significantly to the overall risk, and so it's reasonable to consider them qualitatively in the risk assessment (DeCicco 1995; U.S. EPA 1994i).

U.S. EPA developed the total organic emissions (TOE) test as one approach to account for unidentified organic compounds because pre-existing methods, such as total hydrocarbon analyzers, don't fully determine the total mass of organics present in stack gas emissions (Johnson 1996). We anticipate that trial/risk burns will generally include the TOE test, in order to provide sufficient information to address concerns about the unknown fraction of organic emissions. The TOE test is used in conjunction with the identified organic compounds to calculate a TOE factor. We recommend using the TOE factor to qualitatively evaluate potential risks from the unidentified fraction of organic compounds in the stack gas.

The TOE test is the subject of other guidance, such as the *Guidance for Total Organics* (U.S. EPA 1996d) with additional clarification provided in Section B.7 of U.S. EPA (2001c). Proper use of TOE data depends on a good understanding of the test method and how the data is reported. The TOE method defines total organics as the sum of three fractions:

> ***Fraction 1:*** **Total Volatile Organic Compounds (TO_{VOC}) (referred to as Field GC Component in the TO Guidance) -** The fraction of organic compounds with boiling points less than 100°C. This VOC fraction is collected using U.S. EPA Method 0040. U.S. EPA Method 0040 allows for quantification of the total mass of organic compounds with boiling points less than 100°C, determined by summing the gas chromatograph/flame ionization detector results as described in the TO Guidance.
>
> ***Fraction 2:*** **Total Chromatographical Semivolatiles (TO_{SVOC}) (referred to as Total Chromatographical Organics Component in the TO Guidance) -** The fraction of organic compounds with boiling points from 100°C to and including 300°C. This VOC fraction is collected using modified U.S. EPA Method 0010 procedures as defined by U.S. EPA (1996d). The total mass of organic compounds with boiling points 100°C up to and including 300°C is determined by summing the total gas chromatograph/flame ionization detector results as described in the TO Guidance.
>
> ***Fraction 3:*** **Total Gravimetric Compounds (TO_{GRAV}) (referred to as Gravimetric component in the TO Guidance) -** The fraction of organic compounds with boiling points greater than 300°C. This fraction is determined by using modified U.S. EPA Method 0010 procedures defined by U.S. EPA (1996d), which quantify the mass, above this fractions boiling point, by measuring the total mass by evaporation and gravimetry (weighing) for nonvolatile total organics.

Please note that the TO total (TO_{TOTAL}) is the sum of the sums of each fraction. The sum of the TO fractions is described as follows:

$$TO_{TOTAL} = TO_{VOC} + TO_{SVOC} + TO_{GRAV} \qquad \text{Equation 2-1}$$

where:

TO_{TOTAL} = stack concentration of TO, including identified and unidentified compounds (mg/m^3)
TO_{VOC} = stack concentration of volatile TO, including identified and unidentified compounds (mg/m^3)
TO_{SVOC} = stack concentration of SVOC TO, including identified and unidentified compounds (mg/m^3)
TO_{GRAV} = stack concentration of GRAV TO, including identified and unidentified compounds (mg/m^3)

Use the TOE data in conjunction with the identified data to compute a TOE factor. Previously-computed TOE factors range from 2 to 40. The HHRAP defines the TOE factor as the ratio of the TO_{TOTAL} mass to the mass of identified organic compounds, as calculated by the following equation:

$$F_{TOE} = \frac{TO_{TOTAL}}{\Sigma_i C_i} \qquad \text{Equation 2-2}$$

where

F_{TOE} = TOE factor (unitless)
TO_{TOTAL} = total organic emission (mg/m^3)
C_i = stack concentration of the *i*th identified COPC (mg/m^3)

Identifying the organic compounds in the denominator of Equation 2-2 is one of the most critical components of the TOE factor. Although the permitting authority may not request that you analyze the organic compounds with all possible analytical methods, you may wish to consider the effects that gaps in compound-specific identification may have on the computation of the TOE factor. For example, hazardous waste-burning cement kilns have expressed concern about the amount of light hydrocarbons that may evolve from the raw materials processed in the cement kilns, because these light hydrocarbons have not typically been identified in trial burns. If such concerns are significant, you and the permitting authority might choose to use additional test methods in the trial burn in order to speciate the maximum number of organic compounds.

We also generally recommend including tentatively identified compounds (TICs) in the denominator when computing the TOE factor, so that appropriate credit is given to defensible efforts at identifying the maximum number of organic compounds. Finally, we generally recommend that non-detect COPCs be treated consistently between the risk assessment and TOE evaluation. That is, if a non-detected constituent is deleted as a COPC (See Section 2.3), then it would not be included in the identified fraction of the TOE equation. COPCs identified per Section 2.3, but not detected, might be included in the TOE factor equation at the reliable detection limit (non-isotope dilution methods) or the estimated detection limit (isotope dilution methods).

It's important to carefully evaluate the results of the gravimetric fraction when using the TOE factor. Both regulated industry and U.S. EPA have expressed some concern that the gravimetric fraction may over-report the organic fraction. It's been suggested that the gravimetric fraction may consist of organic and/or inorganic mass not directly attributable to organic incinerator emissions (U.S. EPA 1997a). The U.S. EPA Office of Research and Development (ORD) National Risk Management Research Laboratory (NRMRL) recently conducted a series of experiments to investigate this issue. The results indicate that it is indeed possible for inorganic mass to become soluble and retained in the TOE train methylene chloride extract. More importantly, the ORD/NRMRL research identified and demonstrated techniques for successfully mitigating this problem. Details of the experiments, results, and procedures for mitigating the GRAV bias will be made available in a forthcoming ORD report. Ultimately, these procedures will be contained in the forthcoming TOE guidance currently being revised by ORD. Further information on this topic is also available in U.S. EPA (2001c).

We recommend using the TOE factor in the uncertainty section of the risk assessment report to evaluate the risks from the unknown fraction of organics. The permitting authority can then evaluate the TOE factor and assess to what extent actual risks may be greater than estimated risks. For example, if the risk from the known portion of the emissions show that risks may be borderline and/or the TOE method shows that the unknowns are a significant portion of the emission profile, the permitting authority has several options, including:

1. Describe in a narrative form what is known of the unknown portion of the emissions.
2. As a bounding estimate, attribute a risk to the unknown portion of the emissions.

   An example is presented as a preferred option in U.S. EPA (1994f), which assumes that the unknown compounds are similar in toxicity and chemical properties to the known compounds taken as a whole. The referenced equation is as follows:

$$Q_{i,adj} = Q_i \cdot \frac{TO_{TOTAL}}{\Sigma_i C_i} \qquad \text{Equation 2-2A}$$

where

$Q_{i,adj}$ = adjusted emission rate of compound i (g/s)
$Q_{i,}$ = emission rate of compound i (g/s)
TO_{TOTAL} = total organic emission (mg/m^3)
C_i = stack concentration of the ith identified COPC (mg/m^3)

3. Recommend additional testing to identify a greater fraction of the organic compounds.
4. Specify permit conditions that further control total organic emissions or that further control the risks associated with known emissions.

Variations of the TOE factor can be useful to address site-specific concerns. For example, compute three separate TOE factors based on the apportioning provided by the TOE test (i.e., TO_{VOC}, TO_{SVOC}, and TO_{GRAV}). Then evaluate the unknowns associated with each fraction of unidentified organic compounds separately.

2.2.2 Estimating Emission Rates for Facilities with Multiple Stacks

We generally recommend that the risk assessment consider emissions from all combustors burning hazardous waste at a facility, not just the unit currently undergoing the permitting process. As discussed further in Chapter 3, air dispersion modeling for each combustor (source) is frequently conducted separately, to evaluate risk on a stack- or source-specific basis. An example case is a chemical manufacturing facility which operates both an on-site incinerator and several hazardous waste-burning boilers. Whether it is the incinerator or the boilers being permitted, the risk assessment considers the emissions from all the combustors in the estimate of facility risk. In addition to RCRA combustors, emissions from other RCRA treatment, storage, or disposal units (e.g., open burning/open detonation and thermal desorption) might also be included in the risk evaluation in some cases.

2.2.3 Estimating Stack Emission Rates for Facilities Not Yet Operational

The permitting process for new hazardous waste combustion facilities includes submitting information of sufficient detail for the regulatory authority to evaluate compliance with existing regulations, guidance, and standards of protectiveness. Stack (or other source) locations and dimensions, design flow and emission rate estimates, waste feed characteristics, surrounding building dimension data, facility plot plans, and terrain data are frequently reviewed and used in a pre-operation risk assessment. This assists decision-making and designing permit requirements.

We generally recommend reviewing design emission rates, waste feed characteristics, and other design data, along with supplementary documentation, to make sure they are representative, accurate, and comprehensive. Good engineering practice dictates a check of, and comparison with, data from similar existing units. Stack test reports for facilities of similar technology, design, operation, capacity, auxiliary fuels, waste feed types, and APCSs can be useful in estimating COPC emission rates for new facilities that have not been constructed. In addition to design data, particle size distribution data from a similar type unit that is operational may be useful. Estimated emission rates used to complete pretrial burn risk assessments are frequently compared to the measured emission rates from actual performance tests completed after the new facility receives a permit, is constructed and operational.

If surrogate data from similar facilities aren’t available, some state environmental agencies enforce emission rate limits based on state laws. Since these limits cannot be exceeded, you could use them to develop emission rate estimates for the risk assessment. A trial or risk burn could then demonstrate that facility emissions are less than those considered in the permit and risk assessment.

2.2.4 Estimating Stack Emission Rates for Facilities Previously Operated

We generally recommend that the risk assessment also consider emissions from the historical operation of other combustors burning hazardous waste at the facility, not just the unit currently undergoing the permitting process. The permitting authority will determine the appropriateness of this on a case-by-case basis. An example case might be when the emissions from historical operation of a source or sources have already resulted in potential risk concerns at or near the facility. You could model emissions from historical operations as a separate source or, if applicable, include them in the fate and transport equations by adding the previous years of operation to the anticipated time period of combustion for an existing or newly operating source. In some cases, you might also include historical emissions from other RCRA treatment, storage, or disposal units at the facility (e.g., open burning/open detonation and thermal desorption) in the risk assessment, in addition to RCRA combustors.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- All stack test reports for combustors used to develop emission rate estimates
- If using surrogate data to assess a new facility, descriptions of how the combustion data used represent similar technology, design, operation, capacity, auxiliary fuels, waste feed types, APCSs, and particle size distributions
- Demonstration that the data used to develop the emission rate estimates were collected using appropriate U.S. EPA sampling and analysis procedures
- The range of data obtained, and values used, in completing the risk assessment

2.2.5 Emissions From Process Upsets

It is possible for unburned hazardous waste to be emitted through the stack as a result of various process upsets, such as start-ups, shutdowns, and malfunctions of the combustion unit or APCS. Emissions can also be caused by operating upsets in other areas of the facility (e.g., an upset in a reactor which vents gases to a boiler burning hazardous waste could trigger a process upset in the boiler, resulting in increased emissions). U.S. EPA (1994i) indicates that upsets aren’t generally expected to significantly increase stack emissions over the lifetime of the facility.

To account for the increased emissions associated with process upsets, we generally recommend that the stack emission rates estimated from trial burn data be multiplied by an upset factor. The upset factor is not applied to non-PIC emission rate estimates where the total mass of a constituent in the waste feed is assumed to be emitted. When available, site-specific emissions or process data can be useful to estimate the upset factor. You may also want to consider and evaluate the following types of data to derive the upset factor:

- Data from continuous emissions monitoring systems that measure stack carbon monoxide, oxygen, total hydrocarbon (if requested), or opacity (if appropriate)
- Data on combustion chamber, APCS, or stack gas temperature
- Data on hazardous waste residence time
- Frequency and causes of automatic waste feed cutoffs (AWFCO)
- Frequency of start-up and shut-down events
- Ratio of AWFCO frequency and duration to operating time

- APCS operating variables like baghouse pressure drop, liquid scrubber flow rate, or electrostatic precipitator voltage

- Stack test data collected while the combustor was operated under upset conditions

You might use this information to estimate the magnitude of the increase in emissions and the percentage of time, on an annual basis, that the unit operates at upset conditions. Additional information regarding upset factors for liquid-burning BIFs is available in the Louisiana Chemical Association (LCA) Letter Report on Upset Factors, dated October 27, 1999, available on the U.S. EPA Region 6 web site (www.epa.gov/region06/).

If you don't have site-specific data, or they are inappropriate for deriving an upset factor, we generally recommend estimating upset emission rates using a procedure based on work by the California Air Resources Board (CARB) (1990).

> ***Estimating Emissions from Process Upsets:*** To represent stack emission rates during process upsets, multiply the emission rate developed from the trial burn data by 2.8 for organics and 1.45 for metals. These factors are derived by assuming that emissions during process upsets are 10 times greater than emissions measured during the trial burn. Since the unit doesn't operate under upset conditions continually, the factor is adjusted to account for only the period of time, on an annual basis, that the unit operates under upset conditions. For organic compounds, the facility is assumed to operate as measured during the trial burn 80 percent of the year and operate under upset conditions 20 percent of the year [(0.80)(1)+(0.20)(10)=2.8]. For metals, the combustor is assumed to operate as measured during the trial burn 95 percent of the year and operate under upset conditions the remaining 5 percent of the year [(0.95)(1)+(0.05)(10)=1.45].

Catastrophic process upsets brought about by complete failure of combustion and air pollution control systems (e.g. resulting from non-routine events such as explosions, fires, and power failures) are typically considered accidental releases and consequently aren't addressed by this guidance.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Historical operating data demonstrating the frequency and duration of process upsets

- A discussion of the potential cause(s) of the process upsets

- Estimates of upset magnitude or emissions

- Calculations which describe the derivation of the upset factor.

2.2.6 RCRA Fugitive Emissions

RCRA fugitive emission sources frequently evaluated in site-specific risk assessments include waste storage tanks; process equipment ancillary to the combustor; and the handling and disposal of combustion system residues such as ash. Fugitive emissions from other RCRA treatment, storage, or disposal units (e.g., landfills) may also warrant evaluation in some cases.

This section contains guidance for quantitatively estimating fugitive emissions using procedures outlined in other U.S. EPA guidance. Guidance regarding air dispersion modeling of fugitive emissions is presented in Chapter 3.

2.2.6.1 Quantitative Estimation of RCRA Fugitive Emissions from Process Equipment

We generally recommend the following series of steps to quantitatively estimate RCRA fugitive emissions: (1) identify equipment to evaluate as a fugitive emission source(s); (2) group equipment, as appropriate, into a combined source; and (3) estimate compound-specific emission rates for each resulting source. We illustrate an example in Figures 2-1 and 2-2 and Tables 2-1 and 2-2, to help explain the recommended steps. Figure 2-1 presents the plot plan of a hypothetical facility that includes one RCRA combustion unit (CU-1), two hazardous waste feed storage tanks (WST-1 and WST-2), and ancillary equipment identified in a RCRA Part B permit application.

Step 1: Identify Fugitive Emission Sources **-** Generally, identify RCRA fugitive emission sources such as waste storage tanks and process equipment that comes in contact with a RCRA hazardous waste. Such equipment is specified in Title 40, Code of Federal Regulations (40 CFR) Part 265, Subpart BB. Equipment covered under Subpart BB includes:

- Pumps
- Valves
- Connectors (flanges, unions, tees, etc.)
- Compressors
- Pressure-relief devices
- Open-ended lines
- Product accumulator vessels
- Sampling connecting systems
- Closed vent systems

- Agitators

Note each fugitive emission source on a facility plot map with a descriptor and the location denoted with Universal Transverse Mercator (UTM) coordinates (specify if North American Datum [NAD] of 27 or NAD83).

Step 2: Group Equipment Into a Combined Source **-** To significantly reduce the effort required to complete air dispersion modeling and the subsequent risk assessment, group equipment in close proximity, and evaluate as a single combined source. The speciated emission rates for the group are the summation of the emissions from the grouped individuals. Clearly denote on a facility plot plan or map the area extent of the grouped or combined source, as defined by UTM coordinates (specify if NAD27 or NAD83). Define the area extent of the combined source using the actual locations of the equipment being grouped, without exaggeration to cover areas without fugitive sources. It may also be useful to consider how fugitive emission sources are to be defined when conducting the air dispersion modeling (see Chapter 3).

Equipment in two areas of the hypothetical facility shown in Figure 2-1 are grouped into combined sources; these consist of the Waste Feed Storage Area and the RCRA Combustor Area.

Step 3: Estimate Fugitive Emissions from Tanks **-** Obtain fugitive emission rates for waste storage tanks from the facility's emission inventory or Title V air permit application prepared in compliance with Clean Air Act Amendments of 1990 (see example provided as Figure 2-2). If that information is not available, fugitive emissions from storage tanks can be calculated using U.S. EPA's TANKS Program or by following the procedures outlined in U.S. EPA (1995a), *"Compilation of Air Pollution Emission Factors, January 1995."*

The information needed to accurately estimate fugitive emission rates from storage tanks includes, but is not limited to:

- Dimensions of the tanks
    - Shell height and diameter

- Characteristics of the tank roof
    - Color and shade
    - Condition (e.g., poor, good)
    - Type (e.g., cone, dome)
    - Height
    - Radius or slope
    - Fixed or floating

- Characteristics of the shell
    - Color and shade
    - Condition (e.g., poor, good)
    - Heated

- Settings on breathe vents
    - Vacuum setting

  - Pressure setting

- Characteristics of the stored liquids
  - Maximum and annual average liquid height
  - Working volume
  - Turnovers per year
  - Net throughput
  - Average annual temperature
  - Vapor pressures of speciated constituents (at annual average temperature)

Step 4: Estimate Fugitive Emissions from Process Equipment - Estimate fugitive emissions for each type of equipment listed under 40 CFR Part 265, Subpart BB by using the following four approaches, listed in order of increasing refinement and data requirements:

- Average Emission Factor Approach (AEFA)
- Screening Ranges Approach (SRA)
- U.S. EPA Correlation Approach (EPACA)
- Unit-Specific Correlation Approach (USCA)

These four approaches would generally be applicable to estimate fugitive emission rates of volatile organic compounds (VOCs) from equipment on any facility. Except for the AEFA method, all of the approaches need screening data collected using a portable monitoring device (PMD). Because data on fugitive emissions at a facility is typically limited, the AEFA method is expected to be used in most cases, and therefore has been selected for use in the example illustrated in Figure 2-1, and Tables 2-1 and 2-2. However, we recommend using more refined approaches such as SRA, EPACA, or USCA, if sufficient data is available. U.S. EPA (1995k) provides a detailed discussion on these three approaches. Additional information on estimating fugitive emission rates is available in U.S. EPA (1995k), *“Protocol for Equipment Leak Emission Estimates, EPA-453/R-95-017."*

FIGURE 2-1
EXAMPLE FACILITY PLOT MAP

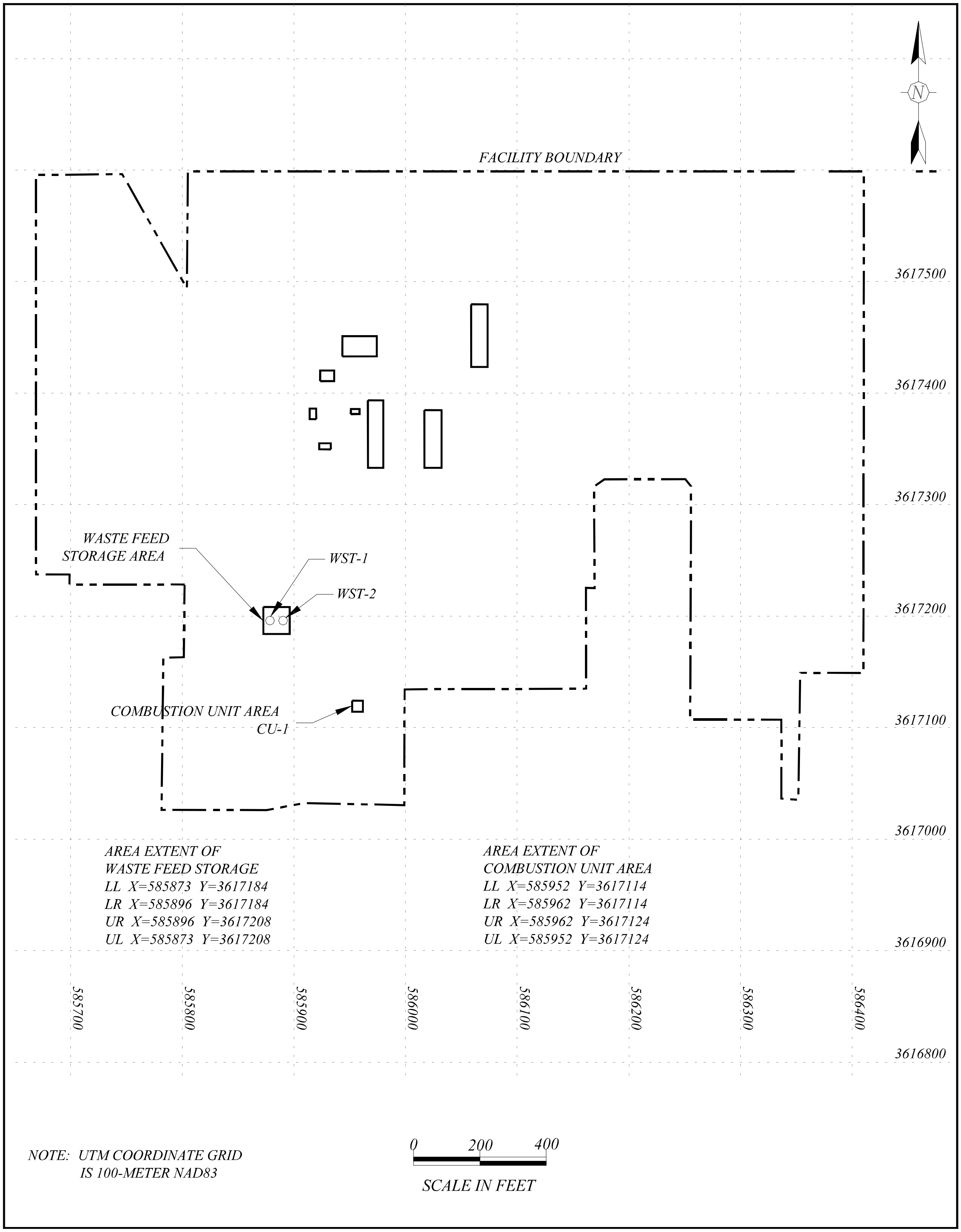
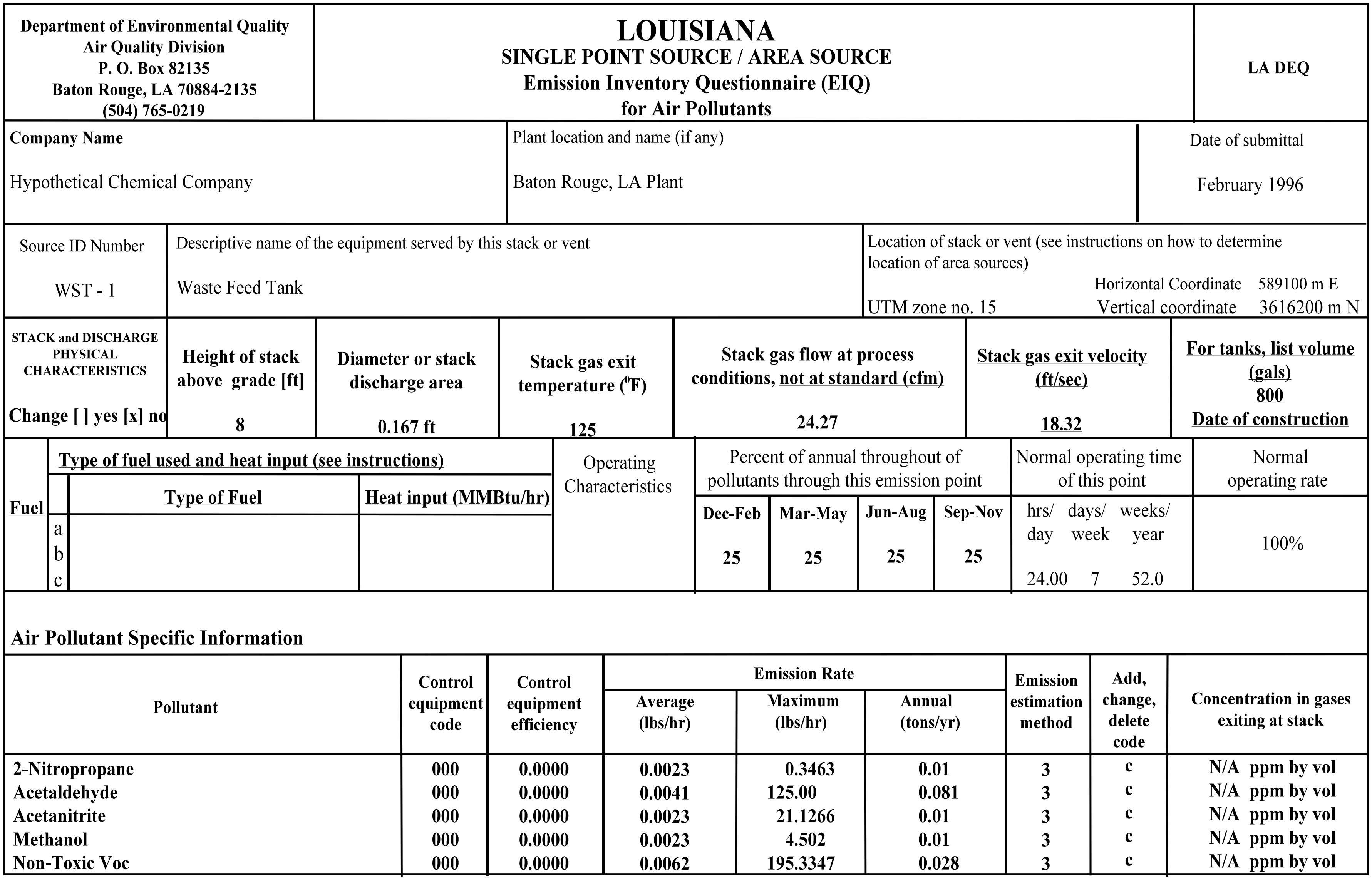


FIGURE 2-2
EXAMPLE EMISSIONS INVENTORY



Department of Environmental Quality **Air Quality Division** **P. O. Box 82135** **Baton Rouge, LA 70884-2135** **(504) 765-0219**	**LOUISIANA** **SINGLE POINT SOURCE / AREA SOURCE** **Emission Inventory Questionnaire (EIQ)** **for Air Pollutants**	**LA DEQ**

Company Name Hypothetical Chemical Company	Plant location and name (if any) Baton Rouge, LA Plant	Date of submittal February 1996

Source ID Number WST - 1	Descriptive name of the equipment served by this stack or vent Waste Feed Tank	Location of stack or vent (see instructions on how to determine location of area sources) Horizontal Coordinate 589100 m E UTM zone no. 15 Vertical coordinate 3616200 m N

STACK and DISCHARGE PHYSICAL CHARACTERISTICS **Change [ ] yes [x] no**	**Height of stack above grade [ft]** **8**	**Diameter or stack discharge area** **0.167 ft**	**Stack gas exit temperature (^{0}F)** **125**	**Stack gas flow at process conditions, not at standard (cfm)** **24.27**	**Stack gas exit velocity (ft/sec)** **18.32**	**For tanks, list volume (gals)** **800** **Date of construction**

Fuel	**Type of fuel used and heat input (see instructions)**			Operating Characteristics	Percent of annual throughout of pollutants through this emission point				Normal operating time of this point			Normal operating rate
		Type of Fuel	**Heat input (MMBtu/hr)**		**Dec-Feb**	**Mar-May**	**Jun-Aug**	**Sep-Nov**	hrs/ day	days/ week	weeks/ year	
	a				25	25	25	25	24.00	7	52.0	100%
	b											
	c											

Air Pollutant Specific Information

Pollutant	**Control equipment code**	**Control equipment efficiency**	**Emission Rate**			**Emission estimation method**	**Add, change, delete code**	**Concentration in gases exiting at stack**
			Average (lbs/hr)	**Maximum (lbs/hr)**	**Annual (tons/yr)**			
2-Nitropropane	**000**	**0.0000**	**0.0023**	**0.3463**	**0.01**	**3**	**c**	**N/A ppm by vol**
Acetaldehyde	**000**	**0.0000**	**0.0041**	**125.00**	**0.081**	**3**	**c**	**N/A ppm by vol**
Acetanitrite	**000**	**0.0000**	**0.0023**	**21.1266**	**0.01**	**3**	**c**	**N/A ppm by vol**
Methanol	**000**	**0.0000**	**0.0023**	**4.502**	**0.01**	**3**	**c**	**N/A ppm by vol**
Non-Toxic Voc	**000**	**0.0000**	**0.0062**	**195.3347**	**0.028**	**3**	**c**	**N/A ppm by vol**

TABLE 2-1

EXAMPLE CALCULATION
TOTAL FUGITIVE EMISSION RATES FOR EQUIPMENT IN WASTE FEED STORAGE AREA

1	2	3	4	5	6		7	8	9	10
Fugitive Emission Source	**Waste Stream**	**Type of Waste Stream In Service**	**Equipment Type**	**Number of Each Equipment Type Per Waste Stream**	**Equipment Emission Factors**		**Total VOC Weight Fraction**	**Operational Time Period of Equipment (days)**	**Total VOC Emissions Rate by Equipment (g/sec)**	**Total Fugitive Emission Rate (g/sec)**
					(kg/hr)	**(g/sec)**				
Waste Feed Storage Area	Process A Wastes	Light Liquid	Pumps	3	0.01990	0.00553	0.9	180	0.01493	0.14926
		Light Liquid	Valves	70	0.00403	0.00112	0.9	180	0.07056	
		Light Liquid	Connectors	30	0.00183	0.00051	0.9	180	0.01377	
		Light Liquid	Tank WST-1	1	--	--	0.9	180	0.02	
		Light Liquid	Tank WST-2	1	--	--	0.9	180	0.03	
	Process B Wastes	Heavy Liquid	Pumps	2	0.00862	0.00239	0.6	180	0.00287	0.06857
		Heavy Liquid	Valves	75	0.00023	0.00112	0.6	180	0.0504	
		Heavy Liquid	Connector	50	0.00183	0.00051	0.6	180	0.0153	
		Heavy Liquid	Tank WST-1	1	--	--	0.6	0	0	
		Heavy Liquid	Tank WST-2	1	--	--	0.6	0	0	

Notes:

Column 1	Equipment in the Waste Feed Storage Area was identified and grouped as a combined RCRA fugitive emission source with an area extent defined by UTM coordinates (NAD83).
Column 2	The waste streams serviced by equipment in the Waste Feed Storage Area can be determined through review of the facility's RCRA Part B Permit Application, Air Emission Standards.
Column 3	The type of waste stream in service, defined as light or heavy for determination of equipment-specific emission factors, can be determined from review of waste stream vapor pressure.
Column 4	Similar types of equipment can be grouped according to the most applicable equipment-specific emission factor and type of waste stream service (light or heavy) provided in U.S. EPA (1995k).

Column 5	The number of equipment type (per waste stream) identified in column 3.
Column 6	Emission factors specific to each type of equipment can be obtained from U.S. EPA (1995k), with the exception of storage tanks.
Column 7	Weight fraction of total volatile organic compounds was obtained from dividing the concentration of VOCs (mg/L) by the density of the waste stream (mg/L).
Column 8	Assumed the equipment is operational for 180 days a year.
Column 9	Equipment-specific fugitive emission rates were determined by multiplying Columns 5, 6, and 7. Emission rates for tanks were obtained from Title V air permit application. In the absence of such data, emission rates for tanks can be calculated using U.S. EPA's TANKS Program or by following the procedures outlined in U.S. EPA (1995a).
Column 10	The total fugitive emission rate for each waste stream is determined by summing emission rates for all the equipment. Table 2-2 presents calculations for estimating speciated fugitive emissions.

TABLE 2-2

EXAMPLE CALCULATION SPECIATED FUGITIVE EMISSIONS FOR EQUIPMENT IN WASTE FEED STORAGE AREA

1	2	3	4	5	6
Fugitive Emission Source	**Waste Stream**	**Waste Stream Composition**	**Weight Fraction of Each VOC In Waste Stream (%)**	**Total Fugitive Emission Rate (g/sec)**	**Speciated Fugitive Emissions (g/sec)**
Waste Feed Storage Area	Process A Wastes	Acetaldehyde	0.20	0.14926	0.0030
		Acetonitrile	0.25		0.0037
		2-Nitropropane	0.25		0.0037
		Nitromethane	0.20		0.0030
	Process B Wastes	Acetaldehyde	0.20	0.06857	0.0137
		Acetonitrile	0.10		0.0069
		Methanol	0.20		0.0137
		Propionitrile	0.05		0.0034

Notes:

Column 1	Equipment in the Waste Feed Storage Area was identified and grouped as a combined RCRA fugitive emission source with an aerial extent defined by UTM coordinates (NAD83).
Column 2	The waste streams serviced by equipment in the Waste Feed Storage Area can be determined through review of the facility’s RCRA Part B Permit Application, Air Emission Standards.
Column 3	The waste stream composition can be determined from analytical data
Column 4	Weight fraction of compounds in the waste stream can be determined from analytical data or review of the facility’s Title V Air Permit Application, Emissions Inventory Questionnaire (EIQ) for Air Pollutants (see example in Figure 2-2).
Column 5	The total fugitive emission rate for each waste stream was obtained from Column 10, Table 2-1.
Column 6	Speciated fugitive emissions were obtained by multiplying Column 4 and 5.

An Example Calculation Using the AEFA Method

Information needed to estimate fugitive emission rates using the AEFA method includes:

- Type of waste stream associated with each equipment type (Columns 2 and 3, Table 2-1)
  - light liquids are those in which the sum of the concentration of individual constituents with a vapor pressure over 0.3 kilopascals (kPa) at 20°C is greater than or equal to 20 weight percent
  - heavy liquids are all others liquids not meeting the definition of light liquids as specified above

- Number of each equipment type associated with each waste stream (Columns 4 and 5, Table 2-1)
- Total VOC weight fraction of each waste stream (Column 7, Table 2-1)
- Weight fraction of each VOC in each waste stream (Columns 3 and 4, Table 2-2)
- Operational time period of equipment (Column 8, Table 2-1)

In the AEFA method, equipment is grouped by waste streams of similar characteristics and VOC composition (Columns 1 and 2, Table 2-1). However, the AEFA method doesn't account for different site-specific conditions such as temperature, vapor pressure, or screening values, among process units within a source category. Site-specific factors can significantly influence fugitive emission rates of leaks from equipment.

U.S. EPA (1995k) presents the average emission factors (Column 6, Table 2-1) for synthetic organic chemicals manufacturing industry process units, refineries, and natural gas plants. The following table is an excerpt from this guidance document. These emission factors are most valid for estimating rates of emissions from a grouping of equipment over a long time period.

TABLE 2-3

SOCMI AVERAGE EMISSION FACTORS

Equipment type	Service	Emission factor (kg/hr/source)
Valves	Gas	0.00597
	Light liquid	0.00403
	Heavy liquid	0.00023
Pump seals	Light liquid	0.0199
	Heavy liquid	0.00862
Compressor seals	Gas	0.228
Pressure relief valves	Gas	0.104
Connectors	All	0.00183
Open-ended lines	All	0.0017
Sampling connectors	All	0.0150

Source: U.S. EPA (1995k)

To calculate the total VOC emissions rate for a specified equipment type, multiply the equipment emission factor by the total VOC weight fraction and the number of each equipment type per waste stream (Column 9, Table 2-1 = Column 6 x Column 7 x Column 5).

Generate the total fugitive emission rate for the waste stream (Column 10, Table 2-1) by summing the total VOC emission rates for each equipment type. Speciated fugitive emissions are then calculated by multiplying the weight fraction of each VOC in the waste stream and the total fugitive emission rate for the waste stream (Column 6, Table 2-2 = Column 4 x Column 5). This speciated emission rate is the emission rate used in the risk assessment.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Summary of the step-by-step process conducted to evaluate fugitive emissions
- Facility plot map clearly identifying each fugitive emission source with a descriptor and the location denoted with UTM coordinates (specify if NAD27 or NAD83).
- Speciated emission rate estimates for each waste stream serviced by each source, with supporting documentation
- Applicable discussion of monitoring and control measures used to mitigate fugitive emissions

2.2.6.2 Fugitive Emissions from Combustor Leaks

We recommend that when appropriate, the risk assessment evaluate fugitive emissions resulting from the construction, design, or operation of a hazardous waste combustor. Examples of fugitive emissions from combustor leaks include:

- Combustors operating under negative pressure may experience temporary positive pressures ("puffing") that cause fugitive emissions. This condition can occur when a slug of high BTU waste is combusted, causing a rapid expansion in the volume of combustion gases that exceeds the volume of the combustion chamber.
- Fugitive emissions resulting from the day-to-day operation of the combustor and APCS. These emissions will typically include (1) leaks that occur due to a positive pressure in the APCS, and (2) routine maintenance activities such as replacement of baghouse collection bags.

Currently, we don't offer any specific guidance on how to quantitatively estimate fugitive emissions from hazardous waste combustors. However, if no site-specific quantitative methods are available, one option is to address risks associated with leaks in the uncertainty section of the risk assessment. Under such an approach, the permitting authority could review facility-specific data to determine whether or not the design addresses equipment leaks and whether the operational data indicate that equipment leaks may be a problem.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Process design information and drawings (if necessary)
- Past operating data indicating the frequency, duration, and magnitude of combustor leaks
- Information regarding the probable cause of combustor leaks
- Summary of procedures in place to monitor or minimize fugitive emissions resulting from combustor leaks

2.2.7 RCRA Fugitive Ash Emissions

Burning hazardous waste may produce flyash. Fugitive particle emissions may result from the associated collection, handling, and disposal of the flyash. Typically, fugitive emissions of flyash collected from an air pollution control device (APCD) will occur during transfer into covered trucks or other conveyance mechanisms prior to disposal. Emissions generated during the loading process can be controlled by APCDs or other types of equipment. However, some of the flyash may still escape into the atmosphere as fugitive emissions.

We generally recommend the following steps to quantitatively estimate RCRA fugitive ash emissions: (1) determine an empirical emission factor, (2) estimate the flyash generation rate, and (3) account for air pollution control equipment, if applicable. As demonstrated in the example calculation below, it is then possible to estimate the fugitive ash emission rate by multiplying the empirical emission factor by the flyash generation rate and, if applicable, the control deficiency of the air pollution control equipment.

Step 1: Determine an Empirical Emission Factor **-** One approach to estimate particle emissions associated with flyash loading and unloading is to use an empirical emission factor of 1.07 lb per ton flyash. This factor is based on a field testing program conducted at a coal fired power plant equipped with an electrostatic precipitator (ESP) (Muleski and Pendleton 1986). Because burning coal and hazardous wastes are similar activities, flyash generated from similar control devices is expected to behave similarly under the same conditions, with respect to fugitive emissions. In general, particle behavior is dependent more on the physical form of the flyash than on the feed (or waste) stream being burned. The emission factor determined during the empirical study (0.107 lb per ton flyash) can be adjusted by a factor (e.g., 10) to account for the fact that the flyash from burning coal (in the study) was wetted. Depending on the facility, the flyash from the hazardous waste combustion facility may or may not be wetted.

Step 2: Estimate the Flyash Generation Rate **-** Obtain the APCD flyash generation rate from the Part B Permit Application. Obtain the total ash content of the “generic” waste streams created from the waste profile. Both values should be approximately the same. Since a major portion of ash fed to the combustor is converted to bottom ash, it is likely that this value can be assumed to be a high estimate of the actual flyash generation rate.

Step 3: Account for Air Pollution Control Equipment **-** If an APCD is used for controlling emissions during flyash handling operations, you can generally apply an efficiency factor (e.g., 99.5 percent) to the emission rate. An efficiency factor of 99.5 percent is based on typical collection efficiencies of particulate matter control devices, for the particle sizes in the range of 2.5 to 10 um (U.S. EPA 1995a).

Example Calculation

Multiply the empirical emission factor (Step 1) times the estimated flyash generation rate (Step 2):

[(1.07 lb per ton) * (5,000 tons per year) = 5,350 lbs per year].

To account for the air pollution control equipment, multiply the product of Steps 1 and 2 times one minus the fabric filter efficiency (Step 3) to calculate the final RCRA fugitive ash emission rate for use in the risk assessment:

[(5,350 lbs per year) * (1 - 0.995) = 26.75 lbs per year].

2.2.8 Cement Kiln Dust (CKD) Fugitive Emissions

CKD is the particulate matter (PM) that is removed from combustion gas leaving a cement kiln. This PM is typically collected by an APCS—such as a cyclone, baghouse, ESP—or a combination of APCSs. Many facilities recycle a part of the CKD back into the kiln. Current and applicable guidance on evaluating CKD includes (1) the *Technical Background Document for the Report to Congress* (U.S. EPA 1993g), and (2) the regulatory determination of CKD (60 FR 7366, February 7, 1995).

Most CKD constituents (for example, metals) aren't volatile but could be released to air through fugitive dust emissions as volatile or semivolatile organics. These emissions can be in gaseous form and present in relatively low concentrations, if at all (U.S. EPA 1993a). Dust particles may be suspended in the air by either wind erosion or mechanical disturbances. The extent to which dust is blown into the air by wind erosion depends on several site-specific characteristics, including (1) the texture (particle size distribution) and moisture content of the CKD on the surface of piles, (2) non-erodible elements, such as clumps of grass or stones on the pile, (3) presence of a surface crust, and (4) wind speeds. Mechanical disturbances that can suspend CKD constituents in the air include (1) vehicular traffic on and around CKD piles, (2) CKD dumping and loading operations, and (3) transportation of CKD around a plant site in uncovered trucks. Cement plants may use various control measures to limit the release of CKD to the air. For example, CKD may be pelletized in a pug mill, compacted, wetted, and covered to make the material less susceptible to wind erosion.

To keep the dust down, many facilities add water to CKD before disposal, to agglomerate individual particles. In addition, as CKD sits in a pile exposed to the elements, occasional wetting by rainfall may form a thin surface crust in inactive areas of the pile. This acts to mitigate air entrainment of particles. However, based on field observations by U.S. EPA (1993g), neither surface wetting nor natural surface crusting eliminates the potential for CKD to be blown into the air. Wetting the dust before disposal provides incomplete and temporary control because water is infrequently applied, and the dust ultimately dries and returns to a fine particulate that is available for suspension and transport. Similarly, a surface crust may develop, but (1) the crust breaks when vehicles or people move on the pile, and (2) fresh dust is regularly added to the pile, providing a continual, exposed reservoir of fine particles. Please note that a crust doesn't always form, for a variety of reasons such as weather and CKD chemistry.

CKD constituents that are released to the air are transported and dispersed by the winds, and are ultimately deposited onto land or water, either by settling in a dry form or by being entrained in precipitation.

2.2.8.1 Composition and Characteristics of CKD

We evaluated the potential direct and indirect risks resulting from on-site and off-site management of CKD (U.S. EPA 1993g; 1993h). These studies highlight the limited amount of available information regarding variation in the chemical constituents of CKD generated by facilities burning hazardous waste as fuel, and by facilities burning only fossil or nonhazardous waste fuels. There may also be differences in composition between the “as-generated” CKD -- a portion of which is recycled back into the system – and the “as-managed” CKD that is disposed of on or offsite.

The air exposure pathway is generally of concern for CKD, because the dust is a fine PM that is readily suspendable, transportable, and respirable in air. In general, particles that are #100 micrometers can be suspended in the wind and transported. Within this range, particles that are #30 micrometers can be transported for considerable distances downwind. However, particles that are #10 micrometers are of primary concern for respiration by humans (U.S. EPA, 1993g). Virtually all of the dust generated at the 15 facilities evaluated by U.S. EPA (1993g) in the *Cement Kiln Dust Report to Congress* may be suspended and transported in the wind (that is, the vast majority of particles are #100 micrometers), and over two-thirds of all CKD particles generated may be transported over long distances. Additionally, a significant percentage of the total dust generated (from 22 to 95 percent, depending on kiln type) comprises respirable particles that are #10 micrometers.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

C Physical data, including particle size distribution and density

C Chemical data, including organic and inorganic analytical tests similar to those used for sampling combustion gases

C Plant net CKD generation rate (how much CKD per year that is available for disposal)

C Ambient air monitoring data

C CKD management, transportation, storage, and disposal methods

C Containment procedures, including fugitive dust prevention measures and the area of exposed CKD

C Meteorological data, including wind speed and precipitation

2.2.8.2 Estimating CKD Fugitive Emissions

In general, the HHRAP doesn‘t address quantitative estimation of risk from fugitive CKD emissions. However, risk assessments of cement manufacturing facilities are still able to evaluate the fugitive CKD emissions qualitatively. The *Technical Background Document for the Report to Congress* (U.S. EPA

1993g), includes methods to estimate the magnitude of fugitive emissions from the handling, storage, and disposal of CKD. Sampling data of CKD collected during maximum waste metal feed rate conditions from the trial burn, risk burn and/or certification of compliance tests may also be useful in evaluating CKD fugitive emissions. You can then evaluate it qualitatively by comparing the risks estimated for the kiln stack emissions, to the high end national screening level estimated by U.S. EPA for CKD in U.S. EPA (1993g) and the regulatory determination of CKD (60 FR 7366, February 7, 1995). If the risks are equivalent, the combined risks appear significant, or the risks attributed to the CKD are greater than the risks estimated for the kiln stack emissions, it might be appropriate to evaluate the risk from CKD emissions in a more quantitative fashion. We generally recommend that the permitting authority make sure that any qualitative evaluation includes a comparison of the conditions at the facility to the conditions at the model facilities we evaluated in U.S. EPA (1993g; 1993h). In addition, an analysis of a specific facility's compliance with other risk-based environmental statutes and regulations is often an appropriate method to qualitatively evaluate risks associated with the handling, storage, and disposal of CKD.

2.3 IDENTIFYING COMPOUNDS OF POTENTIAL CONCERN

Compounds of potential concern (COPCs) are those compounds evaluated throughout the risk assessment. There is no universal list of COPCs, because a compound that's a COPC for one combustor may not be a COPC for another combustor. COPCs in the emissions from hazardous waste combustors vary widely, depending on the type of

- combustor,
- fuel and hazardous waste feed being burned, and
- APCS used.

COPCs include metals, products of incomplete combustion (PICs), and/or reformation products. PICs are any organic compounds emitted from a source that are present in the feed stream (even in trace amounts) and not completely destroyed in the combustion process. Reformation products are organic compounds that are formed immediately after combustion, due to interaction of specific constituents in the combustion gasses and specific unit operating conditions relative to a particular combustion process and associated air pollution control equipment.

PICs can be formed by trace toxic organic compounds in the waste feed stream. Therefore, we generally recommend evaluating these trace compounds as PIC precursors, in addition to those compounds more prevalent in the hazardous waste feed. ***Don't confuse PICs with principal organic hazardous constituents (POHCs).*** POHCs are compounds in the waste feed stream used during a performance test burn to measure combustor DRE. Unburned POHCs and partially destroyed or reacted POHCs are PICs, but PICs are not necessarily POHCs. We've typically subdivided COPCs into seven different constituent categories (U.S. EPA 1994g; 1994i; 1994j; 1994n):

C Polychlorinated dibenzo(p)dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)
C Polynuclear aromatic hydrocarbons (PAHs)
C Polychlorinated biphenyls (PCBs)
C Nitroaromatics
C Phthalates
C Other organics
C Metals

Table A-1 (Appendix A) presents a comprehensive list of compounds typically found (1) in hazardous waste, and (2) in hazardous waste combustion stack gas emissions. Table A-1 identifies the Chemical Abstracts Service (CAS) number for each compound, and states whether the compound has been identified as a carcinogen. Table A-1 also indicates whether a compound has been identified as a potential COPC by

- U.S. EPA and state risk assessment reference documents,
- emission test results that have identified the compound in the emissions from hazardous waste combustion facilities, or
- other literature that suggests that the risks from the compound may be significant.

We provide Table A-1 to help you make sure that the performance test program considers the full range of compounds potentially emitted from a combustor, and the appropriate analytical method. A risk assessment won't necessarily evaluate every metal, potential PIC, and reformation product listed in Table A-1. Once the performance tests are completed, we recommend selecting the risk assessment COPCs from the stack test data and available facility-specific process information, rather than Table A-1.

Identify COPCs from the trial/risk burn data based on their potential to pose increased risk or hazard via one or more of the direct or indirect exposure pathways. We recommend focusing on compounds that

- are likely to be emitted because they (or their precursors) are present in the waste feed,

- are likely to be emitted because they are likely reformation products,
- are potentially toxic to humans, and/or
- have a tendency to bioaccumulate or bioconcentrate in food chains.

Appendix A discusses carcinogenic and noncarcinogenic toxicity of specific compounds. The toxicity information provided in the HHRAP Companion Database is for informational purposes, to help you explain the basis for selecting COPCs. Please keep in mind that toxicity benchmarks and slope factors might change as additional toxicity research is conducted. We recommend consulting the hierarchy of human health toxicity data (see Appendix A, Section A.2.6) before completing the risk assessment, to make sure that you use the most current toxicity data.

We generally recommend the following six-step approach (illustrated in Figure 2-3) for identifying the COPCs to evaluate in a site-specific risk assessment (U.S. EPA 1994i).

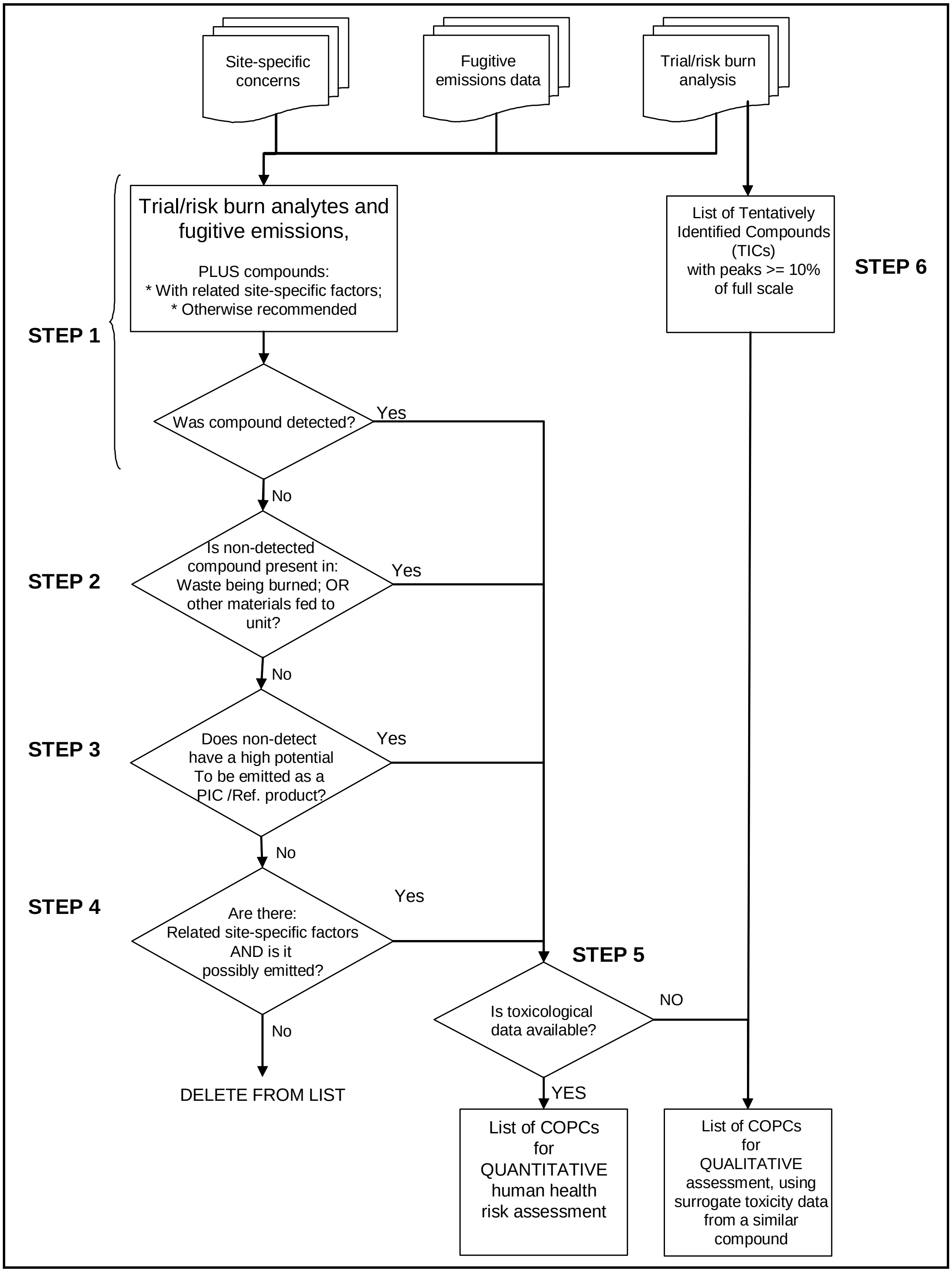
Step 1: Evaluate analytical data from the stack tests performed during the regulatory test burn program, and compounds associated with fugitive emissions (see Section 2.2.6). Prepare a list that includes all the compounds specified in the analytical methods performed in the stack tests, and all compounds found in the fugitive emissions evaluation. Also include compounds of concern due to site-specific factors (e.g., community and regulatory concern, high background concentrations), as well as PCDD/PCDFs, PAHs, and PCBs if not otherwise included. Notate whether each compound was detected or not detected.

In the recommended approach, a detection in any one of the sampling components (e.g., front half rinse, XAD resin, condensate, Tenax tube), in any run constitutes a detection for that specific compound. Evaluating blank contamination results [included in the quality assurance (QA) data section of the trial burn report] may be relevant when determining the non-detect status of the compounds (see Section 2.5).

Regardless of the analytical methods performed in the regulatory test burn program, we recommend that risk assessments consider PCDD/PCDFs, PAHs, and PCBs (the rationale for including these compounds is discussed in greater detail under Step 3 and in Sections 2.3.1 through 2.3.3).

Steps 2 through 4 are unnecessary for compounds detected in the stack test data analysis or identified in the fugitive emissions evaluation; they may jump to Step 5. All other compounds continue to Step 2.

FIGURE 2-3
COPC IDENTIFICATION



Step 2: Evaluate all wastes that the unit will be permitted to burn. Retain for evaluation any non-detected compound present in the waste (Section 2.4 discusses estimating concentrations for non-detects).

For example, if a facility is permitted to burn explosives which characteristically include nitroaromatic compounds, yet the stack test didn't detect any nitroaromatic compounds, it may be appropriate for nitroaromatic compounds to still be evaluated in the risk assessment. It is prudent to also consider other materials fed to the combustor (e.g. raw materials, or coal in a cement kiln).

Steps 3 and 4 are unnecessary for constituents retained as part of the Step 2 evaluation; they may jump to Step 5. All other compounds, i.e. non-detected compounds that did not satisfy Step 2, continue to Step 3.

Step 3: Retain for evaluation any non-detect with a high potential to be emitted as a Product of Incomplete Combustion (PIC).

As defined earlier, PICs are either present in the feed stream and not completely destroyed, or formed during the combustion process. It's therefore important to consider combustion chemistry in identifying COPCs. For example, PCDDs and PCDFs may not themselves be found in any feed stream yet still be emitted, because they can form when chlorine-containing chemicals react with organic matter in the low-temperature regions of the combustion unit or APCS. We therefore generally recommend that PCDDs and PCDFs be assessed. The potential for various PICs to be found in combustor emissions is dealt with in more detail in Sections 2.3.1 through 2.3.13, as well as EPA (2001c).

Identifying/including some compounds (nitroaromatics, phthalates, hexachlorobenzene, and pentachlorophenol) as PICs in the risk assessment may be warranted, considering waste feed composition and their potential to be emitted (e.g., nitrogenated wastes, plastics, or highly chlorinated organic waste streams) (see Sections 2.3.4 through 2.3.6).

Step 4 is unnecessary for PCDDs/PCDFs, PAH's, PCB's, and other compounds with high potential to be emitted as PICs; they may jump to Step 5. All other compounds, i.e. non-detected compounds that did not satisfy Steps 2 or 3, continue to Step 4.

Step 4: Retain for evaluation those compounds that (1) are a concern due to site-specific factors, and (2) may be emitted by the combustor.

As mentioned in Step 1, site-specific factors may contribute COPCs. For example, if there is community/regulatory concern about high background concentrations of a substance which would not have otherwise been assessed (i.e. it was neither a risk/trial burn analyte, nor found in the fugitive emissions evaluation), and there is reasonable potential for it to be emitted, it may be appropriate to include the compound. Also, if a compound found in the trial/risk burn analysis or fugitive emissions evaluation (and therefore included in the COPC list) doesn't satisfy Steps 2 or 3, yet is of concern for site-specific factors and has reasonable potential to be emitted, it may be appropriate for it to continue to Step 5.

If a compound doesn't have a reasonable potential of being present in the stack emissions, we generally recommend that the risk assessment report justify this assertion. This information will generally provide the risk manager with sufficient information to conclude that the facility has not overlooked a serious risk.

Compounds of concern due to site-specific factors with reasonable potential to be emitted continue to Step 5. Delete all other compounds (i.e. non-detected compounds that did not satisfy Steps 2 through 4) from consideration in the risk assessment.

Step 5: Research the recommended hierarchy of human health toxicity data (see Appendix A2.6) for available compound-specific health benchmarks. Add compounds with available toxicity data to the COPC list for ***quantitative*** assessment. Retain compounds that have no toxicity data on the COPC list for ***qualitative*** assessment, and use surrogate toxicity data from a toxicologically similar compound.

As detailed in Appendix A, we recommend a hierarchy of sources for toxicity data appropriate to use in the risk assessment. The tox hierarchy represents a library of sources for scientifically defensible, compound-specific human health benchmarks.

We generally recommend that the assessment of COPCs using surrogate toxicity data not be ***quantitative*** but rather ***qualitative***, and be reported in the Uncertainty section of the risk assessment. The definition of a "toxicologically similar compound" will depend on the original compound, which in turn changes from assessment to assessment. We recognize the uncertainties involved in even defining what constitutes "toxicologically similar." We therefore recommend consulting with the permitting authority when identifying toxicologically similar compounds. It's also within the permitting authority's purview to determine that it's technically appropriate (on a compound-specific basis) to use surrogate toxicity data quantitatively.

Previous guidance on how to qualitatively assess risk is inconsistent. Common practice is also highly variable. One option is to generate quantitative estimates for compounds using surrogate toxicity data. These results, however, aren't typically reported with the rest of the COPCs, nor do they contribute to risk totals. Instead, the surrogate-based risk results are typically reported in the Uncertainty section of the risk assessment report, to inform risk management decision-makers.

> ***Please Note:*** The above is only one option. We recommend consulting with the permitting authority regarding the appropriate level of effort in acquiring surrogate toxicity data, and other methods and processes to use in qualitative assessment.

SPECIAL NOTE:
REGARDING FATE & TRANSPORT DATA

Step 5 in Identifying COPCs focuses on availability of toxicity data because it tends to be the controlling factor: without toxicity parameter values, quantitative assessment is not possible. Depending on the compound, though, availability of fate & transport parameter values could also be a limiting factor.

If these parameter values are available, then it is scientifically reasonable, and in the interest of protecting human health and the environment, to evaluate exposure of receptors to the COPC via various direct and indirect pathways. However, if the necessary fate & transport properties for a particular exposure pathway aren't available, then it seems reasonable to exclude that COPC from consideration for the affected pathway (or pathways). For example, if a biotransfer value for milk (see Chapter 5) is not available for a COPC then it can be assumed that, based on current information, the COPC won't be assessed via the ingestion of milk exposure pathway. This principle holds true for other variables as well. Please note, though, that the lack of fate and transport data doesn't automatically equate to an absence of potential exposure and risk. We generally recommend that as long as sufficient fate & transport properties are available, the calculations for each exposure pathway be completed, and any uncertainties introduced into the risk assessment described in the uncertainty discussion of the risk assessment report (see Chapter 8).

Fate & transport parameter data may be quite limited for some compounds, and acquiring that data can be a labor-intensive and time-consuming process. In an effort to streamline the risk assessment process, the fate & transport parameter values needed to follow this Protocol to assess the 200+ compounds most commonly found in hazardous waste combustor risk assessments are made available in a database companion to the HHRAP (available for download from the HHRAP web site). For those compounds not found in the database, the Superfund Chemical Data Matrix (SCDM) is a good first source to acquire the necessary values. When actual values aren't directly available, HHRAP Appendix A also lists our recommended methods for estimating parameter values.

We generally recommend consulting with the permitting authority on the appropriate level of effort to expend acquiring/estimating fate & transport parameter values.

Step 6: Evaluate the tentatively identified compound (TIC) peaks obtained during gas chromatography (GC) analysis, to determine whether any of the TICs have toxicities similar to the detected compounds. If they do, qualitatively assess using surrogate toxicity data, as recommended for identified compounds in Step 5.

All organic compounds that are identified and quantified are ultimately subtracted from the total organic emissions mass value. It is therefore beneficial for the laboratory to identify and quantify the maximum number of compounds, including TICs. Although it's in your interest to characterize as many TICs as possible, extensive characterization of TICs involves a significant commitment of time and expertise and can reach a point of diminishing returns. We therefore generally recommend characterizing TICs when the peak intensity is 10 percent or more of the full chromatographic scale, and obtaining a quantitative estimate of the value using the nearest eluting internal standard and a response factor of 1. Unless the identification of the TIC is confirmed by the analysis of an authentic standard, it may be appropriate to qualify the quantitative value as "estimated."

We recognize that for many compounds, only limited information on potential health effects is available. Also, for those chemicals with identified health effects, the relationship between dose and response may be poorly understood. We suggest that the risk assessment use the sum of the available toxicological information and evaluate the uncertainty associated with these issues. As stated previously, toxicity benchmarks and slope factors may change as additional toxicity research is conducted. You may wish to consult with the most current versions of the resources found in the tox hierarchy (see Appendix A, Section A2.6) before completing the risk assessment, to make sure that the toxicity data used in the risk assessment is the most current available.

Previous Agency guidance (1989e; 1994j; 1994n; 1998) recommended that the COPC list for indirect exposure analysis consist of only those constituents considered to present the most significant risks. These constituents were selected based on the

1. quantity of the hazardous waste to be burned,
2. toxicity of the hazardous waste to be burned, and
3. potential for the hazardous waste to bioaccumulate.

For direct exposure analysis, however, previous guidance recommended including all constituents for which stack emission data and inhalation health benchmarks exist. We now recommend that a single COPC list apply to **both** indirect **and** direct exposure analysis. We believe that, through the use of computer-based calculations, you can efficiently assess all identified COPCs via both direct and indirect exposure pathways. Savings gained through computer-based calculations will provide for an efficient use of facility and regulatory resources. Assessing the entire list of COPCs - rather than a subset as previously recommended - may help minimize public concern over the exclusion of some COPCs and reduce confusion for those interested in reviewing the results of the risk assessment.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Complete evaluation of hazardous wastes to be burned in the combustor
- Complete evaluation of any raw materials or primary fuels burned in the combustor
- Waste analysis procedures used to monitor the composition of hazardous waste feed streams
- Analytical data and calculations used to complete the COPC identification process

The following subsections provide specific information and guidance on identifying COPCs for each facility—with discussions for specific classes of compounds—that we typically recommend including in risk assessments. Emerging issues surrounding the class of compounds referred to as “endocrine disruptors” are also discussed.

The following subsections also focus on compounds that past experience has shown can drive risk assessments. These compounds include PCDDs/PCDFs, PAHs, PCBs, nitroaromatics, phthalates, hexachlorobenzene and pentachlorophenol, and metals. Volatile organic compounds are also discussed. We also discuss specific issues that affect the COPC identification process, and evaluating these compounds in the risk assessment.

2.3.1 Criteria Pollutants

Under the Clean Air Act, EPA establishes air quality standards to protect public health, including the health of "sensitive" populations such as people with asthma, children, and older adults. EPA has set national air quality standards (40 C.F.R. Part 40) for six principal air pollutants (also called the criteria pollutants): nitrogen dioxide (NO_2), ozone (O_3), sulfur dioxide (SO_2), particulate matter (PM), carbon monoxide (CO), and lead (Pb). We discuss lead in Section 2.3.5.2, and PM in Section 2.3.7.

Nitrogen dioxide is a reddish brown, highly reactive gas that is formed in the ambient air through the oxidation of nitric oxide (NO). Nitrogen oxides (NOx), the generic term for a group of highly reactive gases that contain nitrogen and oxygen in varying amounts, play a major role in the formation of ozone, PM, haze, and acid rain. The major sources of man-made NOx emissions are high-temperature combustion processes such as those that occur in automobiles and power plants. Short-term exposures (e.g., less than 3 hours) to low levels of NO_2 may lead to changes in airway responsiveness and lung function in individuals with preexisting respiratory illnesses. These exposures may also increase respiratory illnesses in children. Long-term exposures to NO_2 may lead to increased susceptibility to respiratory infection and may cause irreversible alterations in lung structure. NOx react in the air to form ground-level ozone and fine particle pollution, which are associated with adverse health effects (U.S. EPA 2005).

Ozone occurs naturally in the stratosphere approximately 10 to 30 miles above the earth's surface and forms a layer that protects life on earth from the sun's harmful rays. Ozone is also formed at ground level by a chemical reaction of various air pollutants combined with sunlight. The pollutants that contribute to ozone formation are oxides of nitrogen (NOx) and volatile organic compounds (VOCs). "Ground-level" ozone is an air pollutant that damages human health and the environment. Even at relatively low levels, ozone may cause inflammation and irritation of the respiratory tract, particularly during physical activity. The resulting symptoms can include breathing difficulty, coughing, and throat irritation. Breathing ozone can affect lung function and worsen asthma attacks. Ozone can increase the susceptibility of the lungs to infections, allergens, and other air pollutants. Medical studies have shown that ozone damages lung tissue and complete recovery may take several days after exposure has ended (U.S. EPA 2004a).

Sulfur dioxide (SO_2) belongs to the family of SOx gases. These gases are formed when fuel containing sulfur (mainly coal and oil) is burned at power plants and during metal smelting and other industrial processes. High concentrations of SO_2 can result in temporary breathing impairment for asthmatic children and adults who are active outdoors. Short-term exposures of asthmatic individuals to elevated SO_2 levels during moderate activity may result in breathing difficulties that can be accompanied by symptoms such as wheezing, chest tightness, or shortness of breath. Other effects that have been associated with longer-term exposures to high concentrations of SO_2, in conjunction with high levels of PM, include aggravation of existing cardiovascular disease, respiratory illness, and alterations in the lungs' defenses. The subgroups of the population that may be affected under these conditions include individuals with heart or lung disease, as well as the elderly and children (U.S. EPA 1986d; 2005).

Carbon monoxide is a colorless and odorless gas, formed when carbon in fuel is not burned completely. It is a component of motor vehicle exhaust, which contributes about 60 percent of all CO emissions nationwide. Other sources of CO emissions include industrial processes, non-transportation fuel combustion, and natural sources such as wildfires. CO enters the bloodstream through the lungs and reduces oxygen delivery to the body's organs and tissues. The health threat from levels of CO sometimes found in the ambient air is most serious for those who suffer from cardiovascular disease such as angina pectoris. At much higher levels of exposure not commonly found in ambient air, CO can be poisonous, and even healthy individuals may be affected. Visual impairment, reduced work capacity, reduced manual dexterity, poor learning ability, and difficulty in performing complex tasks are all associated with exposure to elevated CO levels (U.S. EPA 2000d).

The permitting authority decides whether to include criteria pollutants in the quantitative risk assessment. For example, as noted in the November 14, 1997, decision of the Environmental Appeals Board in reference to the Ash Grove Cement Company Permit No. KSD031203318 and risks associated with exposure to cement kiln dust controlled through the state solid waste permit, compliance with other environmental statutes (e.g., CAA, CWA) may be an appropriate method to consider and control risks from non-RCRA related pollutants (Environmental Appeals Board 1997).

2.3.2 Endocrine Disruptors

Endocrine disruptors are chemicals are thought to mimic natural hormones, inhibit the action of hormones, or alter the normal regulatory function of the immune, nervous, and endocrine systems. Possible human health endpoints affected by these agents include breast cancer and endometriosis in women, testicular and prostate cancers in men, abnormal sexual development, reduced male fertility, alteration in pituitary and thyroid gland functions, immune suppression, and neurobehavioral effects (U.S. EPA 1997g).

Problems were encountered while attempting to classify chemical compounds as endocrine disruptors. Only limited empirical data are available to support the designation of specific chemicals as endocrine disruptors, and some of the data are conflicting. There is a lack of clear structure-activity relationship, as well as a lack of unifying dose-response relationship, among the diverse groups of chemicals considered endocrine disruptors. Also, there are multiple modes of action for chemicals currently considered endocrine disruptors.

Because the information currently available on endocrine disruptors is inconsistent and limited, U.S. EPA has not yet developed a methodology for quantitative assessments of human health risk resulting from exposure to potential endocrine disruptors (U.S. EPA 1996i). However, the methods for addressing endocrine disruptors are developing at a rapid pace. We therefore generally recommend contacting the Economics, Methods and Risk Analysis Division (EMRAD) of the Office of Solid Waste for the latest guidance on how to address endocrine disruptors in site-specific risk assessments. Additional information (e.g., U.S. EPA 1997g) is available for review at the web site http://epa.gov/endocrine/pubs.html.

2.3.3 Hexachlorobenzene and Pentachlorophenol

In past guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) we recommended always including hexachlorobenzene and pentachlorophenol in risk assessments of hazardous waste combustors. However, we no longer recommend automatically including them. Rather, we generally recommend carefully considering the information and issues presented below before deciding whether to include hexachlorobenzene and pentachlorophenol as COPCs for quantitative assessment.

Hexachlorobenzene is an impurity in pentachlorophenol, while pentachlorophenol is formed from hexachlorobenzene in the body as well as in some factories (ATSDR 1994a; ATSDR 1994b). Hexachlorobenzene and pentachlorophenol, like all chlorinated aromatics, are synthesized by the reaction of elemental chlorine with a parent aromatic (Deichmann and Keplinger 1981; Grayson 1985). The addition of the first chlorine atom to the benzene or phenol molecule is rapid, but further chlorination becomes progressively more difficult, requiring ferric chloride or another Lewis acid catalyst to complete the reaction (March 1985). Therefore, these chlorinated compounds are difficult to make even under controlled conditions. Hexachlorobenzene, but not pentachlorophenol, has been reported in emissions from the combustion of municipal solid waste and from other processes (such as the chlorination of wood pulp) that also produce PCDDs and PCDFs (ATSDR 1994a; ATSDR 1994b). The combustion properties of these chlorinated compounds indicate that they aren’t likely to be formed as PICs if they aren’t present in the waste feed stream.

We consider it prudent to include hexachlorobenzene and pentachlorophenol as COPCs for combustors that burn waste feeds containing hexachlorobenzene and pentachlorophenol, wood preservatives, pesticides, or highly variable waste streams such as municipal solid waste. However, we don’t recommend precluding these compounds from analytical testing during the trial burn based only on process knowledge and waste feed characteristics. Because PCDDs and PCDFs can be formed from fly ash-catalyzed reactions between halogens and undestroyed organic material from the furnace, other

Agency guidance (U.S. EPA 1994i; 1998c) recommends including potential precursor compounds in the risk assessment and trial burn (see Section 2.3). These precursor compounds might include chlorinated phenols (such as pentachlorophenol) and chlorinated aromatics (such as hexachlorobenzene). Also, the toxicity and uncertainties associated with combustion chemistry suggest that stack gas testing always confirm the absence of these compounds from stack emissions.

2.3.4 Hydrogen Chloride/Chlorine Gas

Hydrogen chloride (which becomes hydrochloric acid when dissolved in water) and chlorine are major products of the chemical industry, with uses too numerous to list. When chlorine gas dissolves in water (whether during drinking water treatment or when someone inhales chlorine), it hydrolyzes to form equal amounts of hydrochloric acid and hypochlorous acid; the adverse effects of which are similar but not identical (Stokinger 1981; ACGIH 1991).

Hydrochloric acid has many uses. It is used in the production of chlorides, fertilizers, and dyes, in electroplating, and in the photographic, textile, and rubber industries. Hydrochloric acid is corrosive to the eyes, skin, and mucous membranes. Acute (short-term) inhalation exposure may cause eye, nose, and respiratory tract irritation and inflammation and pulmonary edema in humans. Acute oral exposure may cause corrosion of the mucous membranes, esophagus, and stomach and dermal contact may produce severe burns, ulceration, and scarring in humans. Chronic (long-term) occupational exposure to hydrochloric acid has been reported to cause gastritis, chronic bronchitis, dermatitis, and photosensitization in workers. Prolonged exposure to low concentrations may also cause dental discoloration and erosion (U.S. DHHS 1993).

Chlorine is a potent irritant to the eyes, the upper respiratory tract, and lungs. Chronic (long-term) exposure to chlorine gas in workers has resulted in respiratory effects, including eye and throat irritation and airflow obstruction (Cal EPA 2000). Depending on the exposure concentration, acute (short-term) exposure to chlorine elicits reactions ranging from tickling of the nose and throat (Calabrese and Kenyon 1991) to chest pain, vomiting, dyspnea, and cough (U.S. DHHS 1993). Chlorine is also extremely irritating to the skin and can cause severe burns in humans (U.S. DHHS 1993).

2.3.5 Metals

Previous guidance (U.S. EPA 1994g; 1994i; 1998c; NC DEHNR 1997) recommends including the following inorganic substances in the risk assessment: antimony, arsenic, barium, beryllium, cadmium, hexavalent chromium, lead, mercury (elemental and divalent), nickel, selenium, silver, thallium, and zinc. All of these substances, except nickel, selenium, and zinc, are regulated by 40 CFR Part 266, Subpart H (the BIF regulations). We recommend evaluating nickel and selenium, to determine whether additional terms and conditions may need to be incorporated into the permit, pursuant to 42 USC § 6925(c)(3) and 40 CFR Part 270.32(b)(2)–i.e., U.S. EPA's "omnibus" authority. In addition, U.S. EPA (2001c) recommends also characterizing the metals aluminum, copper, manganese, and vanadium. Another potential option is applying the BIF regulation Tier I or MACT MTEC assumptions, which assume that all metals in the waste feed pass through the combustion unit and APCS to the emission stream (U.S. EPA 1992c).

> ***Please Note:*** It may be appropriate to include metals in the risk assessment even if they aren't present in the combustor's feed streams. Although metals cannot be formed as PICs, we are aware of combustors with metal emissions resulting from leaching from stainless steel feed piping.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Waste feed, raw material, and secondary fuel stream analytical data
- Metal emission rate sampling data or assumptions based on waste feed data
- Explanations for excluding specific metals from evaluation during the risk assessment

The following subsections provide additional information on our recommended procedures for evaluating four metals—chromium, lead, mercury, and nickel. When evaluating stack emissions for the risk assessment, we highly recommend considering how each of these metals may be affected by the combustion process, including possible interactions with other constituents.

2.3.5.1 Chromium

The oxidation state of chromium is a crucial issue in evaluating the toxicity of this metal, and the risks associated with exposure. Hexavalent chromium (Cr^{+6}) is the most toxic valence state of chromium and has been shown to be a human carcinogen through inhalation exposure (U.S. EPA 2005i). Trivalent chromium (Cr^{+3}) is a commonly found, less-oxidized form of chromium. Trivalent chromium has not been shown to be carcinogenic in either humans or laboratory animals (U.S. EPA 2005i). U.S. EPA (1990a; 1990b) has indicated that chromium emitted from a combustor is not likely to be in the hexavalent form. However, there is not sufficient evidence to reliably estimate the partitioning of chromium emissions into these two valence states. In addition, we recognize that chromium may exist partially or in some cases entirely as trivalent chromium in various media. For example, Amdur et al. (1991) states that:

> "Trivalent chromium is the most common form found in nature, and chromium in biological materials is probably always trivalent. There is no evidence that trivalent chromium is converted to hexavalent forms in biological systems. However, hexavalent chromium readily crosses cell membranes and is reduced intracellularly to trivalent chromium.."

We generally consider it best to use measured, speciated emissions data in the risk assessment. If site-specific speciated emissions data is unavailable, you may generate a default speciation. We generally recommend using the following method (developed by us through interpretation of data available in the MACT database, as documented in Appendix D) to generate a default speciation:

- When the measured amount of total chromium is <10 :g/dscm, we recommend a default of 5 :g/dscm hexavalent chromium.
- When the measured amount of total chromium is in the range of 10 :g/dscm to 100 :g/dscm, we recommend assuming 45 percent is hexavalent chromium.
- When the measured amount of total chromium is >100 :g/dscm, we recommend assuming 30 percent is hexavalent chromium.

2.3.5.2 Lead

We generally recommend that risk assessments evaluating lead as a COPC use the IEUBK model when soil concentrations are calculated to be above the benchmark.

The Integrated Risk Information System (IRIS) doesn't currently list an RfD or RfC for lead, because a threshold level for exposure to lead has not been established. While the Agency has characterized lead as a probable human carcinogen, it has not developed a quantitative estimate of cancer risk due to a number of uncertainties, some of which may be unique to lead (U.S. EPA 2005b). The Agency has typically relied on the neurological effects observed in children as the sensitive endpoint for evaluating lead toxicity. Consequently, the Agency developed the integrated Exposure Uptake Biokinetic (IEUBK)

Model for Lead in Children. Developed through the efforts of U.S. EPA (1990c) and Kneip, et al. (1983), this model evaluates potential risks based on predicted blood lead levels associated with exposure to lead (U.S. EPA 1994e). The IEUBK model integrates several assumptions about the complex exposure patterns and physiological handling of lead by the body, and it has been validated at several sites at which lead exposure data and human blood lead levels are available (U.S. EPA 1990c). The U.S. EPA Science Advisory Board (U.S. EPA 1992b) and the U.S. EPA's Technical Review Workgroup for Lead have both reviewed and recommended the IEUBK model.

The Agency has developed a computerized version of the IEUBK model that predicts blood lead levels and distributions for children 0 to 7 years of age (U.S. EPA 1994e). The IEUBK model is available for download at http://www.epa.gov/superfund/programs/lead/products.htm. The IEUBK computer model cannot predict potential blood lead levels in adults. The Agency has developed an *Interim Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil* (U.S. EPA 1996r). This interim model is intended for "assessing adult lead risks associated with nonresidential [industrial] exposure scenarios." However, in general, children are more susceptible to lead exposures than adults because of higher soil ingestion rates and greater absorption by the gut, in addition to nutritional variables and lower body weight. In fact, The Agency's interim approach for assessing adult exposures to lead is based not on limiting adult toxicity, but rather on limiting fetal toxicity by limiting indirect fetal exposure through direct maternal exposures to lead (U.S. EPA 1996r).

As stated before, we generally recommend that risk assessments evaluating lead as a COPC use the IEUBK model when soil concentrations are calculated to be above the benchmark. We don't generally recommend evaluating carcinogenic risks or noncarcinogenic hazards of lead. When run with standard recommended default values (these generally represent national averages, or "typical" values), the Agency'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 blood exceeding 10 µg/dL (U.S. EPA 1994e and 1994o).

2.3.5.3 Mercury

We generally recommend that the risk assessment evaluate exposure to three mercury species via varied pathways:

1. Assess elemental mercury only through direct inhalation of the vapor phase;

2. Assess divalent mercury through both direct inhalation and indirect exposure to vapor and particle-bound mercuric chloride; and.

3. Assess methyl mercury only through indirect exposure.

Air emissions of mercury contribute to local, regional, and global deposition. The U.S. Congress explicitly found this to be the case and required the Agency to prioritize maximum achievable control technology (MACT) controls for mercury (U.S. Congress 1989).

The *Mercury Study Report to Congress* (U.S. EPA 1997c) found that anthropogenic mercury releases are thought to be dominated on the national scale by industrial processes and combustion sources that release mercury into the atmosphere. A portion of these anthropogenic releases is in the form of elemental mercury, and a portion in the form of mercuric chloride. Coal combustion is responsible for more than half of all mercury emissions from U.S. anthropogenic sources. The fraction of coal combustion emissions in oxidized form (i.e. mercuric chloride) is thought to be less than the fraction in oxidized form from other combustion emission sources (including waste incineration).

Stack emissions include both vapor and particulate forms of mercury. Most of the total mercury emitted from the stack is in the vapor phase, although exit streams containing soot or particulates can bind up some fraction of the mercury. Vapor mercury emissions are thought to include both elemental (Hg^0) and oxidized (e.g., Hg^{+2}) chemical species. Particulate mercury emissions are thought to be composed primarily of oxidized compounds, due to the relatively high vapor pressure of elemental mercury (U.S. EPA 1997c).

The methods for analyzing mercury speciation in emission plumes are being refined, and there is still controversy in this field. The speciation of mercury emissions is thought to depend on the fuel used, flue gas cleaning, and operating temperatures. True speciation of mercury emissions from the various source types is still uncertain and thought to vary not only among source types, but also between individual plants (U.S. EPA 1997c). Total mercury exiting the stack is assumed to consist entirely of elemental and divalent species, with no emissions of methyl mercury. The exit stream is thought to range from almost all elemental mercury to nearly all divalent mercury. Much of the divalent mercury is thought to be mercuric chloride ($HgCl_2$) (U.S. EPA 1997c), particularly in the combustion of wastes containing chlorine. The divalent fraction is split between vapor and particle-bound phases (Lindqvist et al. 1991).

Please note that data on mercury speciation in stack emissions is very limited. Also, the behavior of mercury emissions close to the point of release has not been extensively studied. It is possible for chemical reactions to occur in the emission plume. This results in a significant degree of uncertainty implicit in modeling mercury emissions. Additional examples of uncertainties include the precision of measurement techniques, estimates of pollution control efficiency, limited data specific to source class and activity level. U.S. EPA (1997c) discusses uncertainty, and sensitivity analyses of several of the assumptions used in the modeling of mercury emissions. Additional discussions and examples of mercury modeling can be found in the proceedings of the IT3 conference (Kaleri 2000).

Site-Specific Mercury Sampling

If site-specific mercury sampling information is available, one option is to estimate the oxidation state and phase distributions. We recommend basing the estimates on the concentration of mercury in various components of the Agency's multiple metals sampling train (i.e., U.S. EPA Method 29 or Method 0060) using the following guidelines:

- Mercury found in the acidic potassium permanganate impingers would be expected to be the elemental form of mercury (Hg^0).
- Divalent mercury ($HgCl_2$) is soluble in water and would be expected to be found in the dilute nitric acid/hydrogen peroxide impinges. This is also referred to as the ionic portion.
- Mercury found in the probe and filter can be assumed to be vapor-phase mercury adsorbed onto particulate matter or a solid-phase compound. This fraction is referred to as Hg(PM).

We understand that these methods can be biased by high SO_2 and trace Cl_2 in the feed. This bias results in the over-reporting of ionic vapor (or divalent form) and the under-reporting of elemental vapor mercury. For risk assessment purposes, we consider this protective.

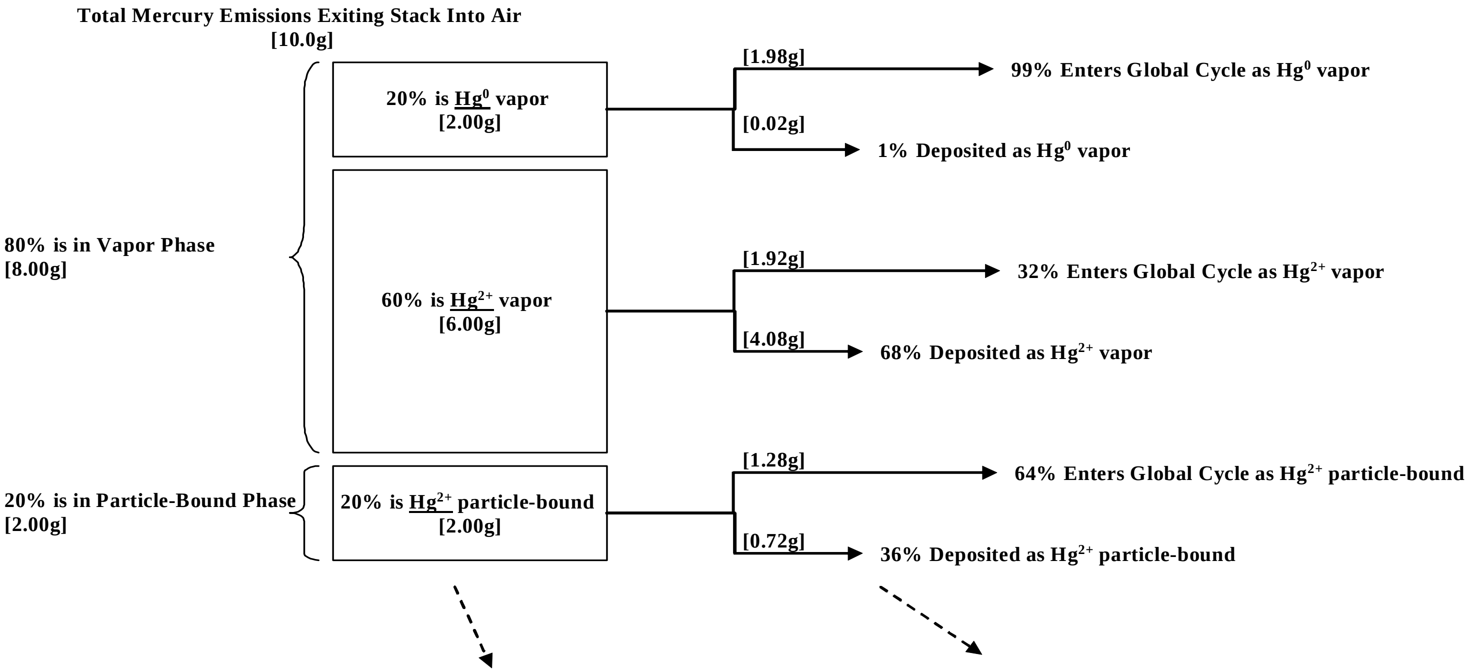
Default Phase Allocation and Speciation of Mercury Exiting the Stack

As discussed above, stack emissions are thought to be speciated into both divalent and elemental mercury, and include both vapor and particle-bound forms. Vapor-phase divalent mercury is thought to be more rapidly and effectively removed by both dry and wet deposition than particle-bound divalent mercury. This is a result of the reactivity and water solubility of vapor-phase divalent mercury (Lindberg et al. 1992; Peterson et al. 1995; Shannon and Voldner 1994). Also, divalent mercury emitted either in the vapor phase or particle-bound, is thought to be subject to much faster atmospheric removal than elemental mercury (Lindberg et al. 1992; Peterson et al. 1995; Shannon and Voldner 1994). A small fraction (about one percent) of vapor-phase elemental mercury may be atmospherically transformed into divalent mercury by tropospheric ozone and adsorbed to particulate soot in the air and subsequently deposited in rainfall and snowfall (U.S. EPA 1997c).

Based on review of mercury emissions data presented for combustion sources in U.S. EPA (1997c) and published literature (Peterson et al. 1995), estimates for the percentage of vapor and particle-bound mercury emissions range widely from 20 to 80 percent. Therefore, unless site-specific mercury sampling information is available, we generally recommend a protective approach that assumes phase allocation of mercury emissions from hazardous waste combustion of 80 percent of total mercury in the vapor phase and 20 percent of total mercury in the particle-bound phase. As illustrated in Figure 2-4, of the 80 percent total mercury in the vapor phase, 20 percent of the total is in the elemental form and 60 percent of the total is in the divalent form (Peterson et al. 1995). Of the 20 percent of the total mercury that is particle-bound, 99 percent (assumed to be 100 percent in Figure 2-4) is in the divalent form. This allocation is:

- Consistent with mercury emissions speciation data for hazardous waste combustion sources reported in literature (Peterson et al. 1995); and
- Believed to be reasonably protective, since it results in the highest percentage of total mercury being deposited in proximity to the source, and therefore, indicative of the maximum indirect risk.

FIGURE 2-4
PHASE ALLOCATION AND SPECIATION OF MERCURY IN AIR



F_v (Total Mercury) = 0.8

<u>Without Considering Global Cycle</u>:
- **20% of Total Mercury Emitted is deposited as Hg^0 [2g / 10g]**
- **80% of Total Mercury Emitted is deposited as Hg^{2+} [(6g + 2g) / 10g]**

<u>Calculated F_v</u>:
- **F_v (Hg^0) = [2g / 2g] = 1.0**
- **F_v (Hg^{2+}) = [6g / (6g+2g)] = 0.75**

<u>Considering Global Cycle</u>:
- **0.2% of Total Mercury Emitted is deposited as Hg^0 [0.02g / 10g]**
- **48% of Total Mercury Emitted is deposited as Hg^{2+} [(4.08g + 0.72g) / 10g]**

<u>Calculated F_v</u>:
- **F_v (Hg^0) = [0.02/(0.02 + 0)] = 1.0**
- **F_v (Hg^{2+}) = [4.08/(4.08 + 0.72)] = 0.85**

<u>Compound-Specific Emission Rate (Q)</u>
- **Actual Q (Hg^0) = 0.2% * Q (Total Mercury)**
- **Actual Q (Hg^{2+}) = 48% * Q (Total Mercury)**

<u>LEGEND</u>

Hg^0 - Elemental Mercury
Hg^{2+} - Divalent Mercury
[ ] - Example Mass Allocation

The Mercury Global Cycle

According to information in U.S. EPA (1997c), a vast majority of mercury exiting the stack doesn't readily deposit, but is vertically diffused to the free atmosphere, transported outside the study area and into the global cycle. Regardless of the source of phase and speciation distribution values (i.e. either site-specific sampling data or default values), we generally recommend using the following fractions from U.S. EPA (1997c):

- A vast majority of the vapor-phase elemental mercury (over 99 percent) doesn't readily deposit, but becomes part of the global cycle;
- Of the mercury emitted as vapor-phase divalent mercury, about 68 percent deposits and about 32 percent diffuses vertically to the global cycle; and
- 36 percent of the particle-bound divalent mercury deposits, and the rest diffuses vertically to the global cycle.

Deposition and Modeling of Mercury

Based on information in U.S. EPA (1997c) and as shown in Figure 2-4, we generally assume that deposition to the various environmental media is almost entirely divalent mercury in either the vapor or particle-bound form. Without considering the global cycle, 80 percent of total mercury would be deposited as divalent mercury and the remaining 20 percent would be deposited as elemental mercury.

We generally recommend using the percentages provided in U.S. EPA (1997c) to account for the global cycle. Using these figures, the percentage of total mercury deposited would be reduced to a total of 48.2 percent (40.8 percent as divalent vapor, 7.2 percent as divalent particle-bound, and 0.2 percent as elemental vapor). As discussed in Appendix A-2, these speciation splits result in fraction in vapor phase (*Fv)* values of 0.85 (40.8/48.0) for divalent mercury, and 1.0 (0.2/0.2) for elemental mercury. Also, to account for the remaining 51.8 percent of the total mercury mass that is not deposited, the deposition and media concentration equations (presented in Appendix B), multiply the compound-specific emission rate (*Q*) for elemental mercury by a default value of 0.002; and divalent mercury by a default value of 0.48.

Methylation of Mercury

The net mercury methylation rate (the net result of methylation and demethylation) for most soils appears to be quite low; with much of the measured methyl mercury in soils potentially resulting from wet deposition (U.S. EPA 1997c). Based on the information in U.S. EPA (1997c), We assume that 98 percent of the deposited mercury remains divalent mercury, and two percent speciates to organic mercury (methyl mercury) in soil. A significant and important exception to mercury methylation rate being low in soils appears to be wetland soils. Wetlands appear to convert a small but significant fraction of the deposited mercury into methyl mercury; which can migrate to nearby water bodies and potentially bioaccumulate in the aquatic food chain (U.S. EPA 1997c). Therefore, we assume the percentage of methyl mercury in wetland soils is higher than the 2 percent assumed for non-wetland soils. However, wetlands soils aren't specifically considered in any of the exposure pathways represented in the recommended human health exposure scenarios (see Chapter 4).

Both watershed erosion and direct atmospheric deposition can be important sources of mercury to a water body (U.S. EPA 1997c). There appears to be a great deal of variability in the processing of mercury among water bodies. As a result, you can generally expect different water body types to have different ranges of methylation, with wetlands generally expected to have higher percentages of methyl mercury than lakes, and lakes subsequently more than rivers or streams (Driscoll et al. 1994; Hurley et al. 1995; Krabbenhoft et al. 1999; Watras et al. 1995). Studies have also shown that rivers or lakes with wetland components (particularly riparian wetlands) have an increased methyl mercury content (Hurley et al. 1995; Krabbenhoft et al. 1999; St. Louis et al. 1996). The percentage of the water body that constitutes a riparian wetland also contributes (i.e., the higher percentage - the higher the methyl mercury concentration). The watershed is also an important factor in determining the methyl mercury concentration of the water body. Waterbodies that are surrounded by agricultural or forested land tend to have higher methylation fractions. Waterbodies that are surrounded by mining activities have high amounts of inorganic mercury in the water, and therefore have a lower methylation efficiency (Krabbenhoft et al. 1999). As briefly discussed later in this section, this variability in methylated mercury concentrations is primarily due to the characteristically wide range of chemical and physical properties of water bodies. Additionally, mercury entering the water body can be methylated predominantly through biotic processes (U.S. EPA 1997c).

In the absence of site-specific measurements to support evaluation of water body properties and biotic conditions relevant to mercury methylation, we generally recommend assuming that 85 percent of total mercury in surface water is divalent mercury, and the remaining mass is methyl mercury. This percentage (i.e., 15 percent as methyl mercury) is based on the average of reported values for the fraction of total mercury that is methyl mercury in surface water (Akagi et al. 1979; Bloom and Effler 1990; Bloom et al. 1991; Gill and Bruland 1990; Kudo et al. 1982; Lee and Hultberg 1990; Parks et al. 1989; Watras and Bloom 1992). These literature sources were originally presented in the SAB Review Draft of the

Mercury Study Report to Congress (U.S. EPA 1996s). The final Mercury Study Report to Congress (U.S. EPA 1997c, Volume III; Appendix D) also presents literature values for the fraction of methyl mercury in the water column. However, the data are specific to the epilimnion and hypolimnion. For the epilimnion, reported values range from 4.6 percent to 15 percent, with a point estimate of 7.8 percent. For the hypolimnion, reported values range from 27 percent to 44 percent, with a point estimate of 36 percent.

We are modifying our previous recommendation for applying the default mercury speciation (85 percent divalent mercury and 15 percent methyl mercury) to each calculated water body loading. Instead, we recommend that a dissolved water concentration first be calculated for total mercury using the fate and transport parameters specified for mercuric chloride. Then, the dissolved total mercury concentration should be apportioned based on an 85 percent divalent and 15 percent methyl mercury speciation split in the water body. Appendix B (Table B-4-24) presents the equations we recommend for applying the speciation assumptions.

For most environmental systems, the literature suggests that various physical and chemical conditions may influence the methylation of mercury. In some cases you might need to consider these conditions, and the magnitude of their potential impact, to assess the potential for over- or under-predicting mercury methylation in media and subsequent biotransfer up the food chain. There is extreme variation between modeled environmental systems, and at times disagreement in the literature regarding the quantitative influence of specific conditions on methylation. Table 2-4 summarizes the qualitative effects that some physical and chemical conditions, as reported in literature, may have on methylation. We therefore generally recommend conducting extensive research of literature specific to the conditions prevalent at the site, before deviating from the protective assumptions recommended above.

More recent advances in scientific understanding of the physical, chemical, and biological processes controlling mercury speciation and partitioning in water bodies are summarized in U.S. EPA (2005g).

TABLE 2-4

QUALITATIVE EFFECTS OF PHYSICAL & CHEMICAL CONDITIONS ON METHYLATION

Physical or Chemical Condition	Qualitative Influence on Methylation	Referenced Literature
Low dissolved oxygen	Enhanced methylation	Rudd et al. 1983; Parks et al. 1989
Decreased pH	Enhanced methylation in water column	Xun 1987; Gilmour and Henry 1991; Miskimmin et al. 1992
Decreased pH	Decreased methylation in sediment	Ramlal et al. 1985; Steffan et al. 1988
Increased dissolved organic carbon (DOC)	Enhanced methylation in sediment	Chois and Bartha 1994
Increased dissolved organic carbon (DOC)	Decreased methylation in water column	Miskimmin et al. 1992
Increased salinity	Decreased methylation	Blum and Bartha 1980
Increased nutrient concentrations	Enhanced methylation	Wright and Hamilton 1982; Jackson 1986; Regnell 1994; Beckvar et al. 1996
Increased selenium concentrations	Decreased methylation	Beckvar et al. 1996
Increased temperature	Enhanced methylation	Wright and Hamilton 1982; Parks et al. 1989
Increased sulfate concentrations	Enhanced methylation	Gilmour and Henry 1991; Gilmour et al. 1992
Increased sulfide concentrations	Enhanced methylation	Beckvar et al. 1996

The deposition and media concentration equations that we generally recommend (presented in Chapter 5 and Appendix B) have been modified specifically to account for the methylation and subsequent biotransfer of mercury, assuming steady-state conditions. The HHRAP companion database provides the parameter values specific for methyl mercury, and Appendix A-2 includes additional discussion and reference on their origin.

As noted above, methylation can be highly variable between environmental systems. This results in a significant degree of uncertainty implicit in the modeling of mercury methylation. To expand on the qualitative information presented in Table 2-4, and to better understand conditions that may influence mercury methylation specific to a site, we recommend reviewing the related information presented in U.S. EPA (1997c; 2005).

We generally recommend using the equations and protective assumptions presented in this guidance to estimate risks associated with mercury. If estimated risks exceed target levels, it may be appropriate to use more extensive site-specific data (if available) and subsequently a more rigorous modeling effort, to further evaluate points of potential exposure. For example, if sufficient site-specific data is available, it could be used in a model that predicts transformation of chemical forms and biotransfer of mercury. One such model is the SERAFM (Spreadsheet Ecological Risk Assessment for the Fate of Mercury) model developed by EPA's Office of Research and Development, National Exposure Research Laboratory, Ecosystems Research Division. SERAFM updates the IEM-2M mercury fate and transport algorithms described in detail in U.S. EPA (1997c) by incorporating more recent advances in scientific understanding of the physical, chemical, and biological processes controlling mercury speciation and partitioning in water bodies. The SERAFM enhancements to IEM-2M are summarized in U.S. EPA (2005g).

SERAFM is written in an easy-to-implement Microsoft Excel format so that all manipulations, parameters and equations are readily available to the user. By specifying only a few additional water body parameters beyond those already utilized in this guidance (i.e., water body pH, dissolved organic carbon and color), the user is able to model specific water body mercury transformation processes instead of using the default speciation assumptions (i.e., 85 percent divalent/15 percent methyl). In the SERAFM model, mercury species are subject to several transformation reactions including photo-oxidation and dark oxidation of elemental mercury in the water column, photo-reduction and methylation of divalent mercury in the water column and sediment layers, and photo-degradation and demethylation of methyl mercury in the water column and sediment layers. For hazardous waste combustion sources, it is recommended that, watershed soil concentrations and water body loadings due to source deposition be calculated externally to SERAFM using the equations presented in Chapter 5 (Equations 5-1E and 5-28) and then linked to the appropriate SERAFM worksheets for calculation of speciated mercury concentrations in the water body.

The decision to use more complex mercury models in a risk assessment is not precluded just because they are different from the model we recommend in the HHRAP. It is for the permitting authority to decide whether the assessment will use more complex mercury models. If you use more complex mercury models, we recommend ensuring that sufficient and reliable site-specific data is readily available, and then carefully identifying and evaluating the models' associated limitations, and clearly documenting the evaluation in the Uncertainty section of the risk assessment report.

Conclusion

We encourage all facilities to implement a combination of waste minimization and control technology options to reduce mercury emission rates on an ongoing basis. Realistic expectations for mercury emission reduction efforts may be established by considering various technology-based mercury emission limits that apply to waste combustors (for example, standards for European combustors, the MACT standards for hazardous waste combustors, or the MACT standards for municipal waste combustors). We acknowledge that site-specific risk assessments as currently conducted may not identify the entire potential risk from mercury emissions. Mercury that doesn’t deposit locally will ultimately enter the global mercury cycle for potential deposition elsewhere.

2.3.5.4 Nickel

We generally recommend evaluating nickel as an inhalation carcinogen using the inhalation unit risk factor for nickel refinery dust. We generally recommend evaluating nickel for other effects using the oral RfD for nickel soluble salts, the only available nickel-related RfD (see Appendix A-2 and the HHRAP companion database).

Nickel refinery dust is identified as a potential human inhalation carcinogen (U.S. EPA 2005i). Major components of nickel refinery dust include nickel subsulfide, nickel oxide and nickel sulfate. IRIS classifies nickel subsulfide - the primary component (roughly 50%) of nickel refinery dust - a Class A human carcinogen (U.S. EPA 2005e). However, all components responsible for the carcinogenicity of nickel refinery dust have not been conclusively established (U.S. EPA 2005c). Because the component (or components) of nickel refinery dust causing it to be carcinogenic have not been conclusively established, we consider it appropriate to evaluate nickel emissions as a potential carcinogen via the inhalation pathway. In addition, nickel oxides can be reduced to nickel sulfates (some of which are carcinogenic) in the presence of sulfuric acid (Weast 1986). Hazardous waste combustors which burn wet wastes containing significant amounts of nickel and sulfur may need to be especially concerned with nickel emissions.

We generally recommend evaluating nickel as an inhalation carcinogen because some forms of nickel—including nickel carbonyl, nickel subsulfide, and nickel refinery dust—are considered carcinogens (U.S. EPA 2005c,d,e). This is contrary to the Agency’s previous analysis of the toxicity of nickel emissions from hazardous waste combustors. These forms of nickel were not considered in developing the BIF regulations, because the BIF regulations assumed that nickel can only be emitted as nickel oxide, which by itself is not considered to be a carcinogen (U.S. EPA 1991a).

If the permitting authority has information at points of potential inhalation exposure that demonstrate the absence of nickel refinery dust components, or the presence only of noncarcinogenic nickel species, it may be appropriate to use this information as the basis for supplemental noncarcinogenic calculations. For exposure pathways other than inhalation, nickel has not been shown to be carcinogenic (U.S. EPA 2005i).

2.3.6 Nitroaromatics

We generally recommend carefully considering the information in the following paragraphs before deciding the appropriateness of including nitroaromatic organic compounds in the risk assessment. It is reasonable to include nitroaromatics as COPCs if the combustor feed streams include nitroaromatic compounds or close relatives (TDA and TDI).

> ***Please Note***: In earlier guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) we recommended that risk assessments always include nitroaromatic organic compounds, including 1,3-dinitrobenzene; 2,4-dinitrotoluene; 2,6-dinitrotoluene; nitrobenzene; and pentachloronitrobenzene. We no longer recommend **automatically** including nitroaromatic organic compounds in risk assessments.

Nitroaromatic organic compounds such as 1,3-dinitrobenzene; 2,4-dinitrotoluene; 2,6-dinitrotoluene; nitrobenzene; and pentachloronitrobenzene (or close relatives such as toluenediamine [TDA] and toluene diisocyanate [TDI]—derivatives of dinitrotoluene) are typically associated with explosives and other highly nitrogenated hazardous wastes. Dinitrotoluene is used to make two products: trinitrotoluene and TDA. TDA is, in turn, used to make TDI. TDI readily reacts with water, and is therefore very unstable at ambient conditions. TDI is typically reacted with a polyol to form polyurethane (PU) plastics.

Combustion properties of these nitroaromatic compounds indicate that they won’t be formed as PICs if they aren’t present in the waste feed stream, mainly because of the thermodynamic and chemical difficulty of adding a nitro group to an aromatic. The process requires that (1) nitronium ions be generated, and (2) an aromatic ring react with the nitronium ion, with the nitronium ion attaching to the ring. This reaction process is not likely to occur in a hazardous waste combustor because (1) the reaction is typically carried out by using a “nitrating acid” solution consisting of three parts concentrated nitric acid to one part sulfuric acid, and (2) nitronium ions aren’t usually formed in a combustor environment (if they are, a thermodynamically more favorable reaction will occur, thereby eliminating the nitronium ion) (Hoggett, et al 1971; Schofield 1980; March 1985).

Combustion conditions most likely to result in nitrogenated PICs are associated with premature quenching of the primary flame—resulting from low temperature or excess air in the primary combustion chamber of the unit (U.S. EPA 1994j). Under such conditions, sampling for hydrogen cyanide is also recommended (U.S. EPA 1994j).

2.3.7 Particulate Matter

We don’t recommend evaluating PM as a separate COPC in the risk assessment. However, PM is generally quite useful as an indicator variable, because it can be measured in real time and is sensitive to changes in combustion conditions.

Particle pollution is a mixture of solid particles and liquid droplets found in the air. Some particles are emitted directly from a source, while others form in complicated chemical reactions in the atmosphere. In general, particle pollution consists of a mixture of larger materials, called "coarse particles," and smaller particles, called "fine particles." Coarse particles have diameters ranging from about 2.5 micrometers (:m) to more than 40 :m, while fine particles, also known as known as PM2.5, include particles with diameters equal to or smaller than 2.5 :m. EPA also monitors and regulates PM10, which refers to particles less than or equal to 10 :m in diameter. PM10 includes coarse particles that are "inhalable" - particles ranging in size from 2.5 to 10 :m that can penetrate the upper regions of the body's respiratory defense mechanisms (U.S. EPA 2004b).

Exposure to particles can lead to a variety of serious health effects. Scientific studies show links between these small particles and numerous adverse health effects. Long-term exposures to PM, such as those experienced by people living for many years in areas with high particle levels, are associated with problems such as decreased lung function, development of chronic bronchitis, and premature death. Short-term exposures to particle pollution (hours or days) are associated with a range of effects, including decreased lung function, increased respiratory symptoms, cardiac arrythmias (heartbeat irregularities), heart attacks, hospital admissions or emergency room visits for heart or lung disease, and premature death. (U.S. EPA 1982c; 2004c).

Ambient PM is a complex mix of constituents derived from many sources, both natural and anthropogenic. Hence, the physicochemical composition of PM generally reflects the major contributing local and regional sources arising locally as well as regionally. It stands to reason that the contribution of any given component within the mix may not be equivalent in value or potency, but may well be highly dependent on other physicochemical attributes (e.g., co- constituents, specific bioavailability, or chelates), as well as the health status of the exposed individual. Evidence collected to date indicates that the discovery of a uniquely responsible physicochemical attribute of PM is not likely to occur (U.S. EPA 2004c).

2.3.8 Phthalates

We generally recommend carefully considering the information in the following paragraphs before deciding the appropriateness of including phthalates in combustor risk assessments. At the same time, due to their toxicity and bioaccumulative potential, don't automatically discount the evaluation of phthalates in the risk assessment. If phthalates are included as COPCs in the risk assessment, we generally recommend using a metabolism factor (*MF*) of 0.01 for BEHP, and 1.0 for all other COPCs.

> ***Please Note***: In earlier guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) we recommended always including BEHP and DNOP in every risk assessment. We no longer recommend ***automatically*** including phthalates in risk assessments.

Phthalates such as bis(2-ethylhexyl)phthalate (BEHP) and di(n)octyl phthalate (DNOP) are synthesized by reacting alcohol with phthalic anhydride in the presence of an acidic catalyst in a nonaqueous solvent (ATSDR 1993; ATSDR 1995b). Among all phthalate plasticizers, BEHP—also referred to as di(2-ethylhexyl)phthalate or dioctyl phthalate—is produced in the largest volume; it is used in the manufacturing of polyvinyl chloride, the most widely produced plastic. DNOP is a plasticizer that is produced in large volumes, and is used in the manufacture of plastics and rubber materials. Because plastics have become so widely used in society, phthalate plasticizers such as BEHP and DNOP have become widely distributed in food, water, and the atmosphere (Howard 1990). The general public's exposure to phthalate-contaminated food averages 0.3 µg/day/individual, with an estimated maximum exposure of 2 mg/day/individual (ATSDR 1992). Phthalate plasticizers are commonly found in the environment and are practically impossible to avoid, especially at the trace concentrations that modern analyses can detect.

Phthalates and their predecessors are readily burned, as indicated by their flash points of 150 to 225 °C (NIOSH 1994). There is no apparent mechanism for phthalate PICs to be formed by burning other chemical compounds. Therefore, phthalates are very unlikely to be emitted from a combustor, although some degradation products, such as PAHs, are likely to be emitted when phthalates are included in the waste feed. However, facilities that burn plastics or materials with phthalate plasticizers should carefully consider the potential for phthalate plasticizers to exist in the stack gas emissions due to incomplete combustion. Also, the uncertainties associated with combustion chemistry suggest that stack gas testing confirm the absence of these compounds from stack emissions, rather than relying on process knowledge or waste feed characterization data.

Based on the findings of long-term animal carcinogenicity studies, the Agency has classified BEHP as a "probable human carcinogen" (class B2) (NTP 1982). Because of its octanol-water coefficient (K_{ow}) value, BEHP has been presumed to have a high tendency to bioaccumulate (Mackay, Shiu, and Ma 1992; Karickoff and Long 1995). Considering its ubiquity, B2 classification, and high tendency to bioaccumulate, BEHP is on most Agency lists of target chemicals (see Table A-1), including the Contract Laboratory Program (CLP) semivolatile organics analysis list; the Groundwater Monitoring List (40 CFR Part 264, Appendix IX); and the Hazardous Substances and Reportable Quantities List (40 CFR Part 302.4).

Evidence indicates BEHP is more readily metabolized and excreted by mammalian species than other contaminants (ATSDR 1987). As stated above, we generally recommend using an *MF* of 0.01 for BEHP, and 1.0 for all other COPCs. An *MF* represents the estimated amount of COPC that remains in fat and muscle. Based on a study by Ikeda et al. (1980), the Agency (EPA 1995h) used a COPC-specific *MF* to account for metabolism in animals and humans. Considering the recommended values for this variable, *MF* has a quantitative effect on animal and human concentrations only for BEHP. No information could be found on the metabolism or disposition of DNOP in the peer-reviewed literature. However, disposition data were found for an isomer of DNOP, diisooctyl phthalate (DIOP), a branched-chain phthalate (Ikeda et al., 1978). Based upon its similarity in structure, it may be assumed that DNOP would behave comparatively to DIOP and BEHP, and therefore, may be over estimated by approximately a factor of 100.

The *MF* applies only to mammalian species, including beef cattle, dairy cattle, and pigs. It does not relate to metabolism in produce, chicken, or fish. In addition, since exposures evaluated using this guidance are intake driven, only apply a metabolism factor to evaluating indirect human exposure via ingestion of beef, milk, and pork. In summary, using an *MF* doesn't apply for direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish. Using an *MF* is further discussed in Section 5.4.4.7 and Appendix B, Tables B-3-10, B-3-11, and B-3-12.

2.3.9 Polychlorinated Biphenyls

Because of evidence that PCBs can be emitted from combustion sources regardless of feed characteristics, and considering the significant toxicity of PCBs, we recommend conducting stack testing for PCBs to support the risk assessment. We also recommend automatically including PCBs as COPCs for combustors that burn PCB-contaminated wastes or waste oils, highly variable waste streams such as municipal and commercial wastes (for which PCB contamination is a reasonable assumption), and highly chlorinated waste streams. Due to the toxicity and uncertainties associated with combustion chemistries, we generally recommend that stack gas testing confirm the absence of these compounds from stack emissions.

The most commercially useful property of PCBs is that they are chemically stable in relatively adverse conditions, such as temperatures of several hundred degrees in an oxygen-containing atmosphere. The more chlorinated congeners are more resistant to reaction. PCBs were produced commercially by the reaction of the aromatic hydrocarbon biphenyl with chlorine gas in the presence of a suitable catalyst, generally ferric chloride or another Lewis acid (ATSDR 1995d). The degree of chlorination was controlled by manipulating the reaction conditions, including temperature, pressure, and the ratio of the reactants (Erickson 1992; Grayson 1985). The uses and distribution of polychlorinated biphenyls (PCBs) were severely restricted in the United States in the late 1970s—with additional bans and restrictions taking effect over the next decade (ATSDR 1995d).

Due to their stability in adverse conditions, destruction of PCBs by burning generally requires contact with high temperatures (at least 1,200 ^{o}C) for an extended period of time (more than 2 seconds), under conditions with adequate oxygen (Erickson 1992). Waste combustors can contribute significantly to total emission inventories of PCBs (Alcock et al. 1999; U.S. EPA 1997e). An increasing body of information supports the likelihood that PCBs may be emitted as by-products of burning, regardless of PCB contamination in the combustor feed.

It is possible that PCBs can be formed by the same types of reactions that produce dioxins and furans, including gas-phase formation, heterogeneous formation from organic precursors, and *de novo* synthesis from flyash-bound carbon. Lemieux et al. (1999) hypothesized that if PCBs and dioxins and furans are formed by similar mechanisms, then emissions of PCBs should correlate with emissions of dioxins and furans. This hypothesis was tested by reviewing data where both PCBs and dioxins and furans were measured. An apparent trend was indeed found showing increased PCB emissions with increased emissions of dioxins and furans. In most cases, PCBs were found in the stack even when there were no PCBs in the combustor feed. Overall, PCB emissions exceeded dioxin and furan emissions by approximately a factor of 20, and this trend appeared to hold over five orders of magnitude in dioxin and furan emissions.

In addition, there is some limited data, from both laboratory and field studies, showing that PCBs may be formed from burning hazardous waste. Stack tests performed in U.S. EPA Region 10 on a boiler and an incinerator burning waste with 0.07 and 1.4 percent chlorine, respectively, confirmed the presence of PCBs in the stack gases (Kalama Chemical, Inc. 1996; Idaho National Engineering Laboratory 1997). The concentration of detected coplanar PCBs (as defined in Section 2.3.9.1) found in the boiler stack gas was 0.55 ng/dscm @ 7% O_2 at low temperature conditions (1,357° F) and 1.12 ng/dscm @ 7% O_2 at high temperature conditions (1,908° F). The concentration of total PCBs detected in the incinerator stack gas was 211 ng/dscm @ 7% O_2 at low temperature conditions (1,750 °F) and 205 ng/dscm @ 7% O_2 at high temperature conditions (2,075° F). PCBs with more than four chlorines comprised 51 percent of the total PCBs in the low temperature test and 59 percent of the total PCBs in the high temperature test.

Other laboratory studies suggest the possible formation of PCBs as PICs from burning hazardous waste with a high chlorine content. Bergman et al. (1984) heated samples of two chlorinated paraffins (CP) in conditions similar to incinerator conditions. A CP containing 70 percent chlorine did produce PCB (up to 0.3 percent of the amount of CP), as well as chlorinated benzenes (up to 0.5 percent), chlorinated toluenes (up to 0.6 percent), and chlorinated naphthalenes (up to 0.2 percent). Similar treatment of a CP containing 59 percent chlorine produced only chlorinated benzenes (up to 0.1 percent of the amount of CP, based on a detection limit of 0.0005 percent for each individual compound) and almost all of those (about 90 percent) were monochlorobenzene (Bergman 1984). This study indicates that burning highly chlorinated wastes (60 percent or greater chlorine) can produce PCBs. Also, in an experiment involving 4 percent charcoal, 7 percent chlorine, and 1 percent copper catalyst heated to 300° C, PCBs were formed at levels approaching 300 ppb for the penta homologue, 200 ppb for the hexa homologue, 150 ppb for the hepta homologue, and less than 50 ppb for the tetra homologue (Stieglitz et al. 1989).

2.3.9.1 PCB Carcinogenic Risks

In earlier guidance (1994g; 1994i; 1994j; 1994r) we recommended that risk assessments treat all 209 PCB congeners as a mixture having a single carcinogenic potency. This recommendation was based on the Agency drinking water criteria for PCBs (U.S. EPA 1988a), which used available toxicological information with the following limitations:

- Aroclor 1260 was the only PCB for which a cancer *SF* had been developed; there was no agreed upon procedure for applying this *SF* for 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 congeners to demonstrate similar toxicological effects was very limited.

Research on PCBs has continued since the compilation of U.S. EPA (1988a),. The most important finding of this research is that some of the moderately chlorinated PCB congeners can have dioxin-like effects (U.S. EPA 1992e; 1994a; 1996q; ATSDR 1995d). This sub-category includes PCB congeners with four or more chlorine atoms and few substitutions in the ortho positions (positions designated 2, 2', 6, or 6'). They are sometimes referred to as “coplanar” PCBs, because the rings can rotate into the same plane if 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 react with the aryl hydrocarbon receptor; the same reaction believed to initiate the adverse effects of PCDDs and PCDFs. The World Health Organization (WHO) used various test results to derive interim toxicity equivalency factors (TEFs) ranging from 0.1 to 0.00001 for the dioxin-like coplanar PCB congeners (WHO 1998).

TABLE 2-5

TOXICITY EQUIVALENCY FACTORS FOR COPLANAR PCBs

CAS Number	Chemical Structure	WHO 1998 TEFs (unitless)
32598-13-3	3,3',4,4'-tetrachlorobiphenyl	0.0001
70362-50-4	3,4,4',5-tetrachlorobiphenyl	0.0001
32598-14-4	2,3,3',4,4'-pentachlorobiphenyl	0.0001
74472-37-0	2,3,4,4',5-pentachlorobiphenyl	0.0005
31508-00-6	2,3',4,4',5-pentachlorobiphenyl	0.0001
65510-44-3	2',3,4,4',5-pentachlorobiphenyl	0.0001
157465-28-8	3,3',4,4',5-pentachlorobiphenyl	0.1
38380-08-4	2,3,3',4,4',5-hexachlorobiphenyl	0.0005
69782-90-7	2,3,3',4,4',5'-hexachlorobiphenyl	0.0005
52663-72-6	2,3',4,4',5,5'-hexachlorobiphenyl	0.00001
32774-16-6	3,3',4,4',5,5'-hexachlorobiphenyl	0.01
39635-31-9	2,3,3',4,4',5,5'-heptachlorobiphenyl	0.0001

Source: World Health Organization (1998)

Additional congeners are suspected of producing similar reactions, but there is not yet enough data to derive TEF values for them. Since available analytical methods can now quantify most if not all individual PCB congeners, we generally consider it reasonable for the permitting authority to request that additional congeners be reported. For instance, 2,2',4,4',5,5'-Hexachlorobiphenyl (CAS No. 35065-27-1) is the most prevalent PCB congener found in human milk and fat (McFarland and Clark 1989). Work is currently underway to develop a separate slope factor for this particular compound (not a coplanar congener). However, until that work is complete, this compound can only be qualitatively assessed in the risk assessment.

We generally recommend estimating risks from coplanar PCBs by computing a toxicity equivalency quotient (TEQ) for PCBs, and then applying a slope factor for dioxin. High resolution gas chromatograph test methods, available at most commercial laboratories with dioxin/furan analytical capabilities, are able to identify the specific concentration of individual coplanar PCBs in stack gas.

In addition to the coplanar (dioxin-like) PCB congeners, We also generally recommend evaluating the remaining PCBs in the risk assessment. After considering 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, USEPA (1996q) derived three new *SF*s to replace the former single *SF* for PCBs. These new *SF*s became effective in IRIS on October 1, 1996. These *SF*s are subject to revision as additional information from continuing research becomes available. The *SF*s and the criteria for their use are as follows (U.S. EPA 1996q):

TABLE 2-6

ORAL SLOPE FACTORS FOR PCBs

Slope Factor (milligrams per kilogram-day)$^{-1}$	**Criteria for Use**
2	Food chain exposure Sediment or soil exposure Early-life (infant and child) exposure by all routes to all PCB mixtures Congeners with more than four chlorines comprise *more than* 0.5 percent of the total PCBs
0.4 (Not Typically Used)	Ingestion of water-soluble (less chlorinated) congeners Inhalation of evaporated (less chlorinated) congeners
0.07	Congeners with more than four chlorines comprise *less than* 0.5 percent of the total PCBs
Source: U.S. EPA 1996q	

An *SF* of 2 (milligrams per kilogram-day)$^{-1}$ is typically used in most circumstances when conducting a risk assessment. An *SF* of 0.07 (milligrams per kilogram-day)$^{-1}$ is generally scientifically defensible for adult exposures, when congener-specific analyses of emissions demonstrate that at least 99.5 percent of the mass of the released PCB mixture has fewer than five chlorine atoms per molecule (U.S. EPA 1996q). Acceptable congener-specific analyses include high-resolution gas chromatography/mass spectroscopy (or similar means) for total PCB concentrations for each mono- through deca-isomer group. We don't expect that the 0.4 SF will be widely used in combustion risk assessments, because the PCB mixture will usually contain 0.5 percent or more PCB congeners with more than 4 chlorines.

2.3.9.2 Potential PCB Non-Cancer Effects

In addition to cancer risk associated with all PCBs, we generally recommend determining noncancer hazard for those Aroclors having *RfD*s. IRIS specifies *RfD*s for Aroclor 1254 and Aroclor 1016 (U.S. EPA 2005i). The *RfD* for Aroclor 1254 ($2x10^{-5}$ milligrams per kilogram-day) will typically be used in most circumstances when conducting a risk assessment. We consider this approach reasonable because approximately 77 percent of Aroclor 1254 is composed of PCB congeners with more than 4 chlorines (Hutzinger et al. 1974). The *RfD* for Aroclor 1016 ($7x10^{-5}$ milligrams per kilogram-day) is scientifically defensible for each homologue group demonstrating that at least 99.5 percent of the mass of the released PCB mixture has more than four chlorine atoms per molecule (U.S. EPA 1996q). We generally consider this approach reasonable because approximately 99 percent of Aroclor 1016 is comprised of PCB congeners with 4 or fewer chlorines (Hutzinger et al. 1974).

We also recommend evaluating th risks to infants from exposure to coplanar PCBs in human breast milk. Please see Section 2.3.10.2 for information on comparing estimated levels of coplanar PCBs (along with dioxins and furans) to background. More information on the breast milk pathway is in Chapter 4, and Tables C-3-1 and C-3-2.

2.3.9.3 Fate & Transport of PCBs

When evaluating coplanar PCB congeners, or PCB congener mixtures of which greater than 0.5 percent contain more than 4 chlorines, we recommend using the fate and transport properties for Aroclor 1254. When assessing risks and hazards from PCB congener mixtures of which less than 0.5 percent contain more than 4 chlorines, we recommend using the fate and transport properties of Aroclor 1016.

2.3.10 Polychlorinated Dibenzo(p)dioxins and Dibenzofurans

As was the case with previous Agency guidance (U.S. EPA 1994i, 1994j, 1994n, 1994r, and 1998c), we recommend including PCDDs and PCDFs in the risk assessment. Information in U.S. EPA (2000b) suggests 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.

PCDDs and PCDFs 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. One mode in which PCDDs and PCDFs form is in dry APCSs, where fly ash catalyzes reactions between halogens and undestroyed organic material from the furnace. 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 such as forest fires, municipal waste and medical incinerators, and hazardous waste combustors. Duarte-Davidson et al. (1997) noted that burning chlorine-containing materials in municipal solid waste is responsible for about two-thirds of the total annual emissions of newly formed TCDDs and TCDFs in the United Kingdom. In the United States, U.S. EPA (1998a) estimated that emissions of dioxin TEQs from municipal solid waste incinerators accounted for 37 percent of all emissions of dioxins into the environment in 1995. The current Agency draft dioxin sources inventory suggests that open barrel burning is the largest current single source of release of these compounds (U.S. EPA 2000).

PCDDs and PCDFs are formed at these combustion sources from the reaction of chlorine-containing chemicals and organic matter. Predicting the formation of PCDDs and PCDFs in a specific situation is difficult because dechlorination, which produces PAHs from PCDDs and PCDFs, occurs under similar conditions. Recent studies (Addink et al. 1996; Environment Canada 1987; Fröese and Hutzinger 1996a, 1996b; Gullett et al.1994; Kilgroe et al. 1991; Luijk et al. 1994; and Shriver 1994) have explored some of these complexities, including (1) the formation of PCDDs and PCDFs from simple organics (such as ethane) and complex organics (such as dibenzofuran), and (2) the catalysis of these reactions by various common metals, such as copper. Wikström et al. (1996) found that the form of chlorine—whether organic, as with chlorinated solvents, or inorganic, as with bleach and salts—has little effect on the quantity of PCDDs and PCDFs formed. However, their study found that the total concentration of chlorine is important. In particular, if the waste being burned exceeds 1 percent chlorine, the PCDD and PCDF formation rate increases significantly. In contrast, Rigo et al. (1995) analyzed over 1,700 test results with varying chlorine feed concentrations and found no statistically significant relationship. The formation rate of PCDDs and PCDFs may also depend on the physical characteristics of the waste feed stream. Solid waste streams or high-ash-content liquid waste feed streams may increase particulate levels in the combustion system between the combustion unit and the APCS. The increased particulate levels provide additional surfaces for catalysis reactions to occur.

A review of currently available dioxin data for combustors reveals that total PCDD/PCDF emission rates vary by more than 28-fold between different facilities, even though they use similar combustion units and APCSs (U.S. EPA 1996b). Site-specific emission data are therefore needed to complete a more refined risk assessment of each combustor.

In evaluating fate-and-transport, it is important to consider the chemical and physical properties of dioxins. In soil, sediment, and the water column, PCDDs and PCDFs are primarily associated with particulate and organic matter because of their high lipophilicity and low water solubility. Ambient air monitoring studies, in which researchers studied the partitioning of dioxin-like compounds between the vapor and particle phases, suggest that the higher chlorinated congeners (the hexa through octa congeners) principally sorb to airborne particulates. The tetra and penta congeners significantly, if not predominantly, partition to the vapor phase (U.S. EPA 2000b). These findings are consistent with vapor/particle partitioning as theoretically modeled in Bidleman (1988). Dioxin-like compounds exhibit little potential for significant leaching or volatilization (U.S. EPA 2000b).

The following subsections clarify the procedures we recommend using (in conjunction with the procedures described in Chapter 7) to estimate risks associated with PCDDs and PCDFs. Also, we’re aware of growing concern regarding the risks resulting from exposure to (1) fluorine- and bromine-substituted dioxins and furans, and (2) sulfur analogs of PCDDs and PCDFs. Research regarding these compounds is ongoing. Until such time as new information is released, though, you can consider the following subsections as our guidance on how to evaluate fluorine, bromine, and sulfur PCDD/PCDF Analogs as potential COPCs in hazardous waste combustor risk assessments.

2.3.10.1 PCDD/PCDF Cancer Risks

We recommend using the TEF method to assess carcinogenic risk on the basis of toxicity relative to 2,3,7,8-TCDD, which is the most toxic dioxin.

There are 210 individual compounds or “congeners” of PCDDs and PCDFs. Seventeen of these 210 congeners are considered to have "dioxin-like" toxicity. In the TEF method, each of these 17 congeners is assigned a value, referred to as a toxicity equivalency factor (TEF), which compares its toxicity to that of 2,3,7,8-TCDD. 2,3,7,8-TCDD then has a TEF of 1.0, and other dioxin-like congeners have TEFs between 0.0 and 1.0. TEF values for these 17 congeners are listed in the Table 2-7.

TABLE 2-7

PCDD/PCDF TOXICITY EQUIVALENCY FACTOR VALUES

Dioxin Congener	TEF (unitless)	Furan Congener	TEF (unitless)
2,3,7,8-Tetrachlorodibenzo(p)dioxin	1.0	2,3,7,8-Tetrachlorodibenzofuran	0.1
1,2,3,7,8-Pentachlorodibenzo(p)dioxin	1.0	1,2,3,7,8-Pentachlorodibenzofuran	0.05
1,2,3,4,7,8-Hexachlorodibenzo(p)dioxin	0.1	2,3,4,7,8-Pentachlorodibenzofuran	0.5
1,2,3,6,7,8-Hexachlorodibenzo(p)dioxin	0.1	1,2,3,4,7,8-Hexachlorodibenzofuran	0.1
1,2,3,7,8,9-Hexachlorodibenzo(p)dioxin	0.1	1,2,3,6,7,8-Hexachlorodibenzofuran	0.1
1,2,3,4,6,7,8-Heptachlorodibenzo(p)dioxin	0.01	1,2,3,7,8,9-Hexachlorodibenzofuran	0.1
1,2,3,4,6,7,8,9-Octachlorodibenzo(p)dioxin	0.0001	2,3,4,6,7,8-Hexachlorodibenzofuran	0.1
		1,2,3,4,6,7,8-Heptachlorodibenzofuran	0.01
		1,2,3,4,7,8,9-Heptachlorodibenzofuran	0.01
		1,2,3,4,6,7,8,9-Octachlorodibenzofuran	0.0001

Source: World Health Organization (1998)
Van den Berg et al (1998).

To estimate the exposure media concentrations for PCDDs and PCDFs, we recommend using the congener-specific emission rates from the stack. Then, model the fate and transport of each of these 17 congeners to the exposure site to estimate congener-specific exposure media concentrations. The HHRAP companion database includes congener-specific fate and transport parameter values, and the media concentration equations are provided in Appendix B. After estimating congener-specific exposure media concentrations, we recommend using the TEFs to estimate a "toxic equivalent" (TEQ) exposure media concentration, and an overall TEQ exposure and cancer risk, as follows:

1. convert the exposure media concentrations of an individual congener to a TEQ concentration for that congener by multiplying the congener’s media concentrations by the congener’s TEF;
2. sum the TEQ concentrations of the individual congeners to get an overall exposure media concentration;
3. estimate the lifetime average daily dose (LADD) for the TEQ concentration; and
4. assess the cancer risk on a TEQ basis using the cancer slope factor for 2,3,7,8-TCDD, in combination with the TEQ-based LADD.

Please see Chapter 7 for a more complete discussion of the steps included in the TEF method.

2.3.10.2 PCDD/PCDF Noncancer Hazards

We generally recommend comparing PCDD and PCDF oral exposure estimates to national average background exposure levels, using 1 pg TEQ/Kg/day for adults and 60 pg TEQ/kg/day for nursing infants. The pertinent exposure estimate would be the ADD, or Average Daily Dose, experienced over the course of the exposure duration, rather than the LADD, which is this ADD averaged over a lifetime.

The Agency typically evaluates noncancer effects of chemicals by comparing exposure levels to health-based reference doses or reference concentrations. However, for reasons discussed in the Agency's Draft Dioxin Reassessment (U.S. EPA 2000b), the Agency has not developed these non-cancer benchmarks for any of the PCDD or PCDF congeners, or for TEQ concentrations/doses.

One approach the Agency has taken to evaluate whether PCDDs and PCDFs emitted from hazardous waste combustion facilities are likely to cause significant noncancer health effects is to compare estimated TEQ exposures to national average background exposure levels (1 pg TEQ/kg/day for adults and 60 pg TEQ/kg/day for nursing infants). The average background level of PCDD/PCDFs and co-planar, dioxin-like PCBs in breast milk is 25 parts per trillion (ppt) of 2,3,7,8-TCDD TEQ (EPA 2000b). The 25- ppt 2,3,7,8-TCDD TEQ is the sum of the average breast milk concentration of 18-ppt TEQ from PCDD/PCDFs and 7-ppt TEQ from co-planar, dioxin-like PCBs (EPA 200b). After normalization for infant body weight, this breast milk concentration of 25 ppt TEQ results in an average, background intake for the infant, ADIb-inf, of 93 picograms per kilogram per day (pg/kg-day) of 2,3,7,8-TCDD TEQ. If exposures due to the facility's emissions during the exposure duration of concern are low compared to background exposures, then the emissions aren't expected to cause an increase in noncancer effects.

In the future, the Agency may develop alternative approaches to evaluate noncancer effects from exposures to PCDDs and PCDFs. In that case, those approaches may be included in future risk assessments.

2.3.10.3 Fluorine, Bromine, and Sulfur PCDD/PCDF Analogs

We generally recommend deciding on a site-specific basis whether to evaluate these compounds, in consultation with the permitting authority. Considering that neither the likelihood of the formation, nor the toxicity of these compounds is well understood, the permitting authority is not likely to request a quantitative toxicity assessment of fluorine, bromine, and sulfur analogs.

The Agency is currently evaluating the potential for the formation of (1) fluorine- and bromine-substituted dioxins and furans, and (2) sulfur analogs of PCDDs and PCDFs (U.S. EPA 1996h; 1996m). Available information indicates that fluorinated dioxins and furans aren't likely to be formed as PICs; although the presence of free fluorine in the combustion gases may increase the formation of chlorinated dioxins (U.S. EPA 1996h). We aren't aware of any studies conducted to evaluate this relationship.

Available information indicates that there is potential for brominated or chlorobrominated dioxins to form (U.S. EPA 1996i). The Agency has not assigned TEF values for brominated dioxins or furans (U.S. EPA 1994k). However, the toxicity of bromo- and chlorobromo-substituted dioxin analogs is comparable to that of chlorinated dioxins in short-term toxicity assays (U.S. EPA 1996m).

Although chlorinated dibenzothiophenes (the sulfur analogs of dibenzofurans) have been reported to form, no information is available indicating the formation of chlorinated dioxin thioethers (the sulfur analogs of dibenzo[p]dioxins) (U.S. EPA 1996h). This may be because the carbon-oxygen bond is stronger than the carbon-sulfur bond, and the compound furan (which is part of the dibenzofuran structure) is more stable than thiophene (which is part of the dibenzothiophene structure) (U.S. EPA 1996c). Another possible reason that chlorinated dioxin thioethers have not been observed is the potential instability of these compounds. Chlorinated dioxin thioethers contain two carbon-sulfur bonds in the central ring of the structure (U.S. EPA 1996h).

> ***Please Note***: There is currently no U.S. EPA-approved method for the sampling or analysis of these dioxin analogs.

We generally recommend using the TOE method (see Section 2.2.1.2) to account for the potential presence of these compounds. The Uncertainty section of the risk assessment report (See Chapter 8) could then discuss the potential for the formation of these analogs.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

C Description of any combustor-specific operating conditions that may contribute to the formation of dioxins

C Any facility-specific sampling information regarding PCDD and PCDF concentrations in air, soil, water, or biota

C Information regarding the concentrations of sulfur, fluorine, and bromine in the combustor feed materials

2.3.11 Polynuclear Aromatic Hydrocarbons (PAHs)

As is the case in previous Agency guidance (U.S. EPA 1994i, 1994j, 1994r, 1998c), we recommend evaluating PAHs as COPCs. The following are commonly detected PAHs:

- benzo(a)pyrene (BaP);
- benzo(a)anthracene;
- benzo(b)fluoranthene;
- benzo(k)fluoranthene;
- chrysene;
- dibenz(a,h)anthracene; and
- indeno(1,2,3-cd)pyrene.

The Agency considers all of these compounds to be carcinogenic. However, an IRIS oral cancer slope factor is only available for benzo(a)pyrene.

PAHs are readily formed in combustors by either (1) dechlorination of other PAHs (such as dioxins) present in the waste feed or emissions stream, or (2) the reaction of simple aromatic compounds (benzene or toluene) present in the waste feed or emissions stream. PAHs are well-known as the principal organic components of emissions from all combustion sources, including coal fires (soot), wood fires, tobacco smoke ("tar"), diesel exhaust, and refuse burning (Sandmeyer 1981). They are generally the only chemicals of concern in particulate matter (Manahan 1991), although the presence of metals and other inorganics in the waste feed can add other contaminants of concern. Based on the toxicity and combustion chemistry of PAHs, we generally recommend that stack gas testing confirm the absence of these compounds from stack emissions.

At present, BaP is the most studied PAH and the only one for which a 2-year feeding rodent bioassay has been conducted (U.S. EPA 1991c). The studies available for the other carcinogenic PAHs were conducted by injection, dermal or gavage. Multiple animal studies in rodent and nonrodent species demonstrated BaP to be carcinogenic following administration by oral, intratracheal, inhalation, and dermal routes. BaP also produced positive results in several *in vitro* bacterial and mammalian genetic toxicity assays, in addition to numerous *in vivo* tests for deoxyribonucleic acid (DNA) damage. BaP metabolizes to reactive electrophiles that are capable of binding to DNA (U.S. EPA 1990h). Therefore, U.S. EPA (1993d) used various nonbioassay results to determine relative potency factors (RPFs) for the class B2 carcinogen PAHs. RPFs for these seven PAHs are listed in Table 2-8.

TABLE 2-8

RELATIVE POTENCY FACTORS FOR CLASS B2 CARCINOGEN PAHs

Compound	RPF
Benzo(a)pyrene	1.0
Benz(a)anthracene	0.1
Benzo(b)fluoranthene	0.1
Benzo(k)fluoranthene	0.01
Chrysene	0.001
Dibenz(a,h)anthracene	1.0
Indeno(1,2,3-cd)pyrene	0.1
Source: U.S. EPA (1993d)	

Obtaining test data for an individual chemical from a standard carcinogenesis bioassay that might be used to develop a cancer slope factor requires:

1. at least 1 kilogram of relatively pure chemical (greater than 98 percent purity is the most common specification),
2. $500,000 to $1,000,000, and
3. 5 to 6 years.

However, an alternative to the full carcinogenesis bioassay is to use *in vitro* studies to compare various PAHs. *In vitro* studies, such as those conducted by Knebel et al. (1994) and many other groups, require a few milligrams of each chemical, a few weeks, and about $1,000 per chemical. Because of these differences, we wouldn't anticipate that many full carcinogenesis bioassays of PAHs will be carried out in the near future.

As with previous guidance (U.S. EPA 1994g), we generally recommend evaluating PAHs using an approach similar to the BaP-RPF method. We generally recommend using the fate-and-transport properties of specific PAHs (versus those of benzo[a]pyrene) to estimate exposure concentrations. Then following the BaP-RPF method, you adjust the concentrations of the individual PAHs and sum them to obtain an equivalent total concentration of BaP. Multiply this summed concentration by the BaP cancer *SF* to estimate total risk from all carcinogenic PAHs.

We don't currently recommend a metabolism factor (MF) for PAHs. A published study (Hofelt et al. 2001), however, highlights the uncertainty in the HHRAP's approach and presents an alternative metabolism factor for use. If this alternative metabolism factor is used, you may wish to consider the following site-specific points:

1. If the PAHs under consideration are metabolized in the animal (beef, pork, etc) has it been determined that the degradation products/metabolites aren't persistent in the meat/and or milk? (This concern has been raised because it is the degradation products of PAHs that cause the toxicity.);

2. Is the extrapolation from the rat to larger animal appropriate?; and

3. If the metabolism factor is appropriate, should it be used equally for all the PAHs being evaluated?

Using an MF is discussed further in Section 5.4.4.7 and Appendix B, Table B-3-10.

In addition to carcinogenic effects, noncarcinogenic health effects are associated with exposure to PAHs. However, RPFs for noncarcinogenic effects of PAHs (similar to those developed for carcinogenic effects) have not been developed. The uncertainties associated with attempting to quantify the potential noncarcinogenic effects of PAHs without RfDs or RfCs is typically considered greater than the uncertainty associated with not evaluating these potential effects. However, if site-specific emissions data indicate that significant amounts of noncarcinogenic PAHs are emitted, we generally recommend that the potential to underestimate the noncarcinogenic health effects associated with exposure to PAHs be discussed in the Uncertainty section of the risk assessment report.

2.3.12 Radionuclides

Radionuclides exist in naturally occurring materials such as coal and other rocks, and as radioactive by-products of industrial processes. The HHRAP doesn’t consider the naturally occurring concentrations of radioactive materials such as uranium and thorium (and their decay elements), based on Agency policy and technical limitations for measuring such low levels. However, radioactive wastes and materials, as defined by the U.S. Nuclear Regulatory Commission (NRC) and the U.S. Department of Energy (DOE), are subject to evaluation through interagency agreements on this subject. The U.S. NRC considers “radioactive waste” as waste that is, or contains, by-product material, source material, or special nuclear material (as defined in 10 CFR Part 20.1003). The U.S. NRC considers “mixed waste” as waste that contains both radioactive waste and hazardous waste (as defined by U.S. EPA). Both radioactive and mixed wastes must be handled in accordance with all relevant regulations, including U.S. EPA and U.S. NRC (10 CFR Part 20.2007) regulations. In particular, U.S. NRC licensees must comply with 10 CFR Part 20.2004—“Treatment or Disposal by Incineration”—and applicable U.S. EPA regulations.

We generally recommend evaluating the burning of mixed waste and radioactive material if those substances are components of the combustor feed. We also generally recommend including a radionuclide as a COPC if it is in the combustor’s feed, and has an available toxicity value (e.g., slope factor). Slope factors for over 300 radionuclides are available in HEAST. The slope factor for a particular radionuclide is multiplied by the intake (pCi) or soil concentration and years of exposure (pCi/g times years of exposure) to estimate cancer risk.

Radionuclide exposure pathways typically evaluated in human health risk assessments include inhalation, ingestion of food products (e.g., meat, milk, vegetables), incidental soil ingestion, external exposure from ground surface deposits, and external exposure from air concentrations (air submersion). The submersion exposure pathway may be of particular concern for radionuclides that emit high-energy beta particles.

Environmental transport and subsequent human exposure are commonly evaluated through the use of radionuclide “dose” codes or computer code/spreadsheet combinations. A dose code combines air dispersion/deposition modeling with terrestrial transport models, human exposure parameters, and pre-calculated dose conversion factors [e.g., from *Federal Guidance Report No. 11* (U.S. EPA 1988c) and *Federal Guidance Report No. 12* (U.S. EPA 1993j)] to obtain dose and/or risk. The following are several available dose codes for evaluating radionuclides from mixed waste combustion facilities:

CAP-88	(Clean Air Act Assessment Package - 1988)
GENII	(The Hanford Environmental Radiation Dosimetry Software System)
MEPAS	(Multimedia Environmental Pollutant Assessment System)
ISCST3	(Industrial Source Complex Dispersion Model)
CALPUFF	(California Puff Model)

To calculate air concentrations and ground deposition rates of radionuclides, we generally recommend the ISCST3 air dispersion model using the exponential decay option. Intake can then be calculated with appropriate exposure scenario equations and parameters. ISCST3 is a good choice for facilities with multiple sources, complex terrain, building downwash, and wet/dry deposition requirements. A second option is to use the air concentration and ground deposition rate output from another dose code (e.g., CAP-88 if the facility has completed its NESHAPs analysis).

We generally recommend that equations for fate and transport of radionuclides in soil and water be consistent with those presented for non-radionuclides, while factoring in decay (and ingrowth if applicable). The recommended food chain biotransfer parameters used to determine food concentrations are available in the user’s guides for most of the dose codes listed previously. However, a comprehensive reference for obtaining these values is the *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments; IAEA Technical Report Series No. 364* (International Atomic Energy Agency 1994).

Decay and ingrowth of radionuclides is a special consideration for integrating radioactive materials into risk calculations. Most radioactive materials undergo radioactive decay through a series of transformations rather than in a single step. Until the last step, these radionuclides emit energy or particles with each transformation and become other radionuclides. As radioactive decay progresses, the concentration of the original radionuclides decreases, while the concentration of their decay products increases and then decreases as the decay products themselves transform. The increasing concentration of decay products and activity is called ingrowth. We recommend that the assessment consider decay over both the air transport time and the surface exposure duration. Ingrowth may be important, and we generally recommend the assessment use radionuclide slope factors that include contributions from decay elements (“+D” slope factors). Ingrowth that involves change of physical state is another situation needing special attention in the fate and transport modeling. For instance, solid radium-226 decays to gaseous radon-222, which then decays through solid polonium-218 to further decay elements.

Options for estimating cancer risks from exposure to radionuclides include using the slope factor methodology presented in the Agency's *Estimating Radiogenic Cancer Risks* (U.S. EPA 1994s), or using estimates of the relationship between radiation dose and cancer risk to convert dose to risk. Federal Guidance Report No. 13 (U.S. EPA 1999d) contains recent estimates of cancer risk for given radiation dose, based on low-dose, low-Linear Energy Transfer uniform irradiation of the body. For these conditions, radiation dose equivalent (rem) and absorbed dose (rad) are approximately equivalent. The Report provides risk estimates in terms of mortality and morbidity. It is important to use the estimate appropriate to the site and assessment in question.

The dose-conversion approach uses a single factor to convert dose to risk. It is limited, then, in that it doesn't take into account variations among radionuclides in the relationship between dose and risk. In general, though, this approach is protective. The slope factor approach generally provides a better estimate of risk. Limitations of the slope factor conversion methodology, however, include:

- It assumes a single chemical form, which is not necessarily site-specific or most protective;
- External radiation slope factors are only provided for soil contaminated to an infinite thickness, which will over-estimate exposure from radionuclide concentrations near the surface;
- Slope factors aren't available for the submersion in water exposure pathway; and
- Slope factors include decay chains for a limited number (18) of parent radionuclides (although these are the most significant decay chains)

Some radioactive materials, such as uranium, also present a non-carcinogenic hazard that it is possible to evaluate. We generally recommend also assessing these non-carcinogenic hazards. We also recommend that the risk summary table in the risk assessment report present the cancer risk from radiological contaminants alongside the risks from non-radiological contaminants.

To enhance transparency, we generally recommend that results include a discussion of additivity and the uncertainties of additivity when combining risks from radiological and non-radiological contaminants. There are fundamental differences between the slope factors for chemical and radionuclide carcinogens.

Most notably, the slope factors for chemical carcinogens generally represent an upper bound or 95th percent confidence limit value, while radionuclide slope factors are best estimate (50% confidence) values.

> ***Please Note***: A prescriptive method for calculating risk from combustion facilities burning mixed waste is beyond the scope of the HHRAP. The above information is provided to outline the method we recommend.

2.3.13 Volatile Organic Compounds

U.S. EPA (1990e) reported that volatile organics listed as probable PICs (based on Freeman 1988 and 1989) produced by burning hazardous waste include:

- benzene;
- chloroform;
- tetrachloroethylene;
- 1,1,1-trichloroethane;
- toluene; and
- methylene chloride.

However, the validity of evaluating volatile organic COPCs through the various indirect exposure pathways (see Chapter 4) is subject to debate. One argument to exclude these COPCs from evaluation is that there is no empirical evidence that VOC emissions pose a hazard via indirect pathways. We're similarly not aware of any such evidence, but we're also unaware of any evidence to the contrary.

Another argument to exclude VOCs from evaluation is based on the conclusion that

1. volatile organic COPCs released into the air are expected to remain in the gas phase unless or until they are transformed into low-volatility compounds, and

2. this transformation (or atmospheric chemical reaction), and the subsequent removal of the reaction products, makes irrelevant the toxicity of the parent volatile organic COPC.

We disagree with both aspects of this argument. First, we're not aware of any information or research documenting the fate-and-transport of volatile organic COPCs from hazardous waste combustors. Second, although we agree that the toxicity of the parent COPC is irrelevant following transformation, this argument ignores the potential toxicity of the reaction products. We're not aware of any available quantitation methods that could be used to predict atmospheric chemical reactions of this nature. We therefore generally believe that evaluating the fate-and-transport (and toxicity) of the parent COPC remains the best available method for protectively accounting for the potential reaction products to which receptors are ultimately exposed.

Finally, another argument to exclude VOCs from evaluation is the assertion that there is no firm technical basis for assessing the rate of deposition of VOCs to soils or uptake by plants (discussed in detail in Appendix A-2). Although we agree with the basic premise of this issue, we're unaware of any other method for evaluating the potential indirect exposure to volatile organic COPCs or their atmospheric reaction products (empirical data aren't available).

To summarize, we agree in principle that the science regarding the fate-and-transport of volatile organic COPCs in the environment is poorly understood. However, because the potential risks associated with indirect exposure to these COPCs is also poorly understood, we believe that evaluating volatile organic COPCs via the indirect exposure pathways—with the proper explanation of the uncertainties associated with this process—generally provides the most reasonable (based on current science) and protective estimate of these potential risks. We also believe that the risk equations generally address this issue because a calculation cannot be completed unless there are sufficient fate and transport properties values for each COPC.

Finally, current sampling and analytical methods aren't always able to positively identify all individual organic compound in stack emissions. We recommend accounting for the mass of unidentified organic compounds in stack emissions on the basis of TOE from the hazardous waste combustor. The methodology for using TOE in a risk assessment is discussed further in Section 2.2.1.2.

2.4 ESTIMATING COPC CONCENTRATIONS FOR NON-DETECTS

One particularly difficult issue in a risk assessment is how to treat data that are reported as below the “detection limit” (i.e. “Non-detects”). The following subsections:

1. define commonly reported detection limits;
2. describe the use of non-detect data in the risk assessment;
3. describe statistical distribution techniques applied to address this issue;
4. summarize our recommendations on quantifying non-detects for use in risk assessments; and
5. clarify use of data flagged as estimated maximum possible concentration (EMPC) in the risk assessment.

2.4.1 Definitions of Commonly Reported Detection Limits

Generically, a “detection limit” is the lowest level of an analyte that can be detected using a particular analytical method. The Agency’s commonly-used definition for the detection limit for non-isotope dilution methods is the method detection limit (MDL), as promulgated in 40 CFR Part 136, Appendix B (U.S. EPA 1995i). A level above the MDL is the level at which reliable quantitative measurements can be made; generically called the “quantitation limit” or “quantitation level.” In practice, numerous terms have been created to describe detection and quantitation levels. We have summarized below the significance and applicability of the levels most widely reported by analytical laboratories. These levels—listed generally from the lowest limit to the highest limit—include the following:

- <u>Instrument Detection Limit (IDL)</u>: The smallest signal above background that an instrument can reliably detect, but not quantify. Also commonly described as a function of the signal-to-noise (S/N) ratio.

- <u>Method Detection Limit (MDL)</u>: The minimum concentration of a substance that can be measured (via non-isotope dilution methods) and reported with 99 percent confidence that the analyte concentration is greater than zero. It is determined from analysis of a sample in a specific matrix type containing the analyte. The MDL is considered the lowest level at which a compound can be reliably detected. The MDL is based on statistical analyses of laboratory data. In practice, MDLs are determined on analytical reagents (e.g., water) and not on the matrix of concern. However, a laboratory may contract to do a matrix-specific “MDL Study” for a particular project or a particular facility’s waste matrix when needed. However, routine MDL determinations (water reagent) are conducted on at least an annual basis or whenever equipment changes occur. MDLs for a given method are laboratory- and compound-specific.

To determine the MDL as specified in 40 CFR Part 136, Appendix A, for example, at least seven replicate samples with a concentration of the compound of interest near the estimated MDL are analyzed. The standard deviation among these analyses is calculated and multiplied by 3.14. The result of the calculation becomes the MDL. The factor of 3.14 is based on a t-test with six degrees of freedom and provides a 99 percent confidence that the analyte can be detected at this concentration (U.S. EPA 1995i).

Please Note: 40 CFR Part 136 is specific to the Clean Water Act, and it identifies the use of water as the matrix for the MDL determination. The MDL was promulgated in 1984, and is incorporated in more than 130 Agency analytical methods for determining several hundred analytes.

C Reliable Detection Level (RDL): A detection level recommended by the National Environmental Research Laboratory in Cincinnati. It is defined as 2.623 times the MDL (U.S. EPA 1995i).

C Estimated Detection Limit (EDL): A quantitation level defined in SW-846 that has been applied to isotope dilution test methods (e.g., SW-846 Method 8290). A variation of the SW-846 defined EDL is also commonly reported by commercial laboratories, however, with the addition of a multiplication factor that generally elevates the EDL value by 3.5 to 5 times that of the SW-846 definition. Commercial laboratories sometimes report EDLs for non-isotope dilution methods, even though an EDL is not defined by the methods.

As defined in SW-846: The EDL is defined in SW-846 (presented in various methods, e.g., Method 8280A) as the estimate made by the laboratory of the concentration of a given analyte required to produce a signal with a peak height of at least 2.5 times the background signal level. The estimate is specific to a particular analysis of the sample and will be affected by sample size, dilution, etc. The presented equation defining EDL is as follows:

$$EDL = \frac{2.5 \cdot Q_{is} \cdot (H_n^1 + H_n^2) \cdot D}{V \cdot (H_{is}^1 + H_{is}^2) \cdot RF_n} \qquad \text{Equation 2-3}$$

where

EDL = Estimated detection limit (ng/L)

2.5 = Peak height multiplier (unitless)

Qis = Nanograms of the appropriate internal standard added to the sample prior to extraction (ng)

H_n^1 and H_n^2 = The peak heights of the noise for both of the quantitation ions of the isomer of interest

H_{is}^1 and H_{is}^2 = The peak heights of both the quantitation ions of the appropriate internal standards

D = Dilution factor - the total volume of the sample aliquot in clean solvent divided by the volume of the sample aliquot that was diluted (unitless)

V = Volume of sample extracted (L)

RF_n = Calculated relative response factor from calibration verification (unitless)

Common commercial laboratory practice: The EDL generally reported by commercial laboratories is defined as the detection limit reported for a target analyte that is not detected or presents an analyte response that is less than 2.5 times the background level. The area of the compound is evaluated against the noise level measured in a region of the chromatogram clear of genuine GC signals times an empirically derived factor. This empirical factor approximates the area to height ratio for a GC signal. This factor is variable between laboratories and analyses performed, and commonly ranges from 3.5 to 5. The equation is as follows:

$$EDL = \frac{2.5 \cdot Q_{\beta} \cdot (F \cdot H) \cdot D}{W \cdot A_{\beta} \cdot RRF_{\sigma}} \qquad \text{Equation 2-4}$$

where

EDL = Estimated detection limit
2.5 = Minimum response required for a GC signal
$Q_{\$}$ = The amount of internal standard added to the sample before extraction
F = An empirical factor that approximates the area to height ratio for a GC signal
H = The height of the noise
D = Dilution factor
W = The sample weight or volume
RRF_{F} = The mean analyte relative response factor from the initial calibration
$A_{\$}$ = The integrated current of the characteristic ions of the corresponding internal standard

C Practical Quantitation Limit (PQL): A quantitation level that is defined in 50 FR 46908 and 52 FR 25699 as the lowest level that can be reliably achieved with specified limits of precision and accuracy during routine laboratory operating conditions (U.S. EPA 1992g; 1995i). The PQL is constructed by multiplying the MDL, as derived above, by a factor (subjective and variable between laboratories and analyses performed) usually in the range of 5 to 10. However, PQLs with multipliers as high as 50 have been reported (U.S. EPA 1995i).

The PQL has been criticized because of the ambiguous nature of the multiplier and because the resulting levels have been perceived as too high for regulatory compliance purposes (U.S. EPA 1995i).

C Target Detection Limit (TDL): A quantitation level constructed similar to the PQL.

C Reporting Limit (RL): A quantitation level constructed similar to the PQL.

C Estimated Quantitation Limit (EQL): A quantitation level constructed similar to the PQL.

C Sample Quantitation Limit (SQL): A quantitation level that is sample-specific and highly matrix-dependent because it accounts for sample volume or weight, aliquot size, moisture content, and dilution. SQLs for the same compound generally vary between samples as moisture content, analyte concentration, and concentrations of interfering compounds vary. The SQL is generally 5 to 10 times the MDL, however, it is often reported at much higher levels due to matrix interferences.

- Contract Required Quantitation Limit (CRQL)/Contract Required Detection Limit (CRDL): A quantitation pre-set by contract, which may incorporate SW-846 methods, Office of Water methods, or other methods deemed necessary to meet study objectives. These limits are typically administrative limits and may actually be one or two orders of magnitude above the MDL.

2.4.2 Using Non-Detect Data In the Risk Assessment

In collecting waste feed or emissions data for use in risk assessments, or in setting regulatory compliance levels, a permitting authority is often faced with data quality objectives that require analyses near or below analytical detection or quantitation levels. In such situations, permittees often argue that the detection levels should be set with a large factor of certainty in order to be confident that measurements are reliable. Environmental groups frequently argue that a level of zero, or a level at which a single researcher can demonstrate that the compound can be detected, should be used as the set level. Measurements made below analytical detection and quantitation levels are associated with increased measurement uncertainty, so it is important to understand the impact they may have when they are applied.

Because of the quantitative differences between the various types of detection levels, “non-detected” compounds pose two questions:

1. Is the compound really present?, and
2. If so, at what concentration?

The first question is generally hard to answer, and is dependent mainly on the analytical resources available. The answer to the second question is “somewhere between true zero and the quantitation level applied.” In earlier guidance (U.S. EPA 1994i) we recommended applying emission rates of one-half the “detection limit” for non-detects. However, which detection limit to use was not explicitly defined or presented in quantitative terms.

For *waste feed* data (e.g., used as a surrogate when emissions data is not available), we generally recommend using the SQL, since waste feeds are typically highly concentrated organic or inorganic matrices that require special analytical clean-up procedures and dilutions of the sample. This approach is consistent with that used in other program offices for highly contaminated media requiring quantification specifically for risk assessment purposes. However, in dealing with stack emissions data, concentrations of constituents are typically found only at trace levels (assuming good combustion). Therefore, to increase consistency and reproducibility in dealing with non-detects for *emissions data*, we generally recommend using the MDL-derived RDL to quantify non-detects for COPCs analyzed with non-isotope dilution methods, and using the method-defined EDL to quantify non-detects for COPCs analyzed with isotope dilution methods. The procedures are as follows:

For COPCs Analyzed With Non-isotope Dilution Methods: Quantify non-detects by using an MDL-derived RDL.

1. Require the laboratory to report the actual MDL as specified in the chosen SW-846 analytical method. The laboratory should report MDLs for every non-detect compound analyzed, in addition to the commonly used reporting limit, such as an EDL, EQL, or PQL.

   Commonly used SW-846 non-isotope dilution methods such as Method 8260 (volatiles), and Method 8270 (semivolatiles) don’t themselves define the MDL. They reference 40 CFR Part 136 instead. Though specified in the Method, some laboratories do not always report MDLs as defined in 40 CFR Part 136.

   This would apply to the analysis of each individual component of multiple-component samples (e.g., front half rinse, XAD resin, condensate, Tenax tube).

   Note: Laboratories typically produce MDLs specific to each non-isotope dilution method performed by the laboratory on an annual basis.

2. Calculate an MDL-derived RDL for each COPC non-detect by multiplying the MDL by 2.623 (interim factor) (U.S. EPA 1995i).

   Another option is to request that the laboratory derive the RDLs for you (per the definition above), as part of the analysis. Good quality assurance/quality control (QA/QC) suggests you check to make sure the RDLs have been generated properly.

3. Adjust the RDL, as appropriate, to account for sample-specific volumetric treatments (e.g., splits and dilutions) that differ from those used in the Part 136 MDL determinations.

   Again, an option is to have the laboratory perform the adjustments for you. We recommend you check to make sure the adjustments have been done properly.

For COPCs Analyzed With Isotope Dilution Methods: Quantify non-detects using the EDL as defined by the analytical method, without the use of empirical factors or other mathematical manipulations specific to the laboratory. Commonly used isotope dilution methods include SW-846 Methods 8290, 1624, and 1625, as well as CARB 429.

Methods for Metals Analysis: Quantify non-detects for metal analysis in the risk assessment by using the IDL as defined by the analytical method, without the use of empirical factors or other mathematical manipulations specific to the laboratory.

Please note this recommendation is an update to the information provided in U.S. EPA [2001c] *Risk Burn Guidance for Hazardous Waste Combustion Facilities.*)

The MDL definition used in 40 CFR Part 136 addresses errors of the first type (i.e. false negatives). The 99 percent confidence limit states that the MDL has only a 1 percent chance the detects will be misidentified as negative, when the compound of concern is actually present. Errors of the second type (false positives) aren't addressed. By not addressing false positives, the statistically-defined default value becomes 50 percent. In other words, where 40 CFR did not address false positives, the system required that 50 percent of the detects at the MDL would be false positives. This is a very protective approach, and biased toward not missing any compounds of potential concern that may be present. The MDL-derived RDL, and to a lesser extent the EDL, somewhat indirectly addresses the false positive issue. As described in defining the RDL (see Section 2.4.1), by the time the standard deviation has been multiplied by 8, the possibility of false positives is usually less than 1 percent.

2.4.3 Statistical Distribution Techniques

Many statistical distribution techniques are available for calculating a range of standard deviations to quantify non-detect concentrations of COPCs. These include random replacement scenarios, such as (Cohen and Ryan 1989; Rao, Ku, and Rao 1991):

- the uniform fill-in (UFI) method, in which each LOD value is replaced with a randomly generated data point by using a uniform distribution;
- the log fill-in (LFI) method which is the same as UFI, except using a logarithmic distribution;
- the normal fill-in (NFI) method which is the same as UFI, except using a log-normal distribution; and
- the maximum likelihood estimation (MLE) technique.

Also, if the permitting authority determines it to be applicable, a Monte Carlo simulation might be used to determine a "statistical" value for each non-detect concentration.

2.4.4 Our Recommendations on Quantifying Non-Detects

Using non-detects in a risk assessment depends on the analytical method(s) used to produce the data. In most cases, the Agency estimates emission rates for undetected COPCs (see Section 2.3) by assuming that COPCs are present at a concentration equivalent to the MDL-derived RDL for non-isotope dilution methods, or to the method-defined EDL for isotope dilution methods. We consider these methods reasonable, and believe they represent a scientifically sound approach that supports maximum protection of human health and the environment while recognizing the uncertainty associated with analytical measurements at very low concentrations in a real world sample matrix. We also recognize that there are subjective components and limitations to each of the non-detect methodologies presented in this and previous guidance, including the recommended methods.

Some risk assessors have expressed the desire to obtain and use non-routine data (e.g., uncensored data) of defensible quality in risk assessments, as a way to deal with non-detect issues. The HHRAP doesn't address what forms or how such data might be used. The decision to use non-routine data in a risk assessment is not precluded just because it is different. Neither does the HHRAP necessarily endorse using non-routine data. We generally recommend consulting with the permitting authority on the appropriateness of using non-routine data. If non-routine data is used, we generally recommend carefully identifying and evaluating the limitations associated with the data, and clearly document this discussion in the Uncertainty section of the risk assessment report.

As stated previously, a pretrial burn risk assessment can help to make sure that the trial burn test will achieve the desired quantitation limit (and, therefore, DREs and COPC stack gas emission rates).

2.4.5 Estimated Maximum Possible Concentration (EMPC)

The EMPC as defined in SW-846 is in most cases only used with the isotope dilution methods. An EMPC is calculated ***for dioxin isomers*** that:

- have a response with a signal to noise ratio of at least 2.5 for both the quantitation ions; and
- meet all the relevant identification criteria specified in the analytical method, except the ion abundance ratio.

Ion abundance ratios are affected by co-eluting interferences that contribute to the quantitative ion signals. As a result, one or both of the quantitative ions signals may possess positive biases.

An EMPC is a worst-case estimate of the concentration. An EMPC is not a detection limit and should not be treated as a detection limit in the risk assessment. We generally recommend using EMPC values as detections without any further manipulation (e.g., dividing by 2). However, because EMPCs are worst-case estimates, you may wish to consult with the permitting authority on techniques to minimize EMPCs when reporting trial and risk burn results. This is especially true when the EMPC values result in risk estimates above regulatory levels of concern. Some techniques to minimize EMPCs include performing additional cleanup procedures (as defined by the analytical method) on the sample or archived extract, and/or reanalyzing the sample under different chromatographic conditions.

> ***Please Note***: using alternative quantitation ions might be acceptable, if the signal-to-noise ratio of the ion signal is at least 2.5 and if the tune data indicate that the mass spectrometer is operating within specifications.

Such actions to reduce the EMPC are expected to be more cost effective than the additional sample cleanup and/or reanalysis.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Actual MDLs for all non-detect stack emissions data, non-isotope dilution methods
- EDLs for all non-detect stack emissions data, isotope dilution methods
- SQLs for all non-detect waste feed or feedstream data used
- Description of the method applied to quantify the concentration of non-detects

2.5 EVALUATING CONTAMINATION IN BLANKS

Blank samples are intended to provide a measure of any contamination that may have been introduced into a sample:

1. in the field while the samples were being collected,
2. in transport to the laboratory, or
3. in the laboratory during sample preparation or analysis.

Blank samples are analyzed the same way as the site samples from the trail burn. To prevent including non-site related compounds in the risk assessment, we generally recommend comparing the concentrations of compounds detected in blanks to concentrations detected in site samples collected during the trial burn. Four types of blanks are defined in the *Risk Assessment Guidance for Superfund*

(U.S. EPA 1989e): trip blanks, field blanks, laboratory calibration blanks, and laboratory reagent or method blanks. Detailed definitions of each are provided below.

> ***Trip Blank*** **-** A trip blank is used to indicate potential contamination due to migration of volatile organic compounds from the air on the site or in sample shipping containers, through the septum or around the lid of sampling vials, and into the sample. The blank accompanies the empty sample bottles to the field as well as with the site samples returning to the laboratory for analysis. The blank sample is not opened until it is analyzed in the lab with the site samples, thus making the laboratory “blind” to the identity of the blanks.
>
> ***Field Blank*** **-** A field blank is used to determine if field sampling or cleaning procedures (e.g., insufficient cleaning of sample equipment) result in cross-contamination of site samples. Like the trip blank, the field blank is transported to the field with empty sample bottles and is analyzed in the laboratory along with the site samples. Unlike the trip blank, however, the field blank sample is opened in the field and recovered in the same manner as the collected samples. As with trip blanks, the field blanks’ containers and labels may be the same as for site samples and blind to the laboratory.
>
> ***Instrument Blank*** **-** An instrument blank is distilled, deionized water injected directly into an instrument without having been treated with reagents appropriate to the analytical method used to analyze actual site samples. This type of blank is used to indicate contamination in the instrument itself.
>
> ***Laboratory Reagent or Method Blank*** **-** A laboratory reagent of method blank results from the treatment of distilled, deionized water with all of the reagents and manipulations (e.g., digestions or extractions) to which site samples will be subjected. Positive results in the reagent blank may indicate either contamination of the chemical reagents or the glassware and implements used to store or prepare the sample and resulting solutions. Although a laboratory following good laboratory practices will have its analytical processed under control, in some instances method blank contaminants cannot be entirely eliminated.
>
> ***Water Used for Blanks*** **-** For all the blanks described above, results are reliable only if the water comprising the blank was clean. For example, if the laboratory water comprising the trip blank was contaminated with VOCs prior to being taken to the field, then the source of VOC contamination in the trip blank cannot be isolated.

Blank data is generally compared to the results with which the blanks are associated. However, if the association between blanks and data can’t be made, blank data is compared to the results from the entire sample data set.

U.S. EPA (1989e) makes a distinction between blanks containing common laboratory contaminants and blanks containing contaminants not commonly used in laboratories. Compounds considered to be common laboratory contaminants are

- acetone,
- 2-butanone (methyl ethyl ketone),
- methylene chloride,
- toluene, and
- the phthalate esters.

If compounds considered to be common laboratory contaminants are detected in the blanks, then sample results are usually not considered to be detected unless the concentrations in the sample are equal to or exceed ten times the maximum amount detected in the applicable blanks. If the concentration of a common laboratory contaminant in a sample is less than ten times the blank concentration, then the compound is usually treated as a non-detect in that particular sample.

In some limited cases, it may be appropriate to consider blanks which contain compounds that aren't considered by the Agency to be common laboratory contaminants. In these limited cases, sample results aren't considered to be detected unless the concentrations in the sample exceed five times the maximum amount detected in the applicable blanks. If the concentration in a sample is less than five times the blank concentration, then the compound would be treated as a non-detect in that particular sample.

Carefully consider the evaluation of blank data in the overall context of the risk assessment and permitting process. We generally expect issues related to non-laboratory contaminant blanks to be minimal, because data collection and analysis efforts in support of trial/risk burns are expected to be of high quality and in strict conformance to QA/QC plans and SOPs. Carefully evaluating the trial/risk burn data will avoid the potential for contaminated blanks to compromise the integrity of the data. It will also help prevent the need for retesting to properly address data quality issues.

We highly recommend practicing caution in applying blank results to correct or qualify sample results for any purpose, as blanks are provided in minimal quantities (e.g., one per test condition or one per test) and therefore are at best qualitative indicators of the validity of a data set. Blank correction can reduce accuracy and often represent a non-conservative uncertainty. Consequently a permit authority might dedice not to allow blank correction as a conservative assumption (consistent with a screening level risk assessment).

When considering blank contamination in the COPC selection process, we recommend that permitting authorities ensure that:

(1) The facility or data gatherer has made every reasonable attempt to ensure good data quality and has rigorously implemented the QA/QC Plan and good industry sampling and testing practices.

(2) Trial/risk burn data has not been submitted to the permitting authority as "blank corrected." Rather, the permitting authority has the full opportunity to review the data absent additional manipulation by the data gatherer.

(3) The effect of the blank correction on the overall risk estimates, if such an effect is considered, is clearly described in the uncertainty section of the risk assessment report.

(4) The risk assessment report lists emissions rates both as measured, and as blank corrected, in situations where there is a significant difference between the two values.

Chapter 3
Air Dispersion and Deposition Modeling

What's Covered in Chapter 3:

3.1 Background on Air Dispersion Models for Risk Assessment

3.2 Partitioning Emissions

3.3 Site-Specific Characteristics Required for Air Modeling

3.4 Meteorological Data Primer

3.5 Meteorological Preprocessor Data Needs

3.6 ISCST3 Model Input Files

3.7 ISCST3 Model Execution

3.8 Using Model Output

3.9 Modeling Fugitive Emissions

3.10 Modeling Acute Risk

The burning of materials produces residual amounts of pollution that could be released to the environment. Knowledge of atmospheric pollutant concentrations and deposition rates in the areas around the combustion facility is an integral part of estimating potential human health risks associated with these releases. Air concentrations and deposition rates are usually estimated using air dispersion models. Air dispersion models are mathematical constructs that attempt to describe the effects of physical processes that occur in the atmosphere on rates of dispersion of emissions from a source (such as the stack of a combustor). These mathematical constructs are coded into computer programs to facilitate the computational process.

> ***PLEASE NOTE***: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.
>
> The "you" to which we speak in this chapter is the air modeler: the person (or persons) who will actually put the recommended air modeling methods into practice.

This chapter provides guidance on criteria to consider when selecting an air model to use to support a risk assessment. It also describes the development and use of the U.S. EPA air dispersion model—the Industrial Source Complex Short-Term Model (ISCST3)—which best addresses the selection criteria for most risk assessments. Although the air dispersion modeling methods presented in the HHRAP focus on use of ISCST3, other available models may be more appropriate if they better meet the selection criteria due to study-specific objectives or assessment area characteristics. Of particular interest is the American Meteorological Society- Environmental Protection Agency Regulatory Model (AERMOD), which has been proposed by EPA as a replacement for the ISCST3 model for Clean Air Act regulatory purposes. More details about AERMOD and other alternative models are provided in Section 3.1.1.

The *Guideline on Air Quality Models* (GAQM) (U.S. EPA 1996k; 1999b; Federal Register 2000) is a primary reference for all US EPA and state agencies on using air models for regulatory purposes. The GAQM is Appendix W of 40 CFR Part 51. The Office of Air Quality Planning and Support (OAQPS) provides the GAQM and extensive information on updates to air dispersion models, meteorological data, data preprocessors, and user's guides on the Support Center for Regulatory Air Models (SCRAM) web site at www.epa.gov/scram001/index.htm. We suggest you review this web site periodically to check for updates and changes to the recommended model. General questions regarding air modeling, or information on the web site may be addressed to atkinson.dennis@epamail.epa.gov. Specific questions on the use of this guidance may be addressed to the appropriate permitting authority. Please refer to the respective models User's Manual for:

- a more in-depth discussion on the physical assumptions embodied in the ISCST3 or other air dispersion model that may be considered; as well as
- concepts of atmospheric processes that are embodied in dispersion models in general.

Discussions in Section 3.1 focus on a background of available models, and selection criteria to consider when deciding which air model to use in a risk assessment. The remainder of the chapter provides information on ISCST3 input data needs, and output file development. Chapter sections include:

- Partitioning emissions (Section 3.2)
- Site-specific characteristics needed for air modeling (Section 3.3)
    - Surrounding terrain (Section 3.3.1)
    - Surrounding land use (Section 3.3.2)
    - Facility building characteristics (Section 3.3.4)
- Meteorological data (Section 3.5)

If you aren't familiar with the types of data used in air concentration and deposition models, Section 3.4 provides a short tutorial on the types of meteorological data needed, and potential data sources. Section 3.5 describes the data needs of the preprocessor computer program we recommend using to prepare, organize, and format meteorological data for use in the ISCST3 model. Section 3.6 describes the structure and format of ISCST3 input files. Section 3.7 describes limitations to consider in executing ISCST3. Section 3.8 describes how model outputs are used in the risk assessment computations. Section 3.9 discusses air modeling of fugitive emissions. Section 3.10 discusses air modeling for acute exposure.

3.1 DESCRIPTION OF AIR MODELS

This section gives a brief background on air dispersion models for risk assessment. This section also introduces some data preprocessing programs developed to aid in preparing air model input files (these preprocessing programs are described in more detail in Sections 3.3.3 and 3.5).

3.1.1 Background on Air Dispersion Models for Risk Assessment

Before 1990, the Agency and the regulated community used several air dispersion models. These models were of limited usefulness in risk assessments because they considered only air concentration, and not the deposition of contaminants to land. The original Agency guidance on completing risk assessments (U.S. EPA 1990e) identified two models that were explicitly formulated to account for the effects of deposition:

- COMPLEX I, from which a new model—COMPDEP—resulted
- Rough Terrain Diffusion Model (RTDM), from which a new model—RTDMDEP—resulted

An updated version of the ISCST model in use at the time, COMPDEP included building wake effects. Subsequent Agency guidance (U.S. EPA 1993f and 1994g) recommended using COMPDEP for air deposition modeling. U.S. EPA (1993f) specified COMPDEP Version 93252, and U.S. EPA (1994g) specified COMPDEP Version 93340. When these recommendations were made, a combined ISC-COMPDEP model (a merger of the ISCST2 and COMPLEX I model) was still under development. The merged model became known as ISCSTDFT. U.S. EPA guidance (1994r) recommended using the ISCSTDFT model. After reviews and adjustments, this model was released as ISCST3. The ISCST3 model contains algorithms for dispersion in simple, intermediate, and complex terrain; dry deposition; wet deposition; and plume depletion.

Operating the COMPDEP, RTDMDEP, and ISCST models is described in more detail in the following User's Manuals. Please note, though, that all models except the current version of ISCST3 are considered obsolete:

- Environmental Research and Technology (ERT). 1987. *User's Guide to the Rough Terrain Diffusion Model Revision 3.20*. ERT Document P-D535-585. Concord, Massachusetts.
- Turner, D.B. 1986. *Fortran Computer Code/User's Guide for COMPLEX I Version 86064: An Air Quality Dispersion Model in Section 4. Additional Models for Regulatory Use.* Source File 31 Contained in UNAMAP (Version 6). National Technical Information Service (NTIS) PB86-222361/AS.
- U.S. EPA. 1979. *Industrial Source Complex Dispersion Model User's Guide, Volume I.* Prepared by the H.E. Cramer Company. Salt Lake City, Utah. Prepared for the Office of Air Quality Planning and Standards. Research Triangle Park, North Carolina. EPA 450/4-79/030. NTIS PB80-133044.
- U.S. EPA. 1980b. *User's Guide for MPTER: A Multiple Point Gaussian Dispersion Algorithm with Optional Terrain Adjustment.* Environmental Sciences Research Laboratory. Research Triangle Park, North Carolina. EPA 600/8-80/016. NTIS PB80-197361.
- U.S. EPA. 1982a. *MPTER-DS: The MPTER Model Including Deposition and Sedimentation*. Prepared by the Atmospheric Turbulence and Diffusion Laboratory (NOAA). Oak Ridge, Tennessee. Prepared for the Environmental Sciences Research Laboratory. Research Triangle Park, North Carolina. EPA 600/8-82/024. NTIS PB83-114207.
- U.S. EPA. 1987b. *On-Site Meteorological Program Guidance for Regulatory Modeling Applications*. Office of Air Quality Planning and Standards. Research Triangle Park, North Carolina.
- U.S. EPA. 1995f. *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models, Volumes I and II.* Office of Air Quality Planning and Standards. Emissions, Monitoring, and Analysis Division. Research Triangle Park, North Carolina. EPA 454/B-95/003a. September.

Four air models in current regulatory use, or proposed by OAQPS, include ISCST3, AERMOD, CALPUFF and ISC-PRIME. However, we generally recommend using the latest version of ISCST3 in most situations to conduct air dispersion and deposition modeling for use in a risk assessment. This recommendation is based on ISCST3's status as the:

- Model most used by regional, state and local agencies; and

- Model satisfying the most of the following criteria:
    - Has broad regulatory acceptance and experience in evaluating the impacts of air contaminants emitted from various industrial sources (e.g., short and tall stack heights, fugitive emissions from process and storage areas);
    - Can evaluate impacts attributable to a single source, with capability to evaluate multiple sources;
    - Can evaluate compounds emitted as vapor or particulate phase with consideration of deposition and removal processes based on the physical characteristics of the compound;
    - Accepts placement of grid nodes at any location within the applicable range of the air model;
    - Can conduct stepwise evaluation of hourly meteorological conditions for multiple years of data producing short-term (acute) and long-term (chronic) averages as outputs; and
    - Can evaluate building downwash effects.

The other three air models are proposed by OAQPS as refined regulatory air models in the revisions to the Guideline on Air Quality Models (U.S. EPA 1996k; 1999b). Of particular interest is AERMOD. AERMOD is proposed to replace ISCST3 as the recommended air quality model for most regulatory applications (Federal Register 2000). AERMOD has been evaluated extensively by the air modeling community for improvements over the ISCST3 algorithms for vertical contaminant distribution from sources with tall stacks, terrain effects in areas with terrain elevations above the top of emission source stacks, and enhanced nighttime dispersion in urban areas. However, AERMOD currently has minimal regulatory experience. AERMOD would be an important consideration for risk assessments if:

- it gains broad regulatory acceptance,
- the assessment area contains significant terrain features (i.e., terrain elevations above emission source stacks)

OAQPS promulgated the California Puff Model (CALPUFF) model for use on a case-by-case basis, for long-range transport (greater than 50 kilometers), as well as the special conditions of very light or calm winds (less than 1 meter per second). CALPUFF may be an important consideration when modeling:

- chemical transformation of highly reactive contaminants;
- long residence times during extended periods of light winds or calm conditions;
- transport over long distances;
- recirculation of contaminants due to local wind effects; or
- hilly terrain and river valleys.

Also, the proposed OAQPS revisions to the GAQM recommend using the version of ISCST with the new downwash algorithm, ISC-PRIME, for applications where aerodynamic building downwash is critical (Federal Register 2000). The necessary information and inputs to evaluate the effects of building downwash are further discussed in Section 3.2.4. We generally recommend determining the need for the improved building downwash calculation on a case-by-base basis.

Although air dispersion modeling methods presented in this guidance use ISCST3, you may want to evaluate other models (such as ISC-PRIME, AERMOD, or CALPUFF, if the revised GAQM adopts them), to see whether they more substantially meet the selection criteria. Specific to its ability to meet the selection criteria listed above, the ISCST3 model is technically capable of evaluating:

- Gaussian dispersion rates in vertical and horizontal plume cross-section;
- Urban and rural dispersion coefficients;
- Terrain effects;
- Source characterization as a discrete point, two-dimensional area, or three-dimensional volume;
- Short-term and long-term averages (1-hour and annual);
- Surface meteorology data includes hourly observations of wind speed (nearest 1/10th mile per second), wind direction (nearest degree), stability class (6 categories), and temperature (nearest degree);
- Mixing height data interpolated from twice daily upper air soundings corresponding to each hour of surface data;
- Deposition processes for conservation of mass with particle wet and dry deposition and removal, and vapor wet deposition and removal (dry vapor deposition implemented within the air modeling; see Section 3.6.1);
- Hourly precipitation amount and type act on wet deposition and removal; and
- Single first order exponential decay rate.

In addition, emission sources can be defined in ISCST3 with either stack dimensions at a discrete point, as a two-dimensional area source, or in three dimensions as a volume source. Therefore, ISCST3 offers the flexibility to model sources based on source type (i.e., stack or fugitive area).

3.1.2 Preprocessing Programs

ISCST3 needs the help of additional computer programs, referred to as "preprocessing" programs. These programs format available information regarding surrounding buildings and meteorological data into a format that ISCST3 can read. Currently, these programs include:

- Building Profile Input Program (BPIP) calculates the maximum crosswind widths of buildings, which ISCST3 then uses to estimate the effects on air dispersion. These effects by surrounding buildings are typically referred to as building downwash or wake effects. The BPIP User's Guide contains detailed information for preparing the necessary building dimensions (length, height, and width) and locations for the ISCST3 model (U.S. EPA 1995c).

- Meteorological Processor for Regulatory Models (MPRM) prepares meteorological data for ISCST3. MPRM merges hourly measurements of surface parameters (e.g., precipitation, wind speed, and wind direction) into rows and columns of information that ISCST3 can read. The MPRM User's Guide contains instructions for using on-site meteorological data to prepare the necessary meteorological input file for ISCST3 (U.S. EPA 1996j; 1999c). The Addendum to the MPRM User's Guide describes the additional data needed in the meteorological input file for ISCST3 to model dry deposition of vapors (U.S. EPA 1999c).

We generally recommend using MPRM to process the meteorological data, because MPRM provides all the meteorological parameter values that ISCST3 needs to function. The peer review draft of the HHRAP described the personal computer version of the meteorological preprocessor for the old RAM program (PCRAMMET). Unfortunately, PCRAMMET doesn't provide the parameters needed for evaluating dry vapor deposition in ISCST3. Therefore, we no longer recommend using PCRAMMET.

When sufficient site-specific data is available on the location, size and shape of structures in the vicinity of the emissions sources, you can use the BPIP program to prepare the ISCST3 input file to consider the effects of building downwash on pollutant transport. If sufficient site-specific data is not available to run BPIP, then you will not be able to address building downwash in air modeling efforts. Prior to performing the air dispersion modeling, we recommend that you consult with the appropriate parties (e.g. regulatory authority and facility) on the decision to include or omit nearby structures.

3.2 PARTITIONING EMISSIONS

COPC emissions to the environment occur in either vapor or particle phase. In general, you can assume:

- most metals and organic COPCs with very low volatility (fraction of COPC in vapor phase [*Fv*] less than 0.05, see Appendix A-3) occur only in the particle phase;
- highly volatile organic COPCs occur only in the vapor phase (*Fv* of 1.0, see Appendix A-3); and
- the remaining organic COPCs occur with a portion of the vapor condensed onto the surface of particulates (i.e. particle-bound).

For modeling COPCs released only as particulates, the mass fractions allocated to each particle size are different than the mass fractions used for modeling organics released in both the vapor and particle-bound phases.

Due to model limitations, you need to run ISCST3 multiple times to generate estimates of vapor phase COPCs, particle phase COPCs, and particle-bound COPCs. An example is the risk assessment for the WTI incinerator located in East Liverpool, Ohio. The study used three runs; a vapor phase run for organic COPCs, a particle run with mass weighting of the particle phase metals and organic COPCs with very low volatility, and a particle run with surface area weighting of the particle-bound organic COPCs.

3.2.1 Vapor Phase Modeling

Vapor phase air modeling runs do not need a particle size distribution in the ISCST3 input file. ISCST3 output for vapor phase runs are vapor phase ambient air concentration, dry vapor deposition, and wet vapor deposition at receptor grid nodes based on the unit emission rate.

3.2.2 Particle Phase Modeling (Mass Weighting)

ISCST3 uses algorithms to compute the rate at which dry and wet removal processes deposit particle phase COPCs onto the Earth’s surface. Particle size is the main determinant of the fate of particles in air flow, whether dry or wet. The key to dry particle deposition rate is the terminal, or falling, velocity of a particle. Particle terminal velocity is calculated mainly from the particle size and particle density. Large particles fall more rapidly than small particles, and are deposited closer to the stack. Small particles have low terminal velocities. Very small particles remain suspended in the air flow. Wet particle deposition also depends on particle size as larger particles are more easily removed, or scavenged, by falling liquid (rain) or frozen (snow or sleet) precipitation. To estimate particle phase deposition rates, ISCST3 needs an initial estimate of the particle size distribution, broken out by particle diameter.

The diameters of small particles contained in stack emissions are usually measured in micrometers. The distribution of particle diameters will differ from one combustion process to another, and is greatly dependent on the:

1. Furnace type;
2. Combustion chamber design;
3. Feed fuel composition;
4. Particulate removal efficiency;
5. APCS design;
6. Amount of air, in excess of stoichiometric amounts, that is used to sustain combustion; and
7. Combustion temperature.

The particle size distribution cannot be calculated, but only directly measured or inferred from prior data. Unfortunately, few studies have been performed to directly measure particle size distributions from a variety of stationary combustion sources (U.S. EPA 1986a).

We recommend that existing facilities perform stack tests to identify particle size distribution. We further recommend that these data represent actual operating conditions for the combustion unit and air pollution control device (APCD) that remove particulate from the stack gas. A table of particle size distribution data may be prepared using stack test data in the format similar to the example illustrated in Table 3-1.

TABLE 3-1

HYPOTHETICAL PARTICLE SIZE DISTRIBUTION DATA TO SUPPORT EXAMPLE CALCULATIONS

1	2	3	4	5	6
Mean Particle Diameter [a] (µm)	**Particle Radius (µm)**	**Surface Area/ Volume (μm^{-1})**	**Fraction of Total Mass[b]**	**Proportion Available Surface Area**	**Fraction of Total Surface Area**
> 15.0	7.50	0.400	0.128	0.0512	0.0149
12.5	6.25	0.480	0.105	0.0504	0.0146
8.1	4.05	0.741	0.104	0.0771	0.0224
5.5	2.75	1.091	0.073	0.0796	0.0231
3.6	1.80	1.667	0.103	0.1717	0.0499
2.0	1.00	3.000	0.105	0.3150	0.0915
1.1	0.55	5.455	0.082	0.4473	0.1290
0.7	0.40	7.500	0.076	0.5700	0.1656
< 0.7	0.40	7.500	0.224	1.6800	0.4880

a Geometric mean diameter in a distribution from U.S. EPA (1980a), as presented in U.S. EPA (1998c)
b The terms mass and weight are used interchangeably when using stack test data

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We expect that actual stack test data will be different from the hypothetical values presented in Table 3-1. This is because actual stack sampling will use particle "cut size" for the different cascade impactor filters (or Coulter counter-based distributions). The test method will drive the range of particle sizes that are presented in the results of the stack test. However, ISCST3 needs mean particle diameter for each particle size distribution, and the stack test data identifies only the mass ("weight") of particles in a range bounded by two specific diameters. To address this, we recommend that stack test data be converted into a mean particle diameter which approximates the diameter of all the particles within a defined range. We recommend using the following equation to calculate the mean particle diameter:

$$D_{mean} = [0.25 \cdot (D_1^3 + D_1^2 D_2 + D_1 D_2^2 + D_2^3)]^{1/3} \qquad \text{Equation 3-1}$$

where

D_{mean} = Mean particle diameter for the particle size category (µm)
D_1 = Lower bound cut of the particle size category (µm)
D_2 = Upper bound cut of the particle size category (µm)

For example, the mean particle diameter of 5.5 µm in Table 3-1 is calculated from a lower bound cut size (assuming a cascade impactor is used to collect the sample) of 5.0 µm to an upper bound cut size of 6.15 µm. In this example, the mean particle diameter is calculated as:

$$D_{mean} = [0.25\ (5.0^3 + (5.0)^2(6.15) + (5.0)(6.15)^2 + 6.15^3)]^{1/3} = 5.5\ \mu m$$

From Table 3-1, the mean particle diameter is 5.5 µm. The mass of particulate from the 5.0 µm stack test data is then assigned to the 5.5 µm mean particle diameter for the purpose of computing the "fraction of total mass."

Typically, eight to ten mean particle diameters are available from stack test results. However, as determined from a sensitivity analysis conducted by The Air Group-Dallas under contract to U.S. EPA Region 6 (http://www.epa.gov/region06), a minimum of three particle size categories (> 10 microns, 2-10 microns, and < 2 microns) detected during stack testing are generally the most sensitive to air modeling with ISCST3 (U.S. EPA 1997f). For facilities with stack test results which indicate mass amounts lower than the detectable limit (or the filter weight is less after sampling than before), a single mean particle size diameter of 1.0 micron may be used to represent all mass (e.g., particle diameter of 1.0 micron or a particle mass fraction of 1.0) in the particle and particle-bound model runs. Because rudimentary methods for stack testing may not detect the very small size or amounts of COPCs in the particle phase, the use of a 1.0 micron particle size will allow these small particles to be included properly as particles in the risk assessment exposure pathways while dispersing and depositing in the air model similar in behavior to a vapor. Consequently, we recommend that a minimum of three particle size categories be used in the air modeling effort.

After calculating the mean particle diameter (Column 1), you can compute the fraction of total mass per mean particle size diameter (Column 4) from the stack test results. For each mean particle diameter, the stack test data provides an associated particulate mass. Calculate the fraction of total mass for each mean particle diameter by dividing the associated particulate mass for that diameter by the total particulate mass in the sample. In many cases, the fractions of total mass will not sum to 1.0 because of rounding errors. In these instances, we advocate forcing the total mass to 1.0 by adding the remaining mass fraction into the largest mean particle diameter mass fraction.

Direct measurements of particle-size distributions may not be available at a proposed new facility. You will need to supply ISCST3 with an assumed particle distributions. In such instances, you may use a representative distribution. We recommend that the combustor on which the representative distribution is based be as similar as practicable to the proposed combustor.

ISCST3 uses the mass-based particle size distribution to apportion the mass of particle phase COPCs (metals and organics with F_v values less than 0.05) according to particle size. The ISCST3 input file uses the data in Column 4 of Table 3-1 (as developed from actual stack test data) to perform a particulate run with the particle phase COPCs apportioned per mass weighting.

3.2.3 Particle-Bound Modeling (Surface Area Weighting)

Use a surface area weighting of particles, instead of mass weighting, in separate particle runs of ISCST3. Surface area weighting approximates the situation where a portion of a semivolatile organic contaminant, volatilized in the high temperature environment of a combustion system, condenses to the surface of particles entrained in the combustion gas after the gas cools in the stack. Apportioning emissions by particle diameter becomes a function of the surface area of particles available for chemical adsorption (U.S. EPA 1998c).

The first step in apportioning COPC emissions by surface area is to calculate the proportion of available surface area of the particles. If particle density is held constant (such as 1 g/cm^3), the proportion of available surface area of aerodynamically spherical particles is the ratio of surface area (S) to volume (V), is estimated as follows:

- Assume aerodynamically spherical particles.
- Specific surface area of a spherical particle with a radius, r—$S = 4\pi r^2$
- Volume of a spherical particle with a radius, r—$V = 4/3\pi r^3$

- Ratio of *S* to *V*—$S/V = 4\ \pi r^2 / (4/3\ \pi r^3) = 3/r$

The following example uses the hypothetical particle size distribution in Table 3-1 to apportion the emission rate of the particle-bound portion of the COPC based on surface area. We generally recommend following this procedure for apportioning actual emissions to the actual particle size distribution measured at the stack.

In Table 3-1, a spherical particle having a diameter of 15 µm (Column 1) has a radius of 7.5 µm (Column 2). The proportion of available surface area (assuming particle density is constant) is S/V = 3/7.5 = 0.4, which is the value in Column 3. Column 4 shows that particles with a mean diameter of 15 µm constitute 12.8 percent of the total mass. Multiplying Column 3 by Column 4 yields the value in Column 5 of 0.0512. This value is an approximation of the relative proportion of total surface area, based on the percent of particles that are 15 µm in diameter. Totaling Column 5 yields the total surface area of all particles in the particle size distribution. In this example, the sum is 3.4423. Column 6 is the fraction of total surface area represented by the specific particle diameter in the distribution. Column 6 is calculated by dividing the relative proportion of surface area (Column 5) for a specific diameter by the total relative proportion of surface area (3.4423 square micrometers [μm^2]). In the example of the 15 µm-diameter particle, the fraction of total surface area available for adsorption is 0.0149 (0.0512/3.4423). This procedure is then repeated for all particle sizes in the array.

ISCST3 uses the surface area-based particulate size distribution to apportion mass of particle-bound COPCs (most organics) according to particle size. The ISCST3 input file uses Column 6 of Table 3-1 (as developed from actual stack test data) to perform a particulate run for the particle-bound COPCs apportioned according to surface area weighting.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Copies of all stack test data used to determine particle size distribution
- Copies of all calculations made to determine particle size distribution, fraction of total mass, and fraction of total surface area

3.3 SITE-SPECIFIC INFORMATION NEEDED FOR AIR MODELING

We generally recommend that the site-specific information for the facility and surrounding area used in air dispersion modeling include:

1. the elevation of the surrounding land surface or terrain;
2. surrounding land uses; and
3. characteristics of on-site buildings that may affect the dispersion of COPCs into the surrounding environment.

You can often obtain some of the site-specific information needed for air dispersion modeling by reviewing available maps and other graphical data on the area surrounding the facility. The first step in the air modeling process is reviewing these resources. Using U.S. Geological Survey (USGS) 7.5-minute topographic maps (1:24,000) extending to 10 kilometers from the facility, and USGS 1:250,000 maps extending out to 50 kilometers, you can identify much of the following:

- site location,
- nearby terrain features,
- waterbodies and watersheds,
- ecosystems,
- nearby residences, and
- land use.

Aerial photographs are frequently available to supplement the depiction of the area. An accurate facility plot plan—showing buildings, stacks, property and fence lines—is also needed. Facility information including stack and fugitive source locations, building corners, plant property, and fence lines, in Universal Transverse Mercator (UTM[1]) grid coordinates in meters east and north might be available in both USGS reference systems. Therefore, knowledge of the horizontal datum of the geographic coordinates from which the UTM coordinates were projected is required. We generally recommend consulting with personnel experienced and knowledgeable in the intricacies of these types of coordinate systems, and the various software programs used to conduct conversions.

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[1] UTM is a map coordinate projection of geographic coordinates.

Most USGS paper 7.5-minute topographic maps are published in the North American Datum system established in 1927 (NAD 27). However, most digital elevation data (e.g., USGS Digital Elevation Mapping) is in the 1983 revised system (NAD 83). We recommend taking special care not to mix source data based on NAD 27 with data based on NAD 83. Also, we recommend obtaining emission source information in the original units from the facility data, and convert it to metric units for air modeling, if necessary. You can get digital terrain data from USGS, or another source.

The following subsections describe the specific information we generally recommend collecting. Entering this information into the ISCST3 input files is described in Section 3.6.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- All site-specific maps, photographs, or figures used in developing the air modeling approach
- Mapped identification of facility information including stack and fugitive source locations, locations of facility buildings surrounding the emission sources, and property boundaries of the facility

3.3.1 Surrounding Terrain Information

Terrain is important to air modeling because air concentrations and deposition rates are greatly influenced by the height of the plume above local ground level. Terrain is characterized by elevation relative to stack height. For air modeling purposes, terrain is referred to as “complex” if the elevation of the surrounding land within the assessment area—typically defined as anywhere within 50 kilometers of the stack—is above the top of the stack evaluated in the air modeling analysis. Terrain at or below stack top is referred to as “simple.” ISCST3 implements U.S. EPA guidance on the proper application of air modeling methods in all terrain if you include terrain elevation for each receptor grid node and specify the appropriate control parameters in the input file.

Even small terrain features may have a large impact on the air dispersion and deposition modeling results and, ultimately, on the risk estimates. We generally recommend that air modeling include terrain elevations for every receptor grid node. Some exceptions may be those sites characterized by very flat terrain where the permitting authority has sufficient experience to comfortably defer the use of terrain data because the terrain’s historical effect on air modeling results have been minimal.

We generally recommend including electronic copies of the digital terrain data used to extract receptor grid node elevations in the risk assessment report. One way to obtain receptor grid node elevations is using digital terrain data available from the USGS on the Internet at web site http://edc.usgs.gov/guides/dem.html. For most locations, an acceptable degree of accuracy is provided by the USGS "One Degree" (e.g., 90 meter data) data available as "DEM 250" 1:250,000 scale for the entire United States free of charge. For areas requiring more accurate terrain, USGS 30-meter (1:24,000 scale) data might be considered, though it is not universally available. Either 90-meter or 30-meter data is sufficient for most risk assessments which utilize 100 meter or greater grid spacing. Digital terrain data is also available for purchase from a variety of commercial vendors, who may require vendor-provided programs to extract the data. You can also manually extract the elevation of each receptor grid node from USGS topographic maps.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Description of the terrain data used for air dispersion modeling
- Summary of any assumptions made regarding terrain data
- Description of the source of any terrain data used, including any procedures used to manipulate terrain data for use in air dispersion modeling

3.3.2 Surrounding Land Use Information

Land use information is needed in the risk assessment for air dispersion modeling, as well as identifying and selecting exposure scenario locations (see Chapter 4). Land use analysis for air dispersion modeling usually occurs out to a radius of 3 kilometers from the centroid of the stacks from which emissions are being modeled. Certain land uses, as defined by air modeling guidance, affect the selection of air dispersion modeling variables. These variables are known as dispersion coefficients and surface roughness. You typically use USGS 7.5-minute topographic maps, aerial photographs, or visual surveys to define the air dispersion modeling land uses.

Land use information is also important for selecting exposure scenario locations, but at a radius further (50 kilometers) from the emission source(s), to make sure all receptors that may be impacted are identified. In most cases, though, air modeling performed out to a radius of 10 kilometers allows adequate characterization for evaluating exposure scenario locations. If you are evaluating a facility with multiple stacks or emission sources, we generally recommend extending the radius from the centroid of a polygon drawn connecting the various stack coordinates.

3.3.2.1 Land Use for Dispersion Coefficients

You need to specify the appropriate dispersion coefficients for ISCST3 to run properly. Land uses need to be defined in order to specify dispersion coefficients. We generally recommend using the Auer method specified in the Guideline on Air Quality Models (40 CFR Part 51, Appendix W) (U.S. EPA 1996k; 1999b) to define land use. Land use categories of "rural" or "urban" are taken from Auer (1978). Areas typically defined as rural include residences with grass lawns and trees, large estates, metropolitan parks and golf courses, agricultural areas, undeveloped land, and water surfaces. Auer typically defines an area as "urban" if it has less than 35 percent vegetation coverage or the area falls into one of the following use types:

TABLE 3-2

URBAN LAND USE TYPES

Urban Land Use		
Type	**Use and Structures**	**Vegetation**
I1	Heavy industrial	Less than 5 percent
I2	Light/moderate industrial	Less than 5 percent
C1	Commercial	Less than 15 percent
R2	Dense single/multi-family	Less than 30 percent
R3	Multi-family, two-story	Less than 35 percent

In general, the Auer method is described as follows:

Step 1 Draw a radius of 3 kilometers from the center of the stack(s) on the site map.

Step 2 Inspect the maps, and define in broad terms whether the area within the radius is rural or urban, according to Auer's definitions.

Step 3 Classify smaller areas within the radius as either rural or urban, based on Auer's definitions. (It may be prudent to overlay a grid [for example, 100 by 100 meters] and identify each square as primarily rural or urban)

Step 4 Count the number of rural squares; if more than 50 percent of the total squares are rural, the area is rural; otherwise, the area is urban.

Alternatively, digital land use databases may be used in a computer-aided drafting system to perform this analysis.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Description of the methods used to determine land use surrounding the facility
- Copies of any maps, photographs, or figures used to determine land use
- Description of the source of any computer-based maps used to determine land use

3.3.2.2 Land Use for Surface Roughness Height (Length)

Surface roughness height—also referred to as (aerodynamic) surface roughness length—is the height above the ground at which the wind speed goes to zero. Surface roughness affects the height above local ground level that a particle moves from the ambient air flow above the ground (for example in the plume) into a "captured" deposition region near the ground. That is, ISCST3 assumes the particle is deposited to the ground at some point above the actual land surface, based on surface roughness height. Surface roughness height is defined by individual elements on the landscape, such as trees and buildings.

There are several methods for computing surface roughness. One uses measurements of hourly surface roughness based on wind direction (fetch). Another uses the change in wind speed with height in the surface layer, as measured on an instrumented tower operated at the site. We recommend deciding which method to use to characterize surface roughness based on the variability of surface features and seasonal values for the site. In lieu of any other method, the paragraphs below describe our recommendations for computing surface roughness.

In order to be consistent with the recommended method for determining land use for dispersion coefficients (Section 3.3.2.1), the land use within 3 kilometers generally is acceptable for determining surface roughness. Surface roughness height values for various land use types are as follows:

TABLE 3-3

SURFACE ROUGHNESS HEIGHTS

Surface Roughness Heights for Land Use Types and Seasons (meters)				
Land Use Type	**Spring**	**Summer**	**Autumn**	**Winter**
Water surface	0.0001	0.0001	0.0001	0.0001
Deciduous forest	1.00	1.30	0.80	0.50
Coniferous forest	1.30	1.30	1.30	1.30
Swamp	0.20	0.20	0.20	0.05
Cultivated land	0.03	0.20	0.05	0.01
Grassland	0.05	0.10	0.01	0.001
Urban	1.00	1.00	1.00	1.00
Desert shrubland	0.30	0.30	0.30	0.15

Source: Sheih, et al. (1979)

If a significant number of buildings are located in the area, higher surface roughness heights (such as those for trees) may be appropriate (U.S. EPA 1995g). Previous guidance documents do not recommend a specific method for determining average surface roughness height. If you are using National Weather Service surface meteorological data, we generally recommend setting the surface roughness height for the measurement site at 0.10 meters (grassland, summer). If you intend to propose a different value for the measurement site, we recommend using the following procedure to determine the value:

Step 1 Draw a radius of 3 kilometers from the center of the stack(s) on the site map.

Step 2 Inspect the maps, and use professional judgment to classify the area within the radius according to land use type (for example water, grassland, cultivated land, and forest); a site visit may be necessary to verify some classifications.

Step 3 Divide the circular area into 12 angular sectors of 30 degrees.

Step 4 Identify a single representative surface roughness height for each sector, based on an area-weighted average of the land use types within the sector using the categories identified above.

Step 5 Input the area-weighted surface roughness height value representative of each of the 12 sectors into MPRM for preprocessing the meteorological data file.

Site-specific conditions might be such that you consider methods other than those described above more appropriate for determining surface roughness height. In such instances, we recommend clearly identifying and discussing the alternative methods with the appropriate parties (regulatory authority or facility) prior to use.

3.3.3 Information on Facility Building Characteristics

Building wake effects, also referred to as "building downwash," have a significant impact on the concentration and deposition of COPCs near the stack. Building wake effects are flow lines that cause plumes to be forced down to the ground much sooner than they would if the building was not there. The ISCST3 model contains algorithms for evaluating this phenomenon. We recommend that the downwash analysis consider all nearby structures with heights at least 40 percent of the height of the shortest stack to be modeled. The 40 percent value is based on Good Engineering Practice (GEP) stack height of 2.5 times the height of nearby structures or buildings (stack height divided by 2.5 is equal to 0.40 multiplied by the stack height [40 CFR Part 51 Appendix W]). BPIP uses Building dimensions and locations, and stack heights and locations, to identify the potential for building downwash. BPIP and the BPIP User's Guide are available for download from the SCRAM web site (http://www.epa.gov/scram001/) to address specific questions. The BPIP output file is in a format that can be copied and pasted into the source (SO) pathway of the ISCST3 input file. We generally recommend the following procedure to identify buildings for input to BPIP:

Step 1 Lay out facility plot plan, with buildings and stack locations clearly identified (building heights identified for each building); for buildings with more than one height or roof line, identify each height (BPIP refers to each height as a tier).

Step 2 Identify the buildings to be included in the BPIP analysis by comparing building heights to stack heights. The building height test specifies that only buildings at least 40 percent of the height of a potentially affected stack be included in the BPIP input file. For example, if a combustion unit stack is 50 feet high, only buildings at least 20 feet tall (0.40 multiplied by 50 feet) will affect air flow at stack top. Any buildings shorter than 20 feet need not be included in the BPIP analysis. The building height test is performed for each stack and each building.

Step 3 Use the building distance test to check each building to be included in BPIP from the building height test. For the building distance test, only buildings "nearby" the stack will affect air flow at stack top. "Nearby" is defined as "five times the lesser of building height or crosswind width" (U.S. EPA 1995c). A simplified distance test may be used by considering only the building height rather than the crosswind width. While some buildings with more height than width will be included unnecessarily using this simplification, BPIP will identify correctly only the building dimensions required for ISCST3.

An example, hypothetical plot plan identifies a 25-foot tall building that is 115 feet from the 50-foot tall combustion unit stack center to the closest building corner. The building distance test, for this building only, is five times the building height, or 125 feet (five multiplied by the building height, 25 feet). This building would be included in the BPIP analysis, because it passes the building height test and building distance test.

Step 4 Repeat steps 2 and 3 for each building and each stack, identifying all buildings to be included in the BPIP. If the number of buildings exceeds the BPIP limit of eight buildings, consider combining buildings, modifying BPIP code for more buildings, or using third-party commercial software which implements BPIP. If two buildings are closer than the height of the taller building, the two buildings may be combined. For example, two buildings are 40 feet apart at their closest points. One building is 25 feet high, and the other building is 50 feet high. The buildings could be combined into one building for input to BPIP. For input to BPIP, the corners of the combined building are the outer corners of the two buildings. For unusually shaped buildings with more than the eight corners allowed by BPIP, approximate the building by using the eight corners that best represent the extreme corners of the building. The BPIP User's Guide contains additional description and illustrations on combining buildings, and BPIP model limitations (U.S. EPA 1995c).

Step 5 Mark off the facility plot plan with UTM grid lines. Extract the UTM coordinates of each building corner and each stack center to be included in BPIP input file. Although BPIP allows the use of "plant coordinates," we recommend that all inputs to the air model be prepared using UTM coordinates (meters) for consistency. UTM coordinates are rectilinear, oriented to true north, and needed in metric units for ISCST3 modeling. Almost all air modeling will need to use USGS topographic data (digital and maps) for receptor elevations, terrain grid files, locating plant property, and identifying surrounding site features. Therefore, using an absolute coordinate system will enable you to check inputs at each step of the analysis. Also, the meteorological data are oriented to true north. Significant errors will result from ISCST3 if incorrect stack or building locations are used, plant north is incorrectly rotated to true north, or incorrect base elevations are used. With computer run times requiring many hours (up to 40 hours for one deposition run with depletion), verifying locations at each step while preparing model inputs will prevent the need to remodel.

We recommend observing several precautions and guidelines while preparing input files for BPIP:

- Graphically confirming the correct locations before BPIP is run. One method is to plot the buildings and stack locations using graphics software. Several commercial programs incorporating BPIP provide graphic displays of BPIP inputs.
- In addition to using UTM coordinates for stack locations and building corners, also use meters as the units for height.
- Carefully include the stack base elevation and building base elevations according to BPIP User's Guide instructions.
- Note that the BPIP User's Guide (revised February 8, 1995) has an error on page 3-5, Table 3-1, under the "TIER(i,j)" description, which incorrectly identifies tier height as base elevation.
- BPIP mixes the use of "real" and "integer" values in the input file. To prevent possible errors in the input file, note that integers are used where a count is requested (for example, the number of buildings, number of tiers, number of corners, or number of stacks).

- Make the stack identifiers (up to eight characters) in BPIP identical to those used in the ISCST3 input file, or ISCST3 will report errors.

For most sites, BPIP executes in less than 1 minute. The array of 36 building heights and 36 building widths (one for each of 36 10-degree direction sectors) are input into the ISCST3 input file by cutting and pasting from the BPIP output file. We recommend deleting the five blank spaces preceding "SO" in the BPIP output file, so that the "SO" begins in the first column of the ISCST3 input file.

One use of BPIP is to design stack heights for new facilities. Another is to determine what stack height increases are necessary to avoid building influence on air flow, which may cause high concentrations and deposition near the facility. The output for BPIP provides the GEP heights for stacks. Significant decreases in concentrations and deposition rates will begin at stack heights at least 1.2 times the building height, and further decreases occur at 1.5 times building height, with continual decreases of up to 2.5 times building height (GEP stack height) where the building no longer influences stack gas.

3.4 METEOROLOGICAL DATA PRIMER

Air dispersion and deposition modeling is extremely complicated, and uses data potentially unfamiliar if your expertise lies outside meteorology and air modeling. Section 3.4 is an introduction to the types of meteorological data used by air dispersion and deposition models, as well as information on potential data sources. Section 3.5 continues this introduction into the specific data parameters required by MPRM, the data preprocessor for the ISCST3 air model.

Generally speaking, air concentration and deposition models attempt to characterize the effects of atmospheric forces on a mass of COPC found in the air. These forces can be divided between:

- those which transport the COPC (the dominant factor is wind);
- those which disturb the straight flow of the COPC away from the source (also known as mixing, examples include building downwash effects, and atmospheric stability); and
- those actors which remove, or conditions which encourage the removal of, the COPC from the atmosphere (An example of a removal actor would be rain, whereas an example "condition" would be the point above the ground at which windspeed slows to zero).

Air concentration and deposition models need a variety of meteorological information, at different levels of temporal definition. The ISCST3 model requires the following:

- Hourly values (also know as "surface data, as they describe conditions closer to ground level)
  - a. Wind direction (degrees from true north)
  - b. Wind speed (m/s)
  - c. Dry bulb (ambient air) temperature (K)
  - d. Opaque cloud cover (tenths)
  - e. Cloud ceiling height (m)
  - f. For dry particle deposition:
    - i. surface pressure (millibar)
    - ii. solar radiation (watts/m^2)
  - g. For wet particle deposition:
    - i. Precipitation amount (inches)
    - ii. Precipitation type (liquid or frozen)

- Daily values (also called "upper air data" as they describe conditions higher in the atmosphere)
  - a. Morning mixing height (m)
  - b. Afternoon mixing height (m)

The following subsections describe important characteristics of the needed data.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Identification of all sources of meteorological data

3.4.1 Wind Direction and Wind Speed

Wind direction and speed are two of the most critical parameters in air modeling. The wind direction promotes higher concentration and deposition if it persists from one direction for long periods during a year. For example, a predominantly south wind, such as on the Gulf Coast of Texas, will contribute to high concentrations and depositions north of the facility. Wind speed is inversely proportional to concentration predicted using air modeling: The higher the wind speed, the lower the concentration will be. If wind speed doubles, the concentration and deposition will be reduced by one-half. Air models need wind speed and wind direction at the stack top. Most air modeling is performed using government sources of surface data. Wind data are typically measured by National Weather Service (NWS) stations at a height of 10 meters. However, since some stations have wind speed recorded at a different height, we recommend always verifying the anemometer height, and that the correct value is input into the meteorological data preprocessing program. ISCST3 uses a wind speed profile to calculate wind speed at stack top. This calculation exponentially increases the measured wind speed from the measured height to a calculated wind speed at stack height. ISCST3 assumes that wind direction at stack height is the same as that measured at the NWS station height (U.S. EPA 1995c).

3.4.2 Dry Bulb Temperature

Dry bulb temperature, or ambient air temperature, is the same temperature reported on the television and radio stations across the country each day. Dry bulb (ambient) temperature contributes to describing the surface conditions that vary from hour to hour in the measured meteorological data needed for air modeling. It therefore has a direct effect on the modeling results, as described below.

Dry bulb temperature is typically measured at 2 meters above ground level. ISCST3 uses air temperature in the buoyant plume rise equations developed by Briggs (U.S. EPA 1995f). The model results are not very sensitive to air temperature, except at extremes. However, buoyant plume rise is very sensitive to the stack gas temperature. Buoyant plume rise is mainly a result of the difference between stack gas temperature and ambient air temperature. Conceptually, it is similar to a hot air balloon: The higher the stack gas temperature, and lower the ambient temperature, the higher will be the plume rise. High plume heights result in low concentrations and depositions as the COPCs travel further and are diluted in a larger volume of ambient air before reaching the surface. A large variation in ambient temperature will affect buoyant plume rise, but not as much as variations in stack gas temperature. The fact that the stack gas temperature is constant (for most modeling analyses) is the very point for noting that it is the changes in the dry bulb temperature that creates the temperature differential affecting plume rise in the air model (see also Section 3.6.2.2 regarding selection of stack gas exit temperature). The temperature is measured in K, so a stack gas temperature of 450°F is equal to 505 K, an ambient temperature of 90°F = 305 K, and 32°F = 273 K.

When determining values for dry bulb temperature (or ambient air temperature), it is important not to appear to be ‘data shopping’ by artificially selecting a period of meteorological data that has either average or above-average ambient temperatures that will reduce plume rise. The skewing of results in one direction or another may over-estimate concentrations near the source, but under-estimate concentrations and depositions away from the source where more sensitive receptors may be located (see Chapter 4 for more on Exposure Scenario location) . Attempts to increase protectiveness by applying such approaches may not achieve the desired result.

If the meteorological data sets you use, and your choice of dry bulb temperature are not consistent with the HHRAP, we recommend clearly identifying and discussing them in the risk plan and/or cost estimate (as appropriate) to ensure clarity and transparency of the final risk assessment results.

3.4.3 Opaque Cloud Cover

Observations of opaque cloud cover are used to calculate the stability of the atmosphere. Stability determines the dispersion, or dilution, rate of COPCs in the atmosphere. Rapid dilution occurs in unstable air conditions, while stable air results in very little mixing, or dilution, of the emitted COPCs.

With clear skies during the day, the sun heats the Earth's surface, which heats the air immediately above it. The warm air rises and overturns with the cooler air above it. During this "unstable" condition, while layers of air are moving through one another, the stack plume mixes as the air mixes.

A cool surface occurs at night because of radiative loss of heat on clear nights. With a cloud cover, surface heating during the day and heat loss at night are reduced, resulting in moderate mixing rates, or neutral stability.

Opaque cloud cover is a measure of the transparency of the clouds. For example, a completely overcast sky with 10/10ths cloud cover may have only 1/10th opaque cloud cover if the clouds are high, translucent clouds that do not prevent sunlight from reaching the Earth's surface.

3.4.4 Cloud Ceiling Height

ISCST3 needs cloud ceiling height to calculate stability. Specifically, the height of the cloud cover affects the heat balance at the Earth's surface.

3.4.5 Surface Pressure

The MPRM preprocessor for ISCST3 requires station (i.e. surface) pressure. MPRM uses station pressure to compute Monin-Obukhov Length and Friction Velocity, two boundary layer parameters that ISCST3 needs to perform dry particle deposition. ISCST3 is not very sensitive to surface pressure.

3.4.6 Incoming Short-wave Radiation\Leaf Area Index

Solar radiation affects the respiratory activity of leaf surfaces, which affects the rate of dry vapor deposition. We recommend using the method described in Section 3.6.1 to address the effects of dry vapor deposition. Specify a single value in the ISCST3 input file for dry vapor deposition velocity for all hours. Even though incoming solar radiation and leaf area index are not used when specifying the dry vapor deposition velocity, the ISCST3 model will not run properly if values for these two data fields aren't included in the meteorological input file.

Though we don't recommend it, ISCST3 is able to compute the hourly dry vapor deposition velocity by combining hourly incoming short-wave (solar) radiation with the user-specified, site-specific leaf area index. The default value specified in the Addendum to the MPRM User's Guide (U.S. EPA 1999c) may be used for leaf area index when site-specific data is not available.

3.4.7 Precipitation Amount and Type

In order to calculate wet deposition of vapor and particles, ISCST3 requires that MPRM process precipitation amount and type into the ISCST3 meteorological file. Precipitation is measured at 3 feet (roughly 1 meter) above ground level. The amount of precipitation will directly influence the amount of wet deposition at a specific location. Particles and vapor are both captured by falling precipitation, known as precipitation scavenging.

Scavenging coefficients are needed as inputs to ISCST3 for vapors, with a rate specified each for liquid and frozen precipitation. The precipitation type in a SAMSON weather report (see Section 3.4.9 for more information on SAMSON) will identify to ISCST3 which event is occurring for appropriate use of the scavenging coefficients entered (see Section 3.6.2.6). MPRM can also read supplemental precipitation files from NCDC, and integrate the data with CD-144 data in the ISCST3 meteorological file. We discuss the importance of precipitation to ISCST3 results further in Section 3.4.9 (Potential Data Sources).

3.4.8 Upper Air Data (Mixing Height)

Upper air data are needed to run the ISCST3 model. MPRM uses measurements of morning and afternoon (twice daily) upper air data to calculate an hourly mixing height using interpolation methods

(U.S. EPA 1996j). We recommend that only years with complete upper air data be used as input for air modeling. It is important that the years you select for upper air data match the years you select for surface data. If matching years of data are not available from a single upper air station, we recommend another upper air station be used for completing the five years. We also recommend discussing your choice of representative data with the appropriate authorities prior to performing air modeling.

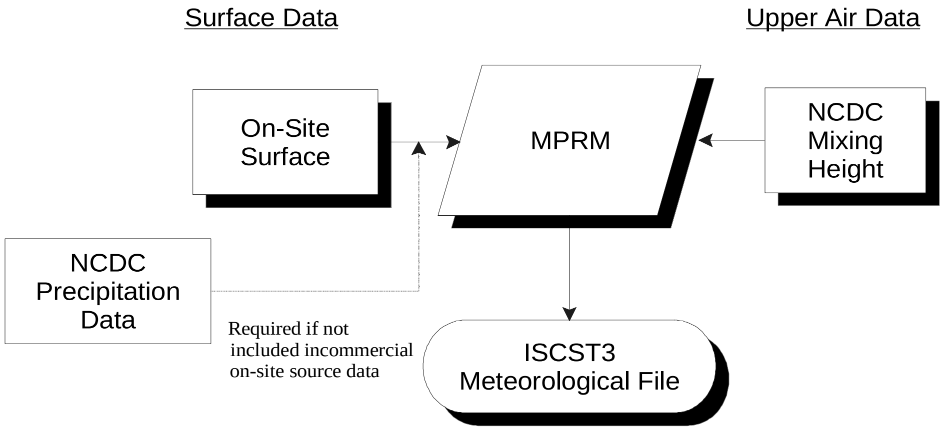
3.4.9 Potential Data Sources

As shown in Figure 3-1, these data are available from several different sources. For most air modeling, we recommend five years of data from a representative NWS station. However, in some instances where the closest NWS data is clearly not representative of site-specific meteorological conditions, and there is insufficient time to collect 5 years of onsite data, you might use 1 year of onsite meteorological data (consistent with GAQM) to complete the risk assessment. We recommend clearly identifying and discussing your choice of representative meteorological data with the appropriate parties (e.g. permitting authority or facility) prior to air modeling.

FIGURE 3-1
SOURCES OF METEOROLOGICAL DATA

Meteorological Data Processing - Government Sources

Meteorological Data Processing - Commercial Sources



Hourly data, also know as "surface data" because it tends to characterize meteorology nearer the surface, can be obtained from the National Climatic Data Center (NCDC) web site at address (http://lwf.ncdc.noaa.gov/oa/ncdc.html). We recommend data in the SAMSON format (available on CD-ROM). SAMSON data are available for 239 airports across the U.S. for the period of 1961 through 1990. SAMSON data contain all of the needed input parameters used by ISCST3 to compute concentration, dry and wet particle deposition, and dry and wet vapor deposition.

You could also get hourly surface data from NCDC in TD-3280 format, then reformat it to CD-144 format for input to MPRM. Precipitation data in TD-3240 format is also available from NCDC. TD-3240 formatted data is processed by MPRM to supplement the hourly surface data.

National Climatic Data Center
Federal Building
37 Battery Park Avenue
Asheville, NC 28801-2733

Customer Service: (704) 271-4871

File type:	File name:
Hourly precipitation amounts	NCDC TC-3240
Hourly surface observations with precipitation type	NCDC TD-3280
Hourly surface observations with precipitation type	NCDC SAMSON CD-ROM (Vol. I , II, and/or III)
Twice daily mixing heights from nearest station	NCDC TD-9689 (also available on SCRAM web site for 1984 through 1991)

Currently, MPRM is the most appropriate Agency meteorological preprocessor program for preparing the surface and upper air data into an ISCST3 meteorological input file. Most air modeling analyses will use MPRM to process the National Weather Service data.

We recommend using the most recent 5 years of complete meteorological data available on SAMSON (or more recent sources) for the air modeling. It's desirable, but not mandatory, that the 5 years be consecutive.

Each year of the 5 years of data should be complete before being processed by MPRM. If data gaps exist, we recommend filling in all missing data. The procedures we recommend for filling missing surface and upper air data are documented on the SCRAM web site under the meteorological data section. If the missing data are not addressed by the Agency objective procedures, then with the approval of the permitting authority, you can develop a subjective method for filling in missing data. If conditions occur such that:

1. missing values are not able to be replaced; and
2. the permitting authority approves the use of the meteorological data in that condition

then specify the `MSGPRO` keyword in the COntrol pathway of the ISCST3 input file. Note that the `DEFAULT` keyword can't be used with `MSGPRO`.

If you wish to use less than 5 years of meteorological data, we recommend clearly identifying and discussing this with all appropriate parties (e.g. permitting authority, or facility).

To prevent the need to repeat air modeling activities, we recommend that your choices of representative upper air and surface data be clearly identified and discussed with the appropriate parties (e.g. regulatory authority, facility) before you begin preprocessing and air modeling. Also, we recommend completely documenting all processing of meteorological data, including sources of data, selection criteria, consideration for precipitation amounts, preprocessor options selected, and filled missing data. If your choices of meteorological data (e.g. selection of upper air data, or determination of dry bulb temperature) are inconsistent with this guidance, we further recommend clearly identifying and discussing these choices in the risk plan and/or cost estimate (as appropriate) to ensure clarity and transparency of the final risk assessment results.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Electronic copy of the ISCST3 input code used to enter meteorological information
- Description of the selection criteria and process used to identify representative years used for meteorological data
- Identification of the 5 years of meteorological selected
- Summary of the procedures used to compensate for any missing data

3.5 METEOROLOGICAL PREPROCESSOR DATA NEEDS

After selecting the appropriate surface and upper air data using the procedures outlined in Section 3.4, you still need to put the data into a form that ISCST3 can use. As stated above, we recommend using the meteorological preprocessor MPRM to do this. The following Section describes the data MPRM itself requires in order to perform the preprocessing. Agency approval is recommended in the selection of MPRM parameter values.

We recommend preparing an ISCST3 meteorological file that can be used to calculate either concentration or deposition (what MPRM terms an “ISCGASW” file). All necessary parameters will then be available to ISCST3 for any subsequent concentration or deposition modeling. For example, if only the concentration option is selected in ISCST3 for a specific run, ISCST3 will ignore the precipitation values in the meteorological file. For subsequent air deposition modeling, ISCST3 will access the precipitation data from the same preprocessed meteorological file.

MPRM includes extensive QA/QC to check for values that are out of range. MPRM also checks for missing data and summarizes values that require editing to fill missing data. After a complete surface file passes the quality checks, it is processed with NCDC upper air data.

MPRM needs the following input parameters representative of the meteorological measurement site (typically the nearest representative National Weather Station):

- Minimum Monin-Obukhov length
- Anemometer height
- Surface roughness length (at measurement site)
- Noon-time albedo
- Bowen ratio
- Anthropogenic heat flux
- Fraction of net radiation absorbed at surface

MPRM also needs the following input parameter representative of the application site (e.g., source location):

- Surface roughness length (at application site)

The MPRM User's Guide contains detailed information for preparing the ISCST3 meteorological input file (U.S. EPA 1996j). The parameters listed are briefly described in the following subsections. These data are not included in the surface or mixing height data files obtained from the U.S. EPA or NCDC. We recommend taking special care, while using the tables in the MPRM User's Guide or reference literature, to select values representative of the meteorological measurement site and the site to be modeled. We recommend clearly identifying and discussing the selected values with the appropriate parties (e.g. permitting authority, or facility) before processing the meteorological data.

3.5.1 Monin-Obukhov Length

The Monin-Obukhov length (L) is a measure of atmospheric stability. It is negative during the day, when surface heating causes unstable air. It is positive at night, when the surface is cooled and the atmosphere more stable. In urban areas during stable conditions, the estimated value of L may not adequately reflect the less stable atmosphere associated with the mechanical mixing generated by buildings or structures. However, MPRM needs an input for minimum urban Monin-Obukhov length, even if the area to be analyzed by ISCST3 is rural. A nonzero value for L must be entered to run MPRM. We recommend using a value of 2.0 meters for L when the land use surrounding the site is rural (see Section 3.3.2.1). For urban areas, Hanna and Chang (1991) suggest that a minimum value of L be set for stable hours to simulate building-induced instability. The following are general examples of *L* values for various land use classifications:

TABLE 3-4
***L* VALUES FOR VARIOUS LAND USES**

Land Use Classification	Minimum *L*
Agricultural (open)	2 meters
Residential	25 meters
Compact residential/industrial	50 meters
Commercial (19 to 40-story buildings)	100 meters
Commercial (>40-story buildings)	150 meters

MPRM will use the minimum *L* value in calculating urban stability parameters. These urban values will be ignored by ISCST3 during the air modeling analyses for rural sites.

3.5.2 Anemometer Height

ISCST3 model results are very sensitive to small variations in wind speed. The height of the wind speed measurements is needed by ISCST3 to calculate wind speed at stack top. The wind sensor (anemometer) height for every National Weather Service station is identified in the station history section of the Local Climatological Data Summary available from NCDC. Since 1980, most National Weather Service stations measure wind speed at the height of 10 meters. However, some stations operate at other heights or have data at more than one height. We generally recommend verifying the correct measurement height for each year of data prior to processing with MPRM and running the ISCST3 model.

3.5.3 Surface Roughness Length at Measurement Site

Surface roughness length (or height) is a measure of the height of obstacles to wind flow. It is important in ISCST3 because it determines how close a particle must be to the ground before it is “captured” for deposition on the ground. Slight variations in surface roughness can lead to dramatic differences in ISCST3 results. For surface meteorological data from a National Weather Station, we typically recommend using a value of 0.10 meters for the “measurement site.” Surface roughness is proportional, but not equal, to the physical height of the obstacles. Table 3-3 (in Section 3.3.2.2) lists available roughness height values. These values are based on the general land use in the vicinity of the measurement site. We recommend considering these values in discussions with the appropriate parties prior to air modeling.

3.5.4 Surface Roughness Length at Application Site

MPRM also needs the surface roughness length at the facility (application site) in order to prepare the ISCST3 meteorological file. ISCST3 model results are very sensitive to the value used in MPRM for this parameter. Table 3-3 in Section 3.3.2.2 is also applicable to the application site. Compute a single surface roughness value representative of the site by using the method described in Section 3.3.2.2. We recommend clearly identifying and discussing the computed surface roughness length for the application site, along with maps or photographs illustrating land use, with the appropriate parties (e.g. permitting authority or facility) prior to use.

3.5.5 Noon-Time Albedo

"Noon-time albedo" is the fraction of the incoming solar radiation that is reflected from the ground when the sun is directly overhead. Albedo is used in calculating the hourly net heat balance at the surface. Net heat balance at the surface is then used in calculating hourly values of Monin-Obukhov length. MPRM automatically adjusts for the variation in albedo with solar elevation angle. Experience suggests that ISCST3 modeling results are not sensitive to the value selected for this parameter. Table 3-5 presents typical albedo values. Albedo values vary from 0.10 to 0.20 on water surfaces from summer to winter. Cultivated farmland values vary the most, from 0.14 during spring when land is tilled to expose dark earth, to 0.60 in winter when areas are snow-covered.

TABLE 3-5
ALBEDO OF NATURAL GROUND COVERS FOR LAND USE TYPES AND SEASONS

Land Use Type	Season[a]			
	Spring	Summer	Autumn	Winter
Water surface	0.12	0.10	0.14	0.20
Deciduous forest	0.12	0.12	0.12	0.50
Coniferous forest	0.12	0.12	0.12	0.35
Swamp	0.12	0.14	0.16	0.30
Cultivated land	0.14	0.20	0.18	0.60
Grassland	0.18	0.18	0.20	0.60
Urban	0.14	0.16	0.18	0.35
Desert shrubland	0.30	0.28	0.28	0.45

Source—Iqbal (1983)

[a] The various seasons are defined by Iqbal (1983) as follows:

Spring:	Periods when vegetation is emerging or partially green; this is a transitional situation that applies for 1 to 2 months after the last killing frost in spring.
Summer:	Periods when vegetation is lush and healthy; this is typical of mid-summer, but also of other seasons in which frost is less common.
Autumn:	Periods when freezing conditions are common, deciduous trees are leafless, crops are not yet planted or are already harvested (bare soil exposed), grass surfaces are brown, and no snow is present.
Winter:	Periods when surfaces are covered by snow and temperatures are below freezing. Winter albedo depends on whether a snow cover is present continuously, intermittently, or seldom. Albedo ranges from about 0.30 for bare snow cover to about 0.65 for continuous cover.

Based on the information in Table 3-5, rural area albedo estimates vary from 0.14 to 0.20 for cultivated land, and from 0.18 to 0.20 for grassland. For urban areas without snow, values vary from 0.14 to 0.18. For practical purposes, it is desirable to process a complete year of meteorological data with a single value for noon-time albedo. For example, the single value of 0.18 may be appropriate to process all meteorological data for an urban site. For rural sites, a single albedo value of 0.18 for grassland and cultivated land may be appropriate for areas without significant snow cover during winter months. For desert shrubland, a single value of 0.28 may be appropriate. A single value of 0.12 could be representative of forested areas. We recommend discussing the proposed values with the permitting authority prior to air modeling.

3.5.6 Bowen Ratio

The Bowen ratio is the ratio of the sensible heat flux to the evaporative or latent heat flux at the ground surface. The presence of moisture affects the heat balance through evaporative cooling, which, in turn, affects the hourly Monin-Obukhov length calculated by MPRM. Surface moisture is highly variable. Daytime Bowen ratios are presented in Table 3-6.

Bowen ratio values vary throughout the country. For example, in urban areas where annual rainfall is less than 20 inches, a single Bowen ratio value of 4.0 may be representative. For rural areas, a Bowen ratio value of 2.0 may be appropriate for grassland and cultivated land. For areas where annual rainfall is greater than 20 inches, we recommend a single Bowen ratio value of 2.0 for urban areas; and 0.7 for rural forests, grasslands, and cultivated lands. You can refine the Bowen ratio estimates provided in Table 3-6 by using site-specific precipitation and wind speed data. We recommend clearly identifying and discussing the proposed values with the appropriate parties (e.g. permitting authority or facility) prior to use.

TABLE 3-6

DAYTIME BOWEN RATIOS BY LAND USE, SEASON, AND PRECIPITATION CONDITIONS

Land Use	Season[a]			
	Spring	Summer	Autumn	Winter
Dry Conditions				
Water (fresh and salt)	0.1	0.1	0.1	2.0
Deciduous forest	1.5	0.6	2.0	2.0
Coniferous forest	1.5	0.6	1.5	2.0
Swamp	0.2	0.2	0.2	2.0
Cultivated land	1.0	1.5	2.0	2.0
Grassland	1.0	2.0	2.0	2.0
Urban	2.0	4.0	4.0	2.0
Desert shrubland	5.0	6.0	10.0	2.0
Average Conditions				
Water (fresh and salt)	0.1	0.1	0.1	1.5
Deciduous forest	0.7	0.3	1.0	1.5
Coniferous forest	0.7	0.3	0.8	1.5
Swamp	0.1	0.1	0.1	1.5
Cultivated land	0.3	0.5	0.7	1.5

TABLE 3-6
(contd.)

Land Use	**Season**[a]			
	Spring	**Summer**	**Autumn**	**Winter**
Grassland	0.4	0.8	1.0	1.5
Urban	1.0	2.0	2.0	1.5
Desert shrubland	3.0	4.0	6.0	6.0
Wet Conditions				
Water (fresh and salt)	0.1	0.1	0.1	0.3
Deciduous forest	0.3	0.2	0.4	0.5
Coniferous forest	0.3	0.2	0.3	0.3
Swamp	0.1	0.1	0.1	0.5
Cultivated land	0.2	0.3	0.4	0.5
Grassland	0.3	0.4	0.5	0.5
Urban	0.5	1.0	1.0	0.5
Desert shrubland	1.0	5.0	2.0	2.0

Source—Paine (1987)

[a] The various seasons are defined by Iqbal (1983) as follows:

Spring:	Periods when vegetation is emerging or partially green; this is a transitional situation that applies for 1 to 2 months after the last killing frost in spring.
Summer:	Periods when vegetation is lush and healthy; this is typical of mid-summer, but also of other seasons in which frost is less common.
Autumn:	Periods when freezing conditions are common, deciduous trees are leafless, crops are not yet planted or are already harvested (bare soil exposed), grass surfaces are brown, and no snow is present.
Winter:	Periods when surfaces are covered by snow and temperatures are below freezing. Winter albedo depends on whether a snow cover is present continuously, intermittently, or seldom. Albedo ranges from about 0.30 for bare snow cover to about 0.65 for continuous cover.

3.5.7 Anthropogenic Heat Flux

Anthropogenic heat is the surface heating caused by human activity, including automobiles and heating systems. It is used to calculate hourly *L* values (Monin-Obukhov lengths). Table 3-7 presents anthropogenic heat flux (Q_f) values for several urban areas around the world (U.S. EPA 1995g). In rural areas, we recommend using a value 0.0 Watts/m^2 for the Q_f. A value of 20.0 Watts/m^2 is appropriate for large urban areas, based on the annual value for Los Angeles.

TABLE 3-7

ANTHROPOGENIC HEAT FLUX (Q_f) AND NET RADIATION (Q_*) FOR SEVERAL URBAN AREAS

Urban Area (Latitude)	Population (Millions)	Population Density (Persons/km^2)	Per Capita Energy Use (MJ x 10^3/year)	Q_f (Watts/m^2) (Season)	Q_* (Watts/m^2)
Manhattan (40° North)	1.7	28,810	128	117 (Annual) 40 (Summer) 198 (Winter)	93 (Annual)
Montreal (45° North)	1.1	14,102	221	99 (Annual) 57 (Summer) 153 (Winter)	52 (Annual) 92 (Summer) 13 (Winter)
Budapest (47° North)	1.3	11,500	118	43 (Annual) 32 (Summer) 51 (Winter)	46 (Annual) 100 (Summer) -8 (Winter)
Sheffield (53° North)	0.5	10,420	58	19 (Annual)	56 (Annual)
West Berlin (52° North)	2.3	9,830	67	21 (Annual)	57 (Annual)
Vancouver (49° North)	0.6	5,360	112	19 (Annual) 15 (Summer) 23 (Winter)	57 (Annual) 107 (Summer) 6 (Winter)
Hong Kong (22° North)	3.9	3,730	34	4 (Annual)	110 (Annual)
Singapore (1° North)	2.1	3,700	25	3 (Annual)	110 (Annual)
Los Angeles (34° North)	7.0	2,000	331	21 (Annual)	108 (Annual)
Fairbanks (64° North)	0.03	810	740	19 (Annual)	18 (Annual)

Source—Oke (1978)

3.5.8 Fraction of Net Radiation Absorbed at the Ground

Also used for calculating hourly values of Monin-Obukhov length, fraction of net radiation absorbed at the ground is the last component of radiative heat balance. Based on:

- the net radiation (Q_*) values presented in Table 3-7, and
- recommendations presented in the MPRM User's Manual, themselves based on Oke (1982),

we recommend values of 0.15 for rural areas and 0.27 for urban areas (U.S. EPA 1995g).

3.6 ISCST3 MODEL INPUT FILES

The ISC3 User's Guide, Volume I (U.S. EPA 1995f) offers a thorough instruction on how to prepare the ISCST3 input files. The User's Guide is available for downloading from the SCRAM at http://www.epa.gov/scram001. We provide an example ISCST3 input file in Figure 3-2.
This example illustrates a single year run (1984), for particle phase COPC emissions from a single stack. The run is used to compute acute (1-hour average) and chronic (annual average) values. It provides single year results in a one hour and annual average plot file for post-processing. Specifying a terrain grid file in the TG pathway is optional. You generally only consider it for modeling dry vapor deposition in highly variable terrain. Each air modeling analysis has unique issues and concerns that we recommend you address in the risk assessment report. We recommend using an air modeling methodology consistently throughout, from data collection and model set-up, to model output. This will assist both you and the permitting authority in interpreting and communicating model results. A transparent and scientifically defensible risk assessment report will identify consistent methods while documenting each section of the ISCST3 input file.

FIGURE 3-2

EXAMPLE INPUT FILE FOR "PARTICLE PHASE"

```
CO STARTING
CO TITLEONE Example input file, particle phase run
CO TITLETWO 1984 met data, Baton Rouge Surface, Boothville Upper Air
CO MODELOPT DFAULT CONC DDEP WDEP DEPOS DRYDPLT WETDPLT RURAL
CO AVERTIME 1 ANNUAL
CO POLLUTID UNITY
CO TERRHGTS ELEV
CO RUNORNOT RUN
CO SAVEFILE 84SAVE1 5 84SAVE2
** Restart incomplete runs with INITFILE, changing '**' to 'CO'
** INITFILE 84SAVE1
CO FINISHED

SO STARTING
SO LOCATION STACK1 POINT 637524. 567789. 347.
SO SRCPARAM STACK1 1.0 23.0 447.0 14.7 1.9
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29
SO BUILDWID STACK1 14.02 15.51 16.53 17.05 17.05 16.53 15.51 14.03
SO BUILDWID STACK1 12.10 14.02 15.51 16.53 17.05 17.05 16.53 15.51
SO BUILDWID STACK1 14.02 12.10 14.02 15.51 16.53 17.05 17.05 16.53
SO BUILDWID STACK1 15.51 14.02 12.10 14.02 15.51 16.53 17.05 17.05
SO BUILDWID STACK1 16.53 15.51 14.02 12.10
SO PARTDIAM STACK1 0.35 0.70 1.10 2.00 3.60 5.50 8.10 12.5 15.0
SO MASSFRAX STACK1 0.22 0.08 0.08 0.11 0.10 0.07 0.10 0.11 0.13
SO PARTDENS STACK1 1.0  1.0  1.0  1.0  1.0  1.0  1.0  1.0  1.0
SO PARTSLIQ STACK1 7E-5 5E-5 6E-5 1.3E-4 2.6E-4 3.9E-4 5.2E-4 6.7E-4 6.7E-4
SO PARTSICE STACK1 2E-5 2E-5 2E-5   4E-5   9E-5 1.3E-4 1.7E-4 2.2E-4 2.2E-4
SO SRCGROUP ALL
SO FINISHED

RE STARTING
RE ELEVUNIT METERS
RE DISCCART 630000. 565000. 352.
RE DISCCART 630500. 565000. 365.
RE DISCCART 631000. 565000. 402.
      ...
      (ARRAY OF DISCRETE RECEPTORS)
      ...
RE DISCCART 635000. 570000. 387.
RE FINISHED

ME STARTING
ME INPUTFIL 84BTR.WET
ME ANEMHGHT 10.0
ME SURFDATA 13970 1984 BATON_ROUGE
ME UAIRDATA 12884 1984 BOOTHVILLE
ME FINISHED

TG STARTING
TG INPUTFIL TERRAIN.TER
TG LOCATION 0.0 0.0
TG ELEVUNIT METERS
TG FINISHED

OU STARTING
OU RECTABLE ALLAVE FIRST
OU PLOTFILE 1 ALL FIRST BTR841.PLT
OU PLOTFILE ANNUAL ALL BTR84A.PLT
OU FINISHED
```

As discussed in Section 3.2, ISCST3 requires separate runs to model vapor phase COPCs, particle phase COPCs, and particle-bound phase COPCs, for a total of three runs per COPC source (stack or fugitive). The ISCST3 "Control Secondary Keywords" used for these three runs are:

Vapor Phase:	`CONC DDEP WDEP DEPOS`
Particle Phase:	`CONC DDEP WDEP DEPOS`
Particle-Bound Phase:	`CONC DDEP WDEP DEPOS`

ISCST3 needs site-specific inputs for source parameters, receptor locations, meteorological data, and terrain features. Prepare the model for execution by creating an input file. The input file is structured in five (or six, if a terrain grid file is used) sections, or pathways, designated by two-letter abbreviations:

TABLE 3-8

ISCST3 INPUT FILE SECTIONS

Section	Abbreviation
Control	CO
Source	SO
Receptor	RE
Meteorology	ME
Terrain Grid (Optional)	*TG*
Output	OU

The following subsections describe how to specify the parameters for each pathway in the ISCST3 input file.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Electronic and hard copies of ISCST3 input file for all air modeling runs

3.6.1 COntrol Pathway

Your choice of model options (`MODELOPT`) in the COntrol pathway will direct ISCST3 in the types of computations to perform. We generally recommend specifying the `DFAULT` parameter for particle and particle-bound phase runs, so that ISCST3 will implement the following regulatory default options:

- Use stack-tip downwash (except for Schulman-Scire downwash).
- Use buoyancy-induced dispersion (except for Schulman-Scire downwash).
- Do not use final plume rise (except for building downwash).
- Use the calms processing routines.
- Use upper-bound concentration estimates for sources influenced by building downwash from super-squat buildings.
- Use default wind speed profile exponents.
- Use default vertical potential temperature gradients.

The `CONC` parameter specifies calculation of air concentrations. The `DDEP` and `WDEP` parameters specify dry and wet deposition. The `DEPOS` specifies computation of total (wet and dry) deposition flux. `DRYDPLT` and `WETDPLT` are used for plume depletion resulting from dry and wet removal. We recommend the following command lines for the vapor and two particle runs (these are for rural areas; substitute `URBAN` for urban areas):

Vapor:

```
CO MODELOPT TOXICS CONC DDEP WDEP DEPOS DRYDPLT
WETDPLT RURAL
CO GASDEPVD 0.01
```

Particle Phase:

```
CO MODELOPT DFAULT CONC DDEP WDEP DEPOS DRYDPLT
WETDPLT RURAL
```

Particle-Bound:

```
CO MODELOPT DFAULT CONC DDEP WDEP DEPOS DRYDPLT
WETDPLT RURAL
```

Note that for vapor phase model runs, the `DFAULT` option is replaced by the `TOXICS` option. This directs ISCST3 to execute the dry vapor deposition algorithm in addition to the regulatory default parameters. You need to add an additional COntrol command line for the vapor phase run, to provide the single value of the dry vapor deposition velocity. In order to set a single dry vapor deposition velocity in ISCST3, use the following options:

- Remove the DEFAULT keyword from the CO MODELOPT card. (This option is incompatible with the TOXICS option.)
- Add the TOXICS keyword on the CO MODELOPT card. (This option is needed when using dry vapor deposition.)
- Add the keyword CO GASDEPVD<Uservd>, where Uservd in the dry vapor deposition velocity (meters/second).

We recommend a dry vapor deposition velocity of 0.5 centimeter per second (cm/s) for organic contaminants, chlorine, and HCl. We recommend a dry vapor deposition velocity of 2.9 cm/s for divalent mercury. The recommended dry vapor deposition velocity value of 0.5 cm/s for organic contaminants is consistent with the range specified for pesticides (0.01 - 1.1 cm/s) and dioxins and furans (0.27 - 0.78 cm/s) (U.S. EPA 2000b). A recent review of dry deposition (Wesely and Hicks 2000) demonstrates considerable uncertainty about dry deposition even for well-measured species such as ozone and sulfur dioxide. Uncertainty is greater for organic compounds, with very few measurements available to support defensible values (see following table).

TABLE 3-9

DRY DEPOSITION VELOCITY ESTIMATES AVAILABLE IN LITERATURE

Chemical	Dry Deposition Velocity (cm/s)	Reference
Acetic acid	0.64 - 1.0	Hartmann et al. (1991)[1]
	0.7	Sanheuza et al. (1992)[1]
Formic acid	1.1	Sanheuza et al. (1992)[1]
Chlordane	0.01 - 0.04	Bidleman (1988)
Arochlor 1242	0.02 - 0.1	
Arochlor 1254	0.08 - 0.2	
p,p'-DDT	0.1 - 0.7	
TCDD	0.5	Trapp and Matthies (1995)
	0.5	McLachlan et al. (1995)
	0.27 - 0.78	Smith and Heinhold (1995)
PCDD/Fs	0.19 (0.06-0.60)	Koester and Hites (1992)

[1] As cited in Wesely and Hicks (2000)

The modeling reported in the Mercury Study Report to Congress (U.S. EPA 1997c) used a value of 0.3 cm/s for nighttime dry deposition of divalent mercury, but used daytime values ranging from 0.20 to 4.83 cm/s depending upon atmospheric stability and land-use category. The daytime values were based on data developed from nitric acid data, not from measurements of divalent mercury. U.S. EPA (1997c) used an average ISC model-calculated dry deposition velocity of 2.9 cm/s for divalent mercury vapor and 0.06 cm/s for elemental mercury. Higher values were expected for chemicals with greater reactivity than acetic acid or formic acid, but no measured values were identified for any organic compounds higher than 1.1 cm/s. As a result, we recommend the default of 2.9 for divalent mercury.

U.S. EPA (1997c) also calculates site- or contaminant-specific dry vapor deposition velocities based on various parameters including molecular diffusivity, a solubility enhancement factor, pollutant reactivity, mesophyll resistance, and Henry's law constant. If you are assessing a facility surrounded by land uses other than pine forest (for instance urban, agricultural lands, or wetlands), you may wish to consider how a site-specific value for this parameter could be calculated. Two important parameters are the stability class (from the air modeling) and the land use surrounding the facility. The following information can be found in Section 5.1.2.3 (Dry Deposition of Vapors), page 5-12, in the technical background document of the MACT rule: "*Human Health and Ecological Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Wastes: Background Document, Final Report (F-1999-RC2F-S0014)*", which explains that:

> To calculate the weighted dry deposition velocity, it is recommended that land use be obtained from 1:250,000 scale quadrangles of land use and GIRAS spatial data obtained from EPA website and placed in an ARC-INFO format (U.S. EPA, 1994b). Table B-6 of Appendix B of the MACT rule background document shows the land use data for the sites that were detailed in the MACT rule. In the MACT rule, the fraction of time in each stability class was based on 5-year hourly meteorological files used in the ISCST3 modeling. Table B-8 of Appendix B in the MACT rule background document shows the weighted dry deposition velocity for divalent mercury vapor at the modeled facilities. Dry deposition of elemental mercury was not included in the MACT analysis, which is consistent with the 1997 Mercury Report to Congress.
>
> Site-specific dry deposition velocity for divalent mercury vapor can be calculated by weighting land use and stability class. Values for dry deposition velocity for each land use category and stability class can be found in the "*Mercury Study Report to Congress Volume III: Fate and Transport of Mercury in the Environment, December 1997, EPA-452/R-97-003*" in Table 4-3 and Table 4-4, respectively. It is recommended that these values be averaged to annualized values. Generally, night time dry deposition velocities can be treated as constant across all stability.

As more research to measure deposition velocities of more organic compounds under a variety of conditions becomes available, it may be appropriate to develop site- and chemical- specific default dry deposition of vapor velocities. It is important to note that although we recommend inputting the dry vapor deposition velocity directly into ISCST3, you might instead elect to execute the dry gas deposition algorithms within ISCST3 to calculate a deposition velocity. However, we caution you to read Section 3.1.2 (Preprocessing Programs) and Section 3.4.6 (Solar Radiation), which note additional data needs and potential limitations to ISCST3 calculating deposition velocities. Having ISCST3 calculate a deposition velocity may also require that you conduct compound-specific air modeling runs, deviating from the unit emission rate approach as outlined in this guidance. This may significantly increase the number of air modeling runs needed.

Note that for each of the three runs for each emission source, 5 complete years of off-site (e.g., National Weather Service from SAMSON) meteorological data are used. For sites with meteorological data collected on-site, we recommend that the permitting authority be notified of the data period needed for a risk assessment. Specify '`ANNUAL`' for the averaging times (AVERTIME) to compute chronic (annual average) health risk, and/or '1' to compute acute health risks based on the maximum 1-hour average concentrations over the 5-year period (see Section 3.10). We generally recommend repeating each phase run five times (one for each year, or a total of 15 ISCST3 runs) to complete a set of 15 runs for the full five years of meteorological data.

Alternatively, you can combine the 5 years of meteorological data into a single meteorological data file and complete only 3 runs for each emission source (one run for each phase). Section 3.5.1.1 of the ISC3 User's Guide (U.S. EPA 1995f), includes a complete discussion of combining multiple years of meteorological data into a single file prior to running ISCST3. You may select the '`ANNUAL`' averaging time for all risk assessment runs, regardless of the number of years in the meteorological data file. The incorrect selection of '`PERIOD`' will not compute the correct deposition rates needed by the risk assessment equations (refer to Section 3.2.3 of the ISC3 User Guide, Volume I). No additional ISCST3 model execution time is needed to obtain 1-year or 5-year air modeling values.

ISCST3 also allows you to specify COPC half-life and decay coefficients. Unless clearly identified and discussed with the appropriate parties (regulatory authority or facility), we don't recommend using these keywords when conducting air modeling for risk assessments. You typically use the `TERRHGTS` keyword with the `ELEV` parameter to model terrain elevations at receptor grid nodes. The `FLAGPOLE` keyword specifies receptor grid nodes above local ground level and is not typically used for most air modeling to perform impacts at ground level.

We also recommend using `SAVEFIL` to restart ISCST3 in the event of a computer or power failure during long runs. `SAVEFIL` is best used by specifying two save files, each with a different name. We recommend that the save interval not be longer than 5 days for large runs. If two save files are used, and a failure occurs during writing to the save file, no more than 10 days will be lost. Use the `INITFILE` command to restart the runs after the failure, as shown in the following example:

```
CO SAVEFILE SAVE1 5 SAVE2
** INITFILE SAVE1
```

ISCST3 will save the results alternately to `SAVE1` and `SAVE2` every 5 days. If the run fails after successfully writing to `SAVE1`, you can restart the ISCST3 run by replacing the two asterisks (*) in the `INITFILE` line with `CO` and running ISCST3 again. The run will begin after the last day in `SAVE1`. Note that you don't use the `MULTYEAR` keyword for computing long-term averages, and it should not be specified.

The following is an example of the COntrol pathway computer code for a single-year ISCST3 particle run:

```
CO STARTING
CO TITLEONE Example input file, particle phase run, 1 year
CO TITLETWO 1984 met data, Baton Rouge Surface, Boothville Upper Air
CO MODELOPT DFAULT CONC DDEP WDEP DEPOS DRYDPLT WETDPLT RURAL
CO AVERTIME 1 ANNUAL
CO POLLUTID UNITY
CO TERRHGTS ELEV
CO RUNORRUN RUN
CO SAVEFILE 84SAVE1 5 84SAVE2
** Restart incomplete runs with INITFILE, changing '**' to 'CO'
** INITFILE SAVE1
CO FINISHED
```

The corresponding COntrol pathway computer code for a single-year ISCST3 vapor run is:

```
CO STARTING
CO TITLEONE Example input file, vapor phase run, 1 year
CO TITLETWO 1984 met data, Baton Rouge Surface, Boothville Upper Air
CO MODELOPT TOXICS CONC DDEP WDEP DEPOS DRYDPLT WETDPLT RURAL
CO GASDEPVD 0.01
CO AVERTIME 1 ANNUAL
CO POLLUTID UNITY
CO TERRHGTS ELEV
CO RUNORRUN RUN
CO SAVEFILE 84SAVE1 5 84SAVE2
** Restart incomplete runs with INITFILE, changing '**' to 'CO'
** INITFILE SAVE1
CO FINISHED
```

Additional runs for the other 4 years are set up with the same COntrol pathway, except for the title description and `SAVEFILE` filenames.

3.6.2 SOurce Pathway

As discussed in Section 3.8, ISCST3 normally uses a unit emission rate of 1.0 g/s. Additional source characteristics the model needs (typically found in the Part B permit application and trial burn report) include the following:

- Source type (point source for stack emissions; area or volume for fugitive emissions)
- Source location (UTM coordinates, m)
- Source base elevation
- Emission rate (1.0 g/s)
- Stack height (m)
- Stack gas temperature (K)
- Stack gas exit velocity (m/s)
- Stack inside diameter (m)
- Building heights and widths (m)
- Particle size distribution (percent)
- Particle density (g/cm^3)
- Particle and gas scavenging coefficients (unitless)

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Input values with supporting documentation for each parameter identified in Section 3.7.2

3.6.2.1 Source Location

The location keyword of the SOurce pathway `(SO LOCATION)` identifies the:

- source type,
- location, and
- base elevation.

The source type for any stack is referred to as a "point" source in ISCST3. Fugitive source emissions are discussed in section 3.9. The source location must be entered into ISCST3. We recommend entering source locations in UTM coordinates. The easterly and northerly coordinates are entered to the nearest meter; for example, 637524 meters UTM-E, or 4567789 meters UTM-N (no commas are used). Enter the base elevation of each stack in meters. Sources for base elevations include USGS topographic maps, facility plot plans or USGS digital data bases.

An example input for the location keyword on the SOurce pathway includes source type, location, and base elevation in the following format:

```
SO LOCATION STACK1 POINT 637524. 4567789. 347.
```

3.6.2.2 Source Parameters

The source parameters keyword of the SOurce pathway `(SO SRCPARAM)` identifies the:

- emission rate,
- stack height,
- stack temperature,
- stack velocity, and
- stack diameter.

Enter the unit emission rate as 1.0 g/s. Stack height is the height above plant base elevation on the `SO LOCATION` keyword. ***Stack gas exit temperature is the most critical stack parameter for influencing concentration and deposition.*** High stack gas temperatures result in high buoyant plume rise, which, in turn, lowers concentration and deposition rates. We recommend basing stack gas temperatures on stack sampling tests for existing stacks. For new or undefined stacks, you might use manufacturer's data for similar equipment. Calculate stack gas exit velocity from actual stack gas flow rates and stack diameter. Actual stack gas flow rates can be measured during existing stacks during stack sampling. You can get representative values for new or undefined sources from manufacturer's data on similar equipment. Stack diameter is the inside diameter of the stack at exit.

We highly recommend using a site- or unit-specific stack temperature (low, average, high). Similar to the Chapter 2 discussion on emissions testing, we recommend choosing the stack gas temperature based on the objectives of the risk assessment. For example, if permitting is the objective, 'typical' operating temperatures with all control devices operating might be a possible testing condition to select. On the other hand, you might select low or high operating temperatures because of the potential for emitting certain COPCs, or the production of specific PICs, regardless of plume rise considerations. We recommend avoiding the practice of always choosing the lowest stack temperature, in order to decrease buoyancy and thereby increase concentration and deposition near the source. This is a gross over-simplification of conditions. Increases in concentration and deposition near the source may underestimate concentrations and deposition rates away from the source where more critical receptors may be located (see Chapter 4).

Following is an example of the source parameter input in the SOurce pathway for emission rate (grams per second), stack height (meters), stack temperature (K), stack velocity (meters per second), and stack diameter (meters):

```
SO SRCPARAM STACK1 1.0 23.0 447.0 14.7 1.9
```

3.6.2.3 Building Parameters

The building height and width keywords of the SOurce pathway `(SO BUILDHGT; SO BUILDWID)` identify the building dimensions that most influence the air flow for each of the 36 10-degree directions surrounding a stack. Calculate the dimensions using BPIP software, as described in Section 3.2.4.

The BPIP output file is transferred into the ISCST3 SOurce pathway as follows:

```
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29

SO BUILDWID STACK1 14.02 15.51 16.53 17.05 17.05 16.53 15.51 14.03
SO BUILDWID STACK1 12.10 14.02 15.51 16.53 17.05 17.05 16.53 15.51
SO BUILDWID STACK1 14.02 12.10 14.02 15.51 16.53 17.05 17.05 16.53
SO BUILDWID STACK1 15.51 14.02 12.10 14.02 15.51 16.53 17.05 17.05
SO BUILDWID STACK1 16.53 15.51 14.02 12.10
```

3.6.2.4 Particle Size Distribution

ISCST3 needs a particle size distribution for determining deposition velocities. We recommend using site-specific stack test data for existing sources.

The following example is an ISCST3 input for a **particle phase** run. From Table 3-1, the distribution for 9 mean diameter sizes includes the data needed for the keywords of the SOurce pathway `(SO PARTDIAM; SO MASSFRAX)`. The `PARTDIAM` is taken from Column 1 (Mean Particle Diameter). The `MASSFRAX` is taken from Column 4 (Fraction of Total Mass).

```
SO PARTDIAM STACK1 0.35 0.70 1.10 2.00 3.60 5.50 8.10 12.5 15.0
SO MASSFRAX STACK1 0.22 0.08 0.08 0.11 0.10 0.07 0.10 0.11 0.13
```

The example for the ISCST3 input for the **particle-bound** run is described below. From Table 3-1, the `PARTDIAM` is the same. The `MASSFRAX` is taken from Column 6 (Fraction of Total Surface Area).

```
SO PARTDIAM STACK1 0.35 0.70 1.10 2.00 3.60 5.50 8.10 12.5 15.0
SO MASSFRAX STACK1 0.49 0.17 0.13 0.09 0.05 0.02 0.02 0.01 0.02
```

3.6.2.5 Particle Density

ISCST3 also needs particle density in order to model the air concentration and deposition rates of particles. We recommend determining site-specific measured data on particle density for all modeled sources when possible. For new or undefined sources requiring air modeling, we recommend using a default particle density of 1.0 g/cm^3. Particles from combustion sources, however, may have densities that are less than 1.0 g/cm^3 (U.S. EPA 1994d), which would reduce the modeled deposition flux.

Following is an example of the particle density portion of the SOurce pathway `(SO PARTDENS)` for the 9 mean particle size diameters of the previous example:

```
SO PARTDENS STACK1 1.0 1.0 1.0 1.0 1.0 1.0 1.0 1.0 1.0
```

3.6.2.6 Scavenging Coefficients

ISCST3 calculates wet deposition flux by multiplying a scavenging ratio times the vertically integrated concentration. The scavenging ratio is the product of a scavenging coefficient and a precipitation rate. Studies show that best fit values for the scavenging coefficients vary with particle size. For vapors, wet scavenging depends on the properties of the COPCs involved. However, not enough data are now available to adequately develop COPC-specific scavenging coefficients. Therefore, assume that vapors are scavenged at the rate of the smallest particles, with behavior in the atmosphere that is influenced more by the molecular processes that affect vapors than by the physical processes that may dominate the behavior of larger particles (U.S. EPA 1995f).

Wet deposition only occurs during precipitation. To use the wet deposition option in ISCST3, input scavenging coefficients for each particle size and a file that has hourly precipitation data. For wet deposition of vapors, we suggest using a scavenging coefficient for a 0.1-µm particle, to simulate wet scavenging of very small (molecular) particles. ISCST3 will also accept site-specific measured washout data, or a value calculated based on Henry's Law constant. If you choose an option other than the coefficient for a 0.1-µm particle, we recommend clearly identifying and discussing it with the appropriate parties (e.g. permitting authority, or facility) prior to analysis. You can establish scavenging coefficients for each particle size from the best fit of the curves presented in the ISC3 User's Guide (U.S. EPA 1995f). The curves are based on the work of Jindal and Heinhold (1991). The curves are limited to a maximum particle size of 10-µm. Assume that all scavenging coefficients for particle sizes greater than or equal to 10-µm are equal. This assumption follows research on wet scavenging of particles (Jindal and Heinhold 1991).

The ISCST3 model differentiates between frozen and liquid scavenging coefficients. Research on sulfate and nitrate data shows that frozen precipitation scavenging coefficients are about one-third of the values of liquid precipitation (Scire et al. 1990; Witby 1978). It is protective to assume that the frozen scavenging coefficients are equal to the liquid scavenging coefficients (Pei and Cramer 1986). If desired, you may input separate scavenging coefficients for frozen precipitation.

The following is an example of the particle liquid (rain) and frozen (sleet or snow) scavenging coefficients input in the SOurce pathway for 9 mean particle size diameters, assuming particles are scavenged by frozen precipitation at 1/3 the rate of liquid precipitation:

```
SO PARTSLIQ STACK1 7E-5 5E-5 6E-5 1.3E-4 2.6E-4 3.9E-4 5.2E-4 6.7E-4 6.7E-4
SO PARTSICE STACK1 2E-5 2E-5 2E-5   4E-5   9E-5 1.3E-4 1.7E-4 2.2E-4 2.2E-4
```

The complete SOurce pathway for the example ***particle phase*** input file is as follows:

```
SO STARTING
SO LOCATION STACK1 POINT 637524. 4567789. 347.
SO SRCPARAM STACK1 1.0 23.0 447.0 14.7 1.9
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29 18.29 18.29 18.29 18.29
SO BUILDHGT STACK1 18.29 18.29 18.29 18.29
SO BUILDWID STACK1 14.02 15.51 16.53 17.05 17.05 16.53 15.51 14.03
SO BUILDWID STACK1 12.10 14.02 15.51 16.53 17.05 17.05 16.53 15.51
SO BUILDWID STACK1 14.02 12.10 14.02 15.51 16.53 17.05 17.05 16.53
SO BUILDWID STACK1 15.51 14.02 12.10 14.02 15.51 16.53 17.05 17.05
SO BUILDWID STACK1 16.53 15.51 14.02 12.10
SO PARTDIAM STACK1 0.35 0.70 1.10 2.00 3.60 5.50 8.10 12.5 15.0
SO MASSFRAX STACK1 0.22 0.08 0.08 0.11 0.10 0.07 0.10 0.11 0.13
SO PARTDENS STACK1 1.0  1.0  1.0  1.0  1.0  1.0  1.0  1.0  1.0
SO PARTSLIQ STACK1 7E-5 5E-5 6E-5 1.3E-4 2.6E-4 3.9E-4 5.2E-4 6.7E-4 6.7E-4
SO PARTSICE STACK1 2E-5 2E-5 2E-5   4E-5   9E-5 1.3E-4 1.7E-4 2.2E-4 2.2E-4
SO SRCGROUP ALL
SO FINISHED
```

When modeling air ***vapors*** using ISCST3, the following is an example of the SOurce pathway input for wet vapor scavenging coefficients that replaces the PARTDIAM, MASSFRAX, PARTDENS, PARTSLIQ and PARTSICE lines in the above example:

```
        SO GAS-SCAV STACK1 LIQ 1.7E-4
        SO GAS-SCAV STACK1 ICE 0.6E-4
```

3.6.3 REceptor Pathway

The REceptor pathway identifies sets or arrays of receptor grid nodes (identified by UTM coordinates) for which ISCST3 generates estimates of air parameters including:

- air concentration,
- dry and wet deposition, and
- total deposition.

Previous U.S. EPA guidance (1994l) recommended using a polar receptor grid to identify maximum values. Polar grids provide coverage over large areas with fewer receptor grid nodes than other grid types, thereby reducing computer run times. However, U.S. EPA Region 6 experience indicates that, although polar grids may reduce computer run times, air modelers typically choose a different option. The benefit of reduced run time is offset by difficulties in identifying polar grid locations in absolute UTM coordinates for (1) extracting terrain values from digital terrain files, and (2) selecting exposure scenario locations (see Chapter 4). Using a Cartesian grid is an alternative to a polar grid.

One method of obtaining a Cartesian grid with terrain elevations is to open both the grid array and the USGS DEM file in a graphics program (e.g., SURFER®). The software samples the DEM file at the user-specified locations (i.e. the grid nodes), each defined as the intersection of east (x) and north (y) values. The software extracts terrain elevation (z) from the DEM file associated with the desired location. These x, y, and z values are saved as a text file with one receptor grid node per line. Use a text editor to prefix each line with "`RE DISCCART`," to specify a discrete receptor grid node in ISCST3 format. Commercial software is available that generates the recommended receptor grid node array ***and*** extracts terrain elevations from the USGS DEM downloaded files, or any terrain file in x-y-z format.

The following is an example of the REceptor pathway for discrete receptor grid nodes at 500-meter spacing that includes terrain elevations (in meters):

```
RE STARTING
RE ELEVUNIT METERS
RE DISCCART 630000. 3565000. 352.
RE DISCCART 630500. 3565000. 365.
RE DISCCART 631000. 3565000. 402.
       ↓
RE DISCCART 635000. 3570000. 387.
RE FINISHED
```

We recommend that air modeling for each risk assessment include, at a minimum, an array of receptor grid nodes covering the area within 10 kilometers of the facility. Locate the origin of the grid at the centroid of a polygon formed by the locations of the stack emission sources. This receptor grid node array may consist of a Cartesian grid with grid nodes spaced 100 meters apart extending from the centroid out to 3 kilometers. For the distances from 3 kilometers out to 10 kilometers, the receptor grid node spacing can be increased to 500 meters. The single grid node array contains both grid node spacings. Include the same receptor grid node array in the REceptor pathway for all ISCST3 runs for all years of meteorological data and for all emission sources.

We recommend specifying individual terrain elevations for all receptor grid nodes. Several methods are available for assigning terrain elevations to grid nodes using digital terrain data. The 1:250,000 scale DEM digital data are available for download at the USGS Internet site (http://edc.usgs.gov/guides/dem.html). This data has horizontal spacing between digital terrain values of approximately 90 meters, which provides sufficient accuracy for air modeling.

During the risk assessment, select air parameter (concentration and deposition) values for a single receptor grid node to evaluate a specific exposure scenario location. You can also compute an area average of air parameter values across multiple receptor grid nodes, to represent the average concentration or deposition over a watershed or water body (see Chapter 4). However, depending on site-specific considerations, a different receptor grid node array may be more appropriate.

In addition to the receptor grid node array evaluated out to 10 kilometers, you might consider additional grid node arrays for evaluating water bodies and their watersheds located beyond 10 kilometers. We recommend a grid node spacing of 500 meters between nodes for arrays that are more than 10 kilometers from the emission source. An equally spaced grid node array facilitates subsequent computation of area averages for deposition rates onto waterbodies and their associated watersheds.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Summary of all information regarding the coordinates and placement of the receptor grid node array used in air modeling
- Copies of any maps, figures, or aerial photographs used to develop the receptor grid node array
- Map presenting UTM locations of receptor grid nodes, along with other facility information.

3.6.4 MEteorological Pathway

The file containing meteorological data is specified in the MEteorological pathway. MPRM creates individual files for each of the 5 years as ASCII files. ISCST3 uses them to compute hourly concentrations and deposition rates. You can either specify a single year of meteorological data in each ISCST3 run, or combine the total period of meteorological data into a single meteorological file, which ISCST3 will process in a single 5-year run. When combining meteorological files, we caution you to consider the following:

- Preprocess each year separately using MPRM into an ASCII format
- Combine the years into a single file (using a text editor or DOS COPY command)
- ISCST3 compares the first line (header) of the combined file to the Surface and Upper Air Station ID numbers specified in the input file ME pathway
- ISCST3 reads the headers of subsequent years unless they're deleted in the combined file. If subsequent year headers are included in the combined file, ISCST3 will compare the station IDs to the input file station ID.

  Your analysis might use meteorological data from more than one surface station or upper air station (e.g., the upper air station is moved after the third year of the period and assigned a new station ID by the National Weather Service). If this is the case, delete the headers for subsequent years in the combined file.
- For sites where the anemometer height is changed during the 5 year period (e.g., during the period 1984-1988, the anemometer was relocated on December 15, 1985, changing the height from 20 feet to 10 meters), we recommend running each year separately. Use the anemometer height in the ISCST3 input file ME pathway which corresponds to that year's meteorological data.

We recommend completing each year in the file with a full year of data (365 days, or 366 days for leap years). Verify the anemometer height for the surface station from Local Climate Data Summary records, or other sources, such as the state climatologist office. We recommend identifying the correct anemometer height `ANEMHGHT` for the wind speed measurements at the surface station before air modeling. Details of specifying the meteorological data file are in the ISC3 User's Guide, Section 3.5.1.1.

The following is an example input section for the MEteorological pathway, using the 1984 Baton Rouge file, with an anemometer height of 10 meters and station identification numbers:

```
ME STARTING
ME INPUTFIL 84BR.WET
ME ANEMHGHT 10.0
ME SURFDATA 13970 1984 BATON_ROUGE
ME UAIRDATA 12884 1984 BOOTHVILLE
ME FINISHED
```

3.6.5 Terrain Grid (TG) Pathway

Computing dry plume depletion is sensitive to terrain elevation. In the absence of a terrain grid file, ISCST3 automatically assumes that the terrain slope between the stack base and the receptor grid node elevation is linear. In concept, this assumption may underestimate plume deposition. However, based on experience, the magnitude of the differences in computed concentrations and deposition rates is nominal. Since including a terrain grid file in the TG pathway significantly increases model execution time, we suggest that a terrain grid file may not be necessary for all sites. If a terrain grid file is desired for a specific site based on highly variable terrain over short distances, the format of the TG file is available in the ISC3 User's Guide.

The location keyword of the TG pathway (`TG LOCATION`) identifies the x and y values to be added to the source and receptor grid to align with the terrain file coordinates. If the source and receptor grid nodes are in relative units such that the source is at location 0,0, the location keywords in the TG pathway would be the UTM coordinates of the source. We recommend specifying all emission sources and receptor grid nodes in UTM coordinates (note NAD27 or NAD83 format), and that the TG file, if used, be in UTM coordinates. Therefore, the location of the origin of the TG file relative to the source location will be 0,0. Also, we recommend presenting the terrain elevations in the TG file in meters. Following is an example of the TG pathway:

```
TG STARTING
TG INPUTFIL TERRAIN.TER
TG LOCATION 0.0 0.0
TG ELEVUNIT METERS
TG FINISHED
```

3.6.6 OUtput Pathway

ISCST3 output comes in two basic file types: the mandatory “output summary file” type (mandatory because ISCST3 automatically creates the file), and numerous optional file formats. As part of its input file structure, ISCST3 requires the name of the output summary file (See Section 3.8.2). Using text and tables, this file summarizes the ISCST3 run results, including repeating back all input data (e.g., sources with parameters, control pathways, grid node list, model options selected), and results summaries for each annual and 1-hour averaging period. It also states if the run finished successfully. The summary file also alerts you to special meteorological conditions that were found during runtime (e.g. number of calm hours, and relationships between source base elevation and grid node elevations). It provides impacts for the highest grid node values, not the results at every grid node required to perform the methods found in the remainder of the HHRAP. We recommend using the output summary file for quality checking, to make sure the run executed successfully and correctly.

ISCST3 provides numerous optional output file formats in addition to the results in the output summary file, as specified in receptor tables (RECTABLE). The “plot files” format is the most useful option for facilitating post-processing of the air parameter values in the model output. For this reason, though “plot files” are optional in executing ISCST3, they are necessary for HHRAP methods. There are two plot files for each ISCST3 run on a single source – the ‘annual’ and ‘1-hour’ plot files. The 'annual' plot file contains all air modeling results at each grid node, for the annual average of five years of met data. The '1-hour' plot file contains the air modeling results at each grid node for the 'highest 1-hour' impact for all five years. The file presents data in tabular form, with impacts at every grid node.

The plot file lists the x and y coordinates, and the concentration or deposition rate values for each averaging period. Data are listed in a format that can easily be pulled into a post-processing program (e.g. spreadsheet). Note that the ISCST3-generated ***plot*** file is not the same format as the ISCST3-generated ***post*** file. We recommend using the plot file, not the post file.

Following is an example OUtput file specification for single-year run of 1-hour and annual average plot files:

```
OU STARTING
OU RECTABLE ALLAVE FIRST
OU PLOTFILE 1 ALL FIRST BTR841.PLT
OU PLOTFILE ANNUAL ALL BTR84A.PLT
OU FINISHED
```

The second line directs ISCST3 to create a table of values for each receptor grid node for all averaging periods in the model run (1-hour and annual). The third line directs ISCST3 to create a separate plot file (named BTTR841.PLT) of the 1-hour average results for all emission sources in the run using the first highest (e.g., maximum) value for all hours of results during the year for each receptor grid node. The fourth line directs ISCST3 to create another separate plot file (named BTR84A.PLT) of the annual average results for all sources in the run for each receptor grid node.

3.7 ISCST3 MODEL EXECUTION

Consider model execution time needs for each analysis. A complete air modeling run—including air concentration, wet and dry deposition, and plume depletion—may require 10 times the run time for the same source and receptor grid nodes for air concentration only. You can avoid wasted modeling effort and analysis time by verifying input parameters and conducting test runs prior to executing ISCST3.

Long run times result mainly from two algorithms—plume depletion and terrain grid file. ISCST3 run times increase as much as tenfold for runs applying plume depletion. We believe that constituent mass is conserved between suspended concentration and deposition rate by allowing for depletion of deposited mass from the plume concentration in ISCST3. The overestimate of plume concentration, and the subsequent overestimate of deposition which results when plume depletion is not allowed, is too protective. At the same time, we don't believe the nominal benefits of including a terrain grid file justify the added run times. Therefore, we recommend that plume depletion always be included. We don't recommend terrain grid files.

3.8 USING MODEL OUTPUT

ISCST3 output (air concentrations and deposition rates) are usually provided on a unit emission rate (1.0 g/s) basis from the combustor or emission source, and aren't COPC-specific. This is to preclude having to run the model for each individual COPC. Use the COPC-specific emission rates from the trial burn (see Chapter 2) to adjust the unitized concentration and deposition output from ISCST3. The resulting COPC-specific air concentrations and deposition rates will be used in the estimating media concentration equations (see Chapter 5). Concentration and deposition are directly proportional to the unit emission rate used in the ISCST3 modeling. However, there might be some instances where the risk assessor will need to convert modeled output to COPC-specific output for the risk assessment. For example, the risk assessor may want to compare modeled COPC concentrations in ambient media to concentrations actually measured in the field.

For facilities with multiple stacks or other emission sources, we recommend that each source be modeled separately. This is because modeling multiple sources at one time results in an inability to estimate source-specific risks. This limits the ability of a permitting agency to evaluate which source is responsible for the resulting risks. Such ambiguity makes it impossible for the agency to specify protective, combustion unit-specific permit limits. If a facility has two or more sources with identical characteristics (emissions, stack parameters, and nearby locations), it may be appropriate to model them with a single set of runs. We recommend getting the approval of the permitting authority prior to such multi-source modeling.

TABLE 3-10

ISCST3 AIR PARAMETER OUTPUT

Air Parameter	Description	Units
(Used for most soil-based exposure pathways)		
Cyv	Unitized yearly average air concentration from vapor phase	μg-s/g-m^3
Cyp	Unitized yearly average air concentration from particle phase	μg-s/g-m^3
Dydv	Unitized yearly average dry deposition from vapor phase	s/m^2-yr
Dywv	Unitized yearly average wet deposition from vapor phase	s/m^2-yr
Dydp	Unitized yearly average dry deposition from particle phase	s/m^2-yr
Dywp	Unitized yearly average wet deposition from particle phase	s/m^2-yr
(Used for fish and drinking water ingestion exposure pathways)		
Cywv	Unitized yearly (water body or watershed) average air concentration from vapor phase	μg-s/g-m^3
Dytwv	Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase	s/m^2-yr
Dytwp	Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase	s/m^2-yr
(Used for evaluation of acute risk via direct inhalation exposure pathway)		
Chv	Unitized hourly air concentration from vapor phase	μg-s/g-m^3
Chp	Unitized hourly air concentration from particle phase	μg-s/g-m^3
Chpb	Unitized hourly air concentration from particle-bound phase	μg-s/g-m^3

3.8.1 Unit Rate Output vs. COPC-Specific Output

The relationship between the unit emission rate and the unit air parameter values (air concentrations and deposition rates) is linear. Similarly, the relationship between the COPC-specific emission rate (Q) and the COPC-specific air parameter values (air concentrations and deposition rates) is also linear ***if*** the COPC-specific emission rate is used in the air model.

In this Section we discuss using the unit emission rate. We also advance the rationale that you should use a unit emission rate instead of the COPC-specific emission rate. Using a unit emission rate precludes having to run ISCST3 separately for each individual COPC. We advocate using a unit emission rate in the air modeling because you can develop a common ratio relationship between the unit emission rate and the COPC-specific emission rate. The ratio is based on the fact that both individual relationships are linear in the air model. This ratio relationship is expressed by the following equation:

$$\frac{\text{COPC}-\text{Specific Air Concentration}}{\text{COPC}-\text{Specific Emission Rate}} = \frac{\text{Modeled Output Air Concentration}}{\text{Unit Emission Rate}} \qquad \text{Equation 3-2}$$

To use this equation, you must know three of the variables. ISCST3 provides the modeled output air concentration (or deposition rate). The unit emission is 1.0 g/s. You can get the COPC-specific emission rate directly from stack or source test data.

3.8.1.1 Determining the COPC-Specific Emission Rate (*Q*)

The COPC-specific emission rate from the stack, prior to any applicable adjustments (e.g., upset emissions scaling, revisions based on hours of operation, feed rate adjustments, etc.) is a function of the stack gas flow rate and the stack gas concentration of each COPC. It can be calculated from the following equation:

$$Q = SGF \cdot \frac{SGC \cdot CFO_2}{1 \times 10^6} \qquad \text{Equation 3-3}$$

where

Q = COPC-specific emission rate (g/s)
SGF = Stack gas flow rate at dry standard conditions (dscm/s)

SGC	=	COPC stack gas concentration at 7 percent O_2 as measured in the trial burn (µg/dscm)
CFO_2	=	Correction factor for conversion to actual stack gas concentration O_2 (unitless)
1×10^6	=	Unit conversion factor (µg/g)

Guidance for adjusting COPC-specific emission rates, as well as determining emission rates for fugitive emission sources, is available in Chapter 2. Also, it is sometimes necessary to derive the COPC-specific emission rate from surrogate data, such as for a new facility that has not yet been constructed and trial burned (see Chapter 2).

3.8.1.2 Converting Unit Output to COPC-Specific Output

Once three of the four variables in Equation 3-1 are known, you can derive the COPC-specific air concentrations and deposition rates by multiplying as follows:

$$\begin{matrix}\text{COPC} - \text{Specific} \\ \text{Air Concentration}\end{matrix} = \frac{\text{Modeled Output Air Concentration} \cdot \text{COPC} - \text{Specific Emission Rate}}{\text{Unit Emission Rate}} \qquad \text{Equation 3-4}$$

For example, if COPC A is emitted at a rate of 0.25 g/s, and the ISCST3 modeled concentration at a specific receptor grid node is 0.2 $\mu g/m^3$ per the 1.0 g/s unit emission rate, the concentration of COPC A at that receptor grid node is 0.05 $\mu g/m^3$ (0.25 multiplied by 0.2). Calculating deposition is similarly proportional to the emission rate of each COPC. You are reminded once again that the process of converting modeled unitized output into COPC-specific output is taken directly into account in the estimating media concentration equations in Chapter 5 and Appendix E.

3.8.2 ISCST3 Model Output

It is important that the risk assessor understand how to read the ISCST3 output structure, in order to ensure accurate use of modeled output in the risk assessment. The output from each ISCST3 model run is written to two separate file formats. The ‘output summary file’ is specified by name at run time in the execution command. Typical command line nomenclature is:

```
ISCST3 inputfile.INP outputfile.OUT
```

where

`ISCST3`: specifies executing the ISCST3 model
`inputfile.INP`: is the input file name you (the modeler) select
`outputfile.OUT`: is the output summary file name you select, typically the same as the input file name (but with a different suffix)

For example, the following ISCST3 input line would run the input file (PART84R.INP) created by the modeler for particulate emissions using 1984 meteorological data on the receptor grid. ISCST3 will automatically write the output summary file (PART84R.OUT) during execution.

```
ISCST3 PART84R.INP PART84R.OUT
```

You specify the second file format, the output ‘plot file,’ in the ISCST3 input file OUtput pathway. ISCST3 creates it during the run (see Section 3.6.6). The “plot file” is typically then imported into a post-processing program (e.g. a spreadsheet) before entry into the risk assessment computations.

Vapor phase and Particle-bound phase runs produce similar plot files. The plot files for the vapor phase runs will include average concentrations and wet deposition rates. The plot files for the particle and particle-bound phase runs will include average concentrations, dry depositions, wet depositions and total depositions. You can average the 1-year values at each receptor grid node for a 5-year value at each node, unless a single five-year ISCST3 run using a combined meteorological file is used. If you use the 5-year combined file, the risk assessor can use the results from the ISCST3 plot file directly in the risk assessment without averaging over the five years. All values are used in the estimating media concentration equations (see Chapter 5).

3.8.3 Using Model Output to Estimate Media Concentrations

Section 3.2 discussed how partitioning of the COPCs affects the development of ISCST3 modeling runs. The choices of which modeled air parameter values (air concentrations and deposition rates) to use in the estimating media concentration equations is based on this same partitioning theory.

3.8.3.1 Vapor Phase COPCs

ISCST3 generates the following output for vapor phase air modeling runs:

- vapor phase air concentrations (unitized *Cyv* and unitized *Cywv*),
- dry vapor deposition (unitized *Dydv*), and
- wet vapor depositions (unitized *Dywv* and unitized *Dywwv*)

for organic COPCs at receptor grid nodes based on the unit emission rate. These values are used in the estimating media concentration equations for all COPC organics except the polycyclic aromatic hydrocarbons (PAHs) dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene[2]. The air concentration (unitized *Cyv*), dry vapor deposition (unitized *Dydv*), and wet vapor deposition (unitized *Dywv*) from the vapor phase run is also used in the estimating media concentration equations for mercury. You select values for these COPCs from the vapor phase run because the mass of the COPC emitted by the combustor is assumed to have either all or a portion of its mass in the vapor phase (see Appendix A-3).

3.8.3.2 Particle Phase COPCs

ISCST3 generates the following output for particle phase air modeling runs:

- air concentration (unitized *Cyp*),
- dry deposition (unitized *Dydp*),
- wet deposition (unitized *Dywp*), and
- combined deposition (unitized *Dytwp*)

for inorganics and relatively non-volatile organic COPCs at receptor grid nodes based on the unit emission rate. These values are used in the estimating media concentration equations for all COPC inorganics (except mercury, see Chapter 2 and Appendix A-3) and polycyclic aromatic hydrocarbons with fraction of vapor phase, Fv , less than 0.05 (e.g., dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene). You choose values for inorganic and relatively non-volatile COPCs from the particle phase run because the all the mass of the COPC is assumed to be in the particulate phase (see Appendix A-3). The mass is apportioned across the particle size distribution based on mass weighting.

---

[2] These two PAHs have vapor phase fractions, Fv, less than five percent.

3.8.3.3 Particle-Bound COPCs

ISCST3 particle-bound runs generate the following output:

- air concentration (unitized *Cyp*),
- dry deposition (unitized *Dydp*),
- wet deposition (unitized *Dywp*), and
- combined deposition (unitized *Dytwp*)

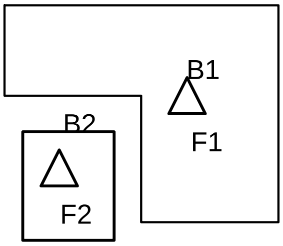
for organic COPCs and mercury (see Chapter 2 and Appendix A-3) at each receptor grid node. Use these values in the "estimating media concentration" equations to account for the portion of the vapor condensed onto particulate surfaces. Select values for these COPCs from the particle-bound run because the mass of the COPC emitted by the combustor is assumed to have a portion of its mass condensed on particulates (see Appendix A-3). ISCST3 uses surface area weighting to apportion the particle-bound mass across the particle size distribution.

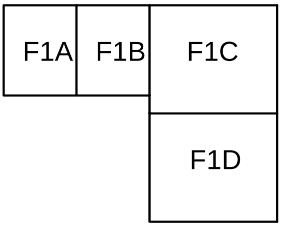
3.9 MODELING FUGITIVE EMISSIONS

The procedures presented in this chapter for modeling stack source emissions are also effective for modeling fugitive source emissions, as defined in Chapter 2. However, you can represent fugitive emissions in the ISCST3 input file SOurce pathway as either "area" or "volume" source types. Model fugitive emissions of volatile organics only in the vapor phase. Model fugitive emissions of ash only in the particle and particle-bound phases, not vapor phase.

As discussed in Chapter 2, fugitive emissions of volatile organic vapors are associated with combustors that include storage vessels, pipes, valves, seals and flanges. Enter the horizontal area of the fugitive source into the ISCST3 input file according to the instructions found in the ISC3 User's Guide, Volume I (U.S. EPA 1995f). You can get the horizontal area from the facility plot plan. The height of the fugitive source is defined as the top of the vertical extent of the equipment. If the vertical extent isn't known, you can use a default height of ground level (release height of zero). This provides a protective estimate of potential impacts. The ISCST3 model run time is faster for volume source types than for area source types. We generally recommend considering the volume source type for most applications. We recommend following the methods in the ISCST3 User’s Guide to define the input parameters representing the fugitive source.

The following example is for organic fugitive emissions only (modeling only vapor phase emissions) modeled as a volume source type. The example includes a facility with two stack emission sources (B1, B2) and two fugitive emission sources (areas F1, F2). A total of four runs for each year (or 5-year combined file) of meteorological data is needed.


Plot Plan


ISC3 Volume

Perform one run for each of the two stacks as point sources. Perform one run for each of the two fugitive areas as volume sources (Note: or you could model as area sources). Since the emissions are fugitive volatile organics, model only the vapor phase. The vertical extent of the pipes, valves, tanks and flanges associated with each fugitive emission area is 15 feet (about 5 meters) above plant elevation. To define the sources for input to ISCST3, specify the release height as 2.5 meters (½ of vertical extent of fugitive emissions). Specify the initial vertical dimension as 1.16 meters (vertical extent of 5 meters divided by 4.3, as described in the ISC3 User’s Guide).

The initial horizontal dimension is the side length of the square fugitive area (footprint) divided by 4.3. If fugitive area F2 has a measured side of 30 meters, the initial horizontal dimension is 6.98 (30 meters divided by 4.3). For fugitive area F1, subdivide the area on the plot plan (ISC3 Volume) to create square areas for input to ISCST3. The four areas depicted represent subdivision into square areas. Fugitive source F1 is input into a single ISCST3 run as four separate volume sources (F1A, F1B, F1C, F1D). The initial horizontal dimension for each volume source is the side of the square divided by 4.3.

It is very important to allocate the unit emission rate (1.0 gram per second) proportionately among the subdivided areas. For example, if the areas of the subdivided squares in the ISC3 Volume figure results in F1A equal to F1B each with 1/8th the total area, the proportion of the unit emissions allocated to each of these volume sources is 0.125 grams per second. The remaining two areas are each 3/8ths of the total area of fugitive F1, so that 0.375 grams per second is specified for the emission rate from each source. The total emissions for the four volume sources sum to the unit emission rate for the F1 fugitive source (0.125 + 0.125 + 0.375 + 0.375 = 1.0 g/s).

If you tell ISCST3 to include all sources in the model results (SO SRCGROUP ALL), the model will appropriately combine all four volume source subdivisions into combined impact results for fugitive source F1. You can use the resulting air parameter values in the plot files directly in the risk assessment equations, the same as if the emissions were modeled as a single point source. The initial vertical dimension is defined the same as F2, using the vertical extent of 5 meters divided by 4.3 and a release height of 2.5 meters (½ vertical extent). For volume sources, the location is specified by the *x* and *y* coordinates of the center of each square area.

The COntrol parameters can follow the recommendations for setting up a vapor phase computation.

```
CO CONC WDEP
```

Fugitive emissions of ash particles are from the storage piles associated with combustion units. Enter the horizontal area of the storage pile into the ISCST3 input file according to the ISCST3 User’s Guide, Volume I (U.S. EPA 1995f). The height of emissions is the top of the pile. If you don’t know the vertical extent, you could use ground level (or zero height). You’ll typically model fugitive ash as area source type. However, the permitting authority might consider volume source type prior to air modeling. We generally recommend following the methods in the ISCST3 User’s Guide when defining the input parameters to represent the ash release as an area source.

We recommend the COntrol parameters follow the recommendations for setting up a particulate phase computation.

```
CO CONC DDEP WDEP DEPOS
```

We recommend documenting the emissions characterization and source type.

3.10 MODELING ACUTE RISK

It might be important for you to evaluate the acute, or short-term, effects due to the direct inhalation of vapor phase, particle phase and particle-bound phase COPCs in the risk assessment. We recommend considering site- or unit-specific factors such as unit operating conditions during emissions testing, emission rates to be used in the acute screen, exposure scenario locations, etc. (see Sections 2.2.1, 4.2, 4.3, and 7.5) prior to conducting air modeling for an acute evaluation.

Since only ambient air concentrations are included in the direct inhalation pathway, You can compute the air parameters needed for a screening acute assessment in the same ISCST3 runs (i.e., same input values for meteorological data, stack gas exit temperature and velocity, etc.) that compute the air parameters for the long-term chronic effects. However, as with chronic modeling, a non-steady state dispersion model (e.g., CALPUFF) might provide a more representative estimate of concentration and deposition. This is because of differences in methodology for modeling time-varying emissions and time- and space-varying meteorological fields.

Methods outlined in this section focus on supporting a screening type acute assessment using the existing air modeling runs executed for the chronic risk assessment. ***However***, it is important to note that while this approach provides some obvious efficiencies with regard to air modeling, ***you might need to execute separate air modeling runs***. For example, you might consider site- or unit-specific characteristics (i.e., conditions of most concern for a short term or acute release scenario) in order to provide the most protective or appropriate acute assessment. As discussed in Section 3.1, site-specific conditions and assessment objectives might result in your choosing a different air model for the acute portion than for the chronic assessment.

In air modeling for an acute type assessment, the goal is typically to compute the highest 1-hour average air concentration for each phase (particle-bound, etc) for each source, for the entire period of analysis. In most cases, this period is the five years of meteorological data. For ISCST3 to identify the highest one-hour average concentration at each grid node, we recommend making two specifications in the ISCST3 input files.

First, you must specify the 1-hour average as one of the averaging times in the COntrol pathway. The example of this specification is included in Section 3.6.1. The ISCST3 input file includes:

```
CO AVERTIME 1 ANNUAL
```

where “1" specifies the 1-hour averaging time ISCST3 computes for each hour of meteorological data.

Remember that the ‘`ANNUAL`’ is specified for the chronic effects.

Second, we recommend that the OUtput pathway include the specification of a 1-hour average plot file which contains the highest 1-hour concentration at each receptor grid node for all the hours evaluated in the ISCST3 run. For a single year run, the 1-hour concentration reported in the plot file is the highest value computed at each x, y grid node location for the total hours in the year (8760, or 8784 for leap years). The acute risk assessment needs the highest 1-hour concentration for the 5-year period. We recommend reviewing each of the five single year values to identify the highest for all five years at each receptor grid node. However, if you run a combined 5-year meteorological input file, the resulting plot file will already identify the highest value for the 5-year period at each grid node with no additional processing needed. The OUtput pathway instructions to create the plot file for one-hour average concentrations are:

```
OU PLOTFILE 1 ALL FIRST BTR841.PLT
```

where

1:	specifies the 1-hour averaging period,
ALL:	instructs ISCST3 to include all sources in the run,
FIRST:	instructs the model to include only the first highest value at each receptor grid node, and
BTR841.PLT:	is the name for the plot file.

The plot file name is unique for the run and the one-hour averaging period results.

The highest air concentration for the 1-hour averaging period is input as the air parameters, *Chv*, *Chp,* and *Chpb* in the acute risk assessment equations (see Section 7.5 and Appendix B, Table B-6-1).

Chapter 4
Exposure Scenario Identification

What's Covered in Chapter 4:

4.1 Characterizing the Exposure Setting

4.2 Recommended Exposure Scenarios
  - Farmer
  - Farmer Child
  - Resident
  - Resident Child
  - Fisher
  - Fisher Child
  - Acute Receptor

4.3 Selecting Exposure Scenario Locations

The purpose of this chapter is to provide guidance on identifying "exposure scenarios" to evaluate in the risk assessment. Evaluating exposure scenarios will estimate the type and magnitude of human exposure to COPC emissions from hazardous waste combustors (including fugitive emissions). In this document, identifying exposure scenarios consists of :

- characterizing the exposure setting,
- identifying recommended exposure scenarios, and
- selecting exposure scenario locations.

Characterizing the exposure setting includes defining the dimensions of the assessment area (or "study area"). It also includes identifying the current and potential human activities and land uses within those boundaries. Within the context of the exposure setting, an exposure scenario is a combination of "exposure pathways" to which a "receptor" may be subjected.

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak in this chapter is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

For this guidance, we define a receptor as a human being potentially exposed to COPCs emitted to the atmosphere from a hazardous waste combustion facility. An exposure "route" is the particular means of entry into the body. For the purposes of the HHRAP, receptors come into contact with COPCs via two primary exposure routes: either directly—via inhalation; or indirectly—via COPC deposition and subsequent ingestion of water, soil, vegetation, and animals that have been contaminated by COPCs through the food chain.

An exposure "pathway" is the course a chemical takes from its source to the person being exposed. An exposure pathway consists of four fundamental components:

1. a source and mechanism of COPC release (see Chapter 2);
2. a retention medium, or a transport mechanism and subsequent retention medium in cases involving media transfer of COPCs (see Chapter 3 for air transport of COPCs, and Chapter 5 for bioaccumulation of COPCs in the food chain);
3. a point of potential human contact with the contaminated medium; and
4. an exposure route.

Exposure to COPCs can occur via numerous exposure pathways, such as ingestion of diary products and home grown produce (see Section 4.2).

The HHRAP identifies a number of generic exposure scenarios (Farmer, Farmer Child; Fisher, Fisher Child; Resident, and Resident Child). Used as presented, these standardized scenarios should be reproducible across most sites and land use areas. We intend these scenarios to be appropriate for a broad range of situations, rather than to represent actual scenarios. We believe that the recommended exposure scenarios and associated assumptions are reasonable. They represent a scientifically sound approach that is protective of human health and the environment, while recognizing the uncertainties associated with evaluating real world exposures. For example, the scenarios are designed with a level of protectiveness intended to address potential receptors not directly evaluated, such as populations with somewhat higher exposures than the general public. At the same time, you can easily alter these scenarios to more closely reflect site-specific conditions. To be transparent, we recommend well-documenting, supporting and discussing any changes (i.e. deletions, additions, or modifications) to a recommended exposure scenario or scenario location with the appropriate parties (regulatory agency, facility, interested community members).

Selecting exposure scenario locations involves identifying the physical positions of the exposure scenarios within the study area. For example, you can position scenarios based on current or future human activities or land use. Alternatively, you can position scenarios artificially, as part of a screening assessment. You could, for example, locate all selected receptors at the area of greatest contaminant deposition, to maximize potential exposure. The HHRAP focuses on placement based on actual or potential activities and land use.

The following sections describe how we recommend

1. characterizing the exposure setting,
2. identifying which of the recommended exposure scenarios are appropriate for the risk assessment, and
3. selecting the exposure scenario locations.

4.1 CHARACTERIZING THE EXPOSURE SETTING

The purpose of characterizing the exposure setting is to identify the human receptors, their land uses and activities, which might be impacted by exposure to emissions from the facility being assessed. The exposure setting might include multiple sources (e.g., multiple stacks, fugitive emissions), as well as terrain both inside and outside the facility boundary (or “fenceline”). We believe both **current** and **reasonable potential** human activities or land uses are relevant, when determining which recommended exposure scenarios are appropriate for the risk assessment (see Section 4.2).

Experience has shown us that most significant deposition occurs within a 10 km radius, as measured from the centroid of a polygon centered on the stacks of the facility being assessed. Consequently, resources for characterizing the exposure setting might initially be focused within this area. Also, most recommended exposure scenarios appropriate for the assessment will likely be located within this area. It may be prudent, however, to also characterize the exposure setting beyond the 10 km radius, to determine if conditions exist which warrant additional exposure scenarios. Such conditions might include (but are not limited to) recommended exposure scenarios or special populations (see Section 4.1.3) not found within the 10 km area, or topographic features - such as hills - that tend to increase potential deposition. A 50 km radius is the recognized limit of the ISCST3 air dispersion model, and can be used as the outer boundary for characterizing additional exposure settings (See Chapter 3 for information on air modeling beyond a 50 km radius). All affected parties (i.e. regulators, facilities, interested community members) can then discuss if additional scenarios need to be assessed, and if so, their locations.

The study area might include land use and water bodies both inside and outside the facility fenceline. It's important to understand that some of the recommended scenarios might most appropriately be placed within facility boundaries. For example, some facilities located on substantial property rent portions of the property to the public for farming, ranching, or recreational purposes (e.g., fishing).

When characterizing the exposure setting, we highly recommend considering

- current and reasonable potential future land use,
- waterbodies and their associated watersheds, and
- special populations.

The following subsections provide information on these aspects.

4.1.1 Current and Reasonable Potential Future Land Use

Land use is an important factor in characterizing the exposure setting. When land use is overlaid with the air dispersion modeling results, the combination will demonstrate which recommended exposure scenarios (and their locations) are most relevant for the risk assessment. We recommend considering both current and reasonable potential future land use (i.e. "future land use"), because risk assessments typically evaluate the potential risks from facilities over long periods of time (greater than 30 years).

One can typically identify current land use, and indications of future land use, by reviewing hard copy and/or electronic versions of Land Use/Land Cover (LULC) maps, topographic maps, and aerial photographs. We list some sources below, and general information associated with several potential data and map resources. Also, as noted in Chapter 3, we recommend verifying that all mapping information you use is in the same Universal Transverse Mercator (UTM) coordinate system format (NAD27 or NAD83), to ensure consistency and prevent erroneous geo-referencing of locations and areas.

> ***Land Use/Land Cover (LULC) Maps*** **-** you can download LULC maps directly from the United States Geological Survey (USGS) web site (http://mapping.usgs.gov/index.html), at a scale of 1:250,000, in the GIRAS file format. LULC maps are also available from the EPA web site

(ftp://ftp.epa.gov/pub), at a scale of 1:250,000, in an Arc/Info export format. Within your study area, we recommend verifying the exact boundaries of polygons defining land use areas using available topographic maps and aerial photographs.

***Topographic Maps* -** Topographic maps are readily available in both hard copy and electronic format directly from the USGS or numerous other vendors. These maps are commonly at a scale of 1:24,000, and in TIFF file format with a TIFF World File included for georeferencing.

***Aerial Photographs* -** You can purchase hard copy aerial photographs directly from the USGS in a variety of scales and coverages. Electronic format aerial photographs or Digital Ortho Quarter Quads (DOQQs) are also available for purchase directly from the USGS, or from an increasing number of commercial sources.

Properly georeferenced DOQQs covering a 3-km or more radius of the assessment area, combined with overlays of the LULC map coverage and the ISCST3 modeled receptor grid node array, provide an excellent reference for identifying land use areas and justifying your choices of exposure scenario locations. The information above does not represent the universe of data available on human activities or land use. They are, however, readily available for little or no cost from a number of government sources, often via the Internet.

If feasible, we recommend verifying the accuracy of land use information with a site visit. Also, organizations exist that routinely collect and evaluate land use data (agricultural extension agencies, U.S. Department of Agriculture, natural resource and park agencies, and local governments). You may find discussions with these organizations helpful in updating current land use information or providing information regarding future land use. Local planning and zoning authorities are also potential sources of information on reasonable potential future land use. These authorities have information on the level of development allowed under current regulations, and what development may be expected in the future. The general public is another excellent source of information about land use in the area. Conducting a public workshop early in the data gathering process for the risk assessment can provide valuable information on land use, crops, special populations, etc. as well as starting a positive dialogue with the community. For example, by communicating with local tribes you might find that certain locations hold special significance for cultural or religious activities.

You can also use site-specific data on physiographic features (e.g., plant types, soil characteristics, land use, etc.) to verify the land uses identified using the resources listed above. You can readily determine the presence, type, and extent of physiographic features from the following sources:

- USGS topographic maps,

- Soil Conservation Service reports,
- county and local land use maps, and
- information from state departments of natural resources or similar agencies.

A study area might include multiple land uses, with differing current or potential human activity/land use characteristics. Your activity/land use analysis could identify multiple population centers (e.g., communities, residential developments, or rural residences), farms and ranches, or other land use types in the study area that would support recommended exposure scenarios. For example, if a study area includes a farm and a small residential community, you could consider both areas as possible exposure scenario locations (see Sections 4.2 and 4.3).

Once you've identified current land uses, we generally recommend also identifying areas with different reasonable potential ***future*** land use characteristics. For example, if a study area includes undeveloped property which could be converted to a residential community in the future, you might consider both of these land use types (i.e. undeveloped property, and residential community) in the risk assessment (see Sections 4.2 and 4.3). We recommend considering only potential future land uses which might reasonably be expected to occur. For example:

1. A rural area currently characterized as undeveloped open fields, could reasonably be expected to become farmland if it is able to support agricultural activities;
2. A rural area currently characterized by open fields and intermittent housing, could reasonably be expected to become a residential subdivision; and
3. An area currently characterized as a tidal swamp would **not** reasonably be expected to become farm land.

For transparency and clarity, we recommend describing any current or reasonably expected future land use in the risk assessment report. Of all the land use areas you identify, we generally recommend focusing on those areas that could be impacted by the COPC emissions you're evaluating in the risk assessment.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Identification and/or mapping of ***current*** land uses in the area, a description of the use, the area of the land described by the use, and the source of the information. You might choose to focus initially on those land use areas impacted by emissions of COPCs.
- Identification and/or mapping of the reasonable potential ***future*** land use areas, a description of the use, the source or rationale on which the description is based. You might choose to focus initially on those land use areas impacted by emissions of COPCs.

4.1.2 Water Bodies and Their Associated Watersheds

Surface water bodies and their associated watersheds are important factors in evaluating some of the recommended exposure scenarios. Specifically, water bodies can be important sources of fish for the fish ingestion pathway, or sources of water for the drinking water pathway (see Section 4.2). Your careful consideration is warranted when identifying which water bodies in the study area to assess. For the Fisher scenario, an appropriate water body (and/or its associated watershed) would receive deposition from the emission source, and be able to sustain a fish population harvested by humans. For the drinking water ingestion pathway, an appropriate waterbody (and/or its associated watershed) would receive deposition from the emission source, and be used as a direct drinking water source (i.e. not processed by a drinking water treatment facility). We recommend considering both current and potential human uses of water bodies found within the study area. In addition to identifying the human uses of water bodies, we recommend defining the surface areas and exact locations of the water bodies, and their associated watersheds. See Section 4.3 for a further discussion of selecting exposure scenario locations and their associated water bodies.

You can typically identify the use, area, and location of water bodies and their associated watersheds by reviewing the same hard copy and /or electronic versions of LULC maps, topographic maps, and aerial photographs used to identify land uses. We present sources and general information associated with each of these data types or maps in Section 4.1.1.

You might also get information on water body use from local authorities (e.g., state environmental agencies, fish and wildlife agencies, or local water control districts). This might include information about viability to support fish populations and drinking water sources. Surface water bodies that are used

As drinking water sources in the assessment area are generally evaluated in the risk assessment. While water bodies closest to the facility will generally have higher deposition rates, risk estimates are also affected by other physical parameters (e.g. the size of the water body and the associated watershed) and by the properties of the COPCs being emitted.

Once you've selected a water body, we recommend identifying the area extent (defined by UTM coordinates) of its watershed. Watershed runoff can be a significant contributor to overall water body COPC loadings. Media concentration equations use the extent of pervious and impervious areas in the watershed, as well as COPC concentrations in watershed soil, to calculate the water body COPC concentrations (see Chapter 5 and Appendix B). We therefore recommend clearly identifying and discussing the area extent of the watershed with the interested parties (both permitting authority and facility).

You generally define the area extent of a watershed by identifying topographic highs that result in downslope drainage into the water body. We recommend ensuring that the watershed and it's contribution to the water body are defined relative to the exposure scenario location associated with the water body (e.g. location on the water body of the drinking water intake, fishing pier, etc.), and subsequent risk estimates. Please keep in mind that the total watershed area can be very extensive relative to the area that is impacted from facility emissions.

For example, if facility emissions principally impact an area of land which drains into a specific tributary of a large river system and immediately upstream of a private drinking water intake point, you may wish to consider evaluating an "effective" watershed area rather than the entire watershed area of the large river system. For such a large river system, the watershed area can be on the order of thousands of square kilometers and can include numerous tributaries draining into the river at points that would have no net impact on the drinking water intake or on the water body COPC concentration at the exposure point of interest.

To use the HHRAP as recommended, you will need the following water body and watershed parameters (on an average annual basis):

- Water body surface area
- Watershed surface area
- Impervious watershed area

- Average water body volumetric flow rate
- Water body current velocity
- Depth of water column
- Total suspended solids (TSS)
- Universal Soil Loss Equation (USLE) rainfall/erosivity factor

The impervious watershed area is generally a function of urbanization within the watershed, and is typically presented as a percentage of the total watershed area. Volumetric flow rate and water body current velocity are typically annual average values. State or local geologic surveys often keep records on flow rate and current velocity of larger water bodies. You can calculate the volumetric flow rates for smaller streams or lakes by multiplying the watershed area by one-half of the local average annual surface runoff. Lacking site-specific data, you can calculate current velocities by dividing the volumetric flow rate by the cross-sectional area (***NOTE***: current velocities are not used in the equations for lakes). State or local sources sometimes have information on the depths of water bodies available. Discussions on determining the USLE rainfall/erosivity factor are included in Chapter 5 and Appendix B.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Identification and/or mapping of water bodies and associated watersheds potentially impacted by facility emissions of COPCs, including surface area of the water body and area extent of the contributing watershed, defined by UTM coordinates
- Rationale for selecting or excluding water bodies within the assessment area from evaluation
- Information on water body use that may justify including or excluding the water body from evaluation
- Documentation of water body area, watershed area, impervious area, volumetric flow rate, current velocity, depth of water column, total suspended solids (TSS), and the USLE rainfall/erosivity factor
- Description of assumptions made to limit the watershed area to an “effective” area
- Copies of all maps, photographs, or figures used to define water body and watershed characteristics

4.1.3 Special Population Characteristics

Special populations are human receptors or segments of the population that may be at higher risk due to increased sensitivity and/or increased exposure to COPCs. Fetuses, infants and children, and the elderly are examples of human life stages (i.e. populations) which might be more sensitive to COPC exposure. You might consider some tribal groups a special population because their ingestion of fish at rates higher than the general public increases their exposure to chemicals that bioaccumulate. Subsistence residents are also likely to have higher exposures from ingestion of meat (locally harvested game), produce (wild berries and onions, for example), and soil. There may be special locations where cultural activities are conducted, or that are sacred to the tribes, and we encourage evaluating exposures at these locations.

We've developed the assumptions specified in this guidance – such as the protective nature of the recommended exposure scenarios (see Section 4.2), and the use of RfDs which have been developed to account for toxicity to sensitive receptors – to also protect the health of special populations. However, you may also need to specifically address populations that are located in impacted areas because of unique characteristics of the exposure setting or to address particular community concerns. For example, a day care center or hospital may be located in an area that is directly impacted by the facility stack emissions. Receptors at these locations may be especially sensitive to the adverse effects and/or the exposure setting is particularly conducive to exposure. Consequently, due to site-specific exposure characteristics, exposure to children at the day care center, or to the sick in the hospital, might need to be specifically evaluated. Section 4.2 provides additional discussion on evaluating potential exposure of special populations, as part of evaluating recommended exposure scenarios. Additionally, the Agency has a stated policy focused on consistently and explicitly evaluating environmental health risks to infants and children in all risk assessments (U.S. EPA 1995j).

Concerns about special populations can arise at any time in the permitting process. We therefore recommend identifying special populations as part of characterizing the exposure setting. You can identify special populations in the assessment area based on the location of schools, hospitals, nursing homes, day care centers, parks, community activity centers, etc. If available information indicates that there are children exhibiting pica behavior (defined for risk assessment purposes as "an abnormally high soil ingestion rate") in the assessment area, these children could also represent a special population (see Section 6.2.3.1).

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- Identification and/or mapping of the locations of special populations at potentially higher risk from exposure to facility sources (anticipated to be located in areas impacted by facility emissions); focusing on the characteristics of the exposure setting to ensure that selected exposure scenario locations are protective of the special populations.

4.2 RECOMMENDED EXPOSURE SCENARIOS

We recommend evaluating the following exposure scenarios when they are consistent with site-specific exposure settings (also see Table 4-1):

- Farmer
- Farmer Child
- Resident
- Resident Child
- Fisher
- Fisher Child
- Acute Receptor
- Nursing Infant (Covered as exposure pathway under adult exposure scenarios)

These are the same exposure scenarios recommended by earlier OSW guidance, with the exception of the Farmer Child, Fisher Child, and acute receptor. The Farmer Child scenario was introduced into the indirect screening process in the risk assessment completed to support the proposed Hazardous Waste Combustion Rule and by NC DEHNR (1997). We include the Fisher Child scenario in order to be consistent with the adult/child pairings we recommend for the Resident and Farmer scenarios. We include the acute receptor scenario to ensure that the assessment evaluates all receptors that may be significantly exposed to emissions from facility sources.

In addition to the recommended exposure scenarios listed above, we recommend evaluating, where appropriate, special populations (as defined in Section 4.1.3) and communities of concern. Do this by identifying their locations, and determining whether they are located in areas with exposure setting characteristics that are particularly conducive to COPC impacts from facility emissions. Examples of additional exposure scenarios include hunters, trespassers, workers (see below), recreational fishers, etc.

You could evaluate some populations using a combination of a recommended exposure scenario expected to overestimate exposure compared to the populations, and maximum modeled air parameter values specific to the location (see Section 4.3). If this initial evaluation suggests that the receptors are protected, then no additional assessment is necessary. If, on the other hand, this evaluation estimates levels of risk which are of concern, a refined evaluation may be needed. The refined exposure scenario would evaluate the specific exposure pathways appropriate to the special population.

Take, for example, a children's school or day care center located in an area receiving deposition of facility emissions. You could evaluate potential exposure of children at this location using the Resident Child scenario at the location of the school or day care center. In most cases, evaluating this scenario at the school location will over-estimate exposure. This is because the Resident Child scenario includes an exposure pathway (ingestion of homegrown produce) which is most likely not occurring at that location. Also, the residential scenario assumes that a child breaths the air 24 hours/day, ingests 100 mg of soil/day; and is exposed for 6 years - when the child is probably only at day care 5 days/week and up to 10 hours/day. If this generates risk estimates of concern, you could conduct a more refined evaluation that adjusts the exposure assumptions to be more representative of the site.

We don't routinely recommend assessing workers at a facility that burns hazardous waste in the risk assessment, because we assume that those workers are protected by regulation and guidance of the U.S. Occupational Safety and Health Administration (OSHA). There are, however, some instances where workers impacted by exposure to facility emissions are not covered by the appropriate OSHA regulations. For example, workers located at a nearby but separate facility or commercial area, whose duties are independent of combustor operations, are not necessarily covered by the appropriate OSHA regulations. Also, on a site with multiple on-site activities (e.g., manufacturing, hazardous waste combustion, and military operations) the OSHA regulations would address the worker at the manufacturing operations with respect to those operations and not the emissions from the separate hazardous waste combustion operations. Considering such instances in the risk assessment may be appropriate.

We no longer refer to our recommended farmer and fisher exposure scenarios as "subsistence" scenarios. The associated daily consumption amounts (see Table 4-2, as well as Appendix C) are more comparable to reasonable (versus subsistence) amounts.

As mentioned above, an exposure scenario is defined as a combination of exposure pathways to which a receptor is subjected at a particular location. Table 4-1 presents the exposure pathways we recommend evaluating for each of the exposure scenarios. Food-related ingestion pathways could represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1994l; 1994g; 1998c; NC DEHNR 1997), due primarily to the potential for COPCs to bioaccumulate up the food chain.

TABLE 4-1

RECOMMENDED EXPOSURE SCENARIOS FOR A HUMAN HEALTH RISK ASSESSMENT

Exposure Pathways	**Recommended Exposure Scenarios**[a]						
	Farmer	**Farmer Child**	**Resident**	**Resident Child**	**Fisher**	**Fisher Child**	**Acute Receptor**[b]
Inhalation of Vapors and Particulates	•	•	•	•	•	•	•
Incidental Ingestion of Soil	•	•	•	•	•	•	--
Ingestion of Drinking Water from Surface Water Sources	•	•	•	•	•	•	--
Ingestion of Homegrown Produce	•	•	•	•	•	•	--
Ingestion of Homegrown Beef	•	•	--	--	--	--	--
Ingestion of Milk from Homegrown Cows	•	•	--	--	--	--	--
Ingestion of Homegrown Chicken	•	•	d	d	d	d	--
Ingestion of Eggs from Homegrown Chickens	•	•	d	d	d	d	--
Ingestion of Homegrown Pork	•	•	--	--	--	--	--
Ingestion of Fish	d	d	d	d	•	•	--
Ingestion of Breast Milk	c	--	c	--	c	--	--

Notes:

•	Pathway is included in exposure scenario.
--	Pathway is not included in exposure scenario.
a	Exposure scenarios are defined as a combination of exposure pathways evaluated for a receptor at a specific location.
b	The acute receptor scenario evaluates short-term 1-hour maximum COPC air concentrations (see Chapter 3) at any land use area that would support the other recommended exposure scenarios, as well as commercial and industrial land use areas (excluding workers at the facility being directly evaluated in the risk assessment).
c	Infant exposure to PCDDs, PCDFs, and dioxin-like PCBs via the ingestion of their mother's breast milk is evaluated as an additional exposure pathway, separately from the recommended exposure scenarios identified in this table (see Chapter 2).
d	Site-specific exposure setting characteristics (e.g., presence of ponds on farms, or presence of ponds or small livestock within semi-rural residential areas) may warrant the permitting authority consider adding this exposure pathway to the scenario (see Section 4.2).

As indicated in Table 4-1, some exposure setting characteristics may warrant you consider including additional exposure pathways when evaluating a particular exposure scenario. For example, the recommended Farmer exposure scenario doesn't typically include the fish ingestion exposure pathway. However, in some areas of the country it's common for farms to have stock ponds that are fished on a regular basis for the farm family's consumption. Since the ingestion rates we recommend for those food pathways already considered in the evaluation are not significant enough to preclude the Farmer also ingesting the fish caught from the local pond, the fish ingestion exposure pathway may also be relevant in such locations. You could use the same rationale for residential scenarios where residents are located in semi-rural areas which allow small livestock (e.g., free range poultry for eggs), and/or residents located by small ponds suitable for fishing, or wetlands that support crawfish harvest.

We also recommend evaluating infant exposure to PCDDs and PCDFs via the ingestion of their mother's breast milk as an additional exposure pathway at all recommended adult exposure scenario locations. Chapter 2 and Appendix C further describe the ingestion of breast milk exposure pathway.

In addition, although some risk assessments conducted by U.S. EPA (1996b) have discounted the direct inhalation risks to all receptors except the adult Resident (nonfarmer) and Resident Child (nonfarmer), we generally recommend evaluating the direct inhalation exposure pathway for all receptors.

We don't typically recommend evaluating the following exposure pathways as part of an exposure scenario:

> ***Ingestion of Ground Water*** - U.S. EPA (1998c) found that ground water is an insignificant exposure pathway for combustion emissions; in addition, U.S. EPA (1994k) noted that uptake from ground water into food crops and livestock is minimal because of the hydrophobic nature of dioxin-like compounds. We anticipate potential exposure to COPCs through ingestion of drinking water from surface water bodies to be much more significant. Ingestion of ground water is further discussed in Section 6.2.4.2.
>
> ***Inhalation of Resuspended Dust*** - U.S. EPA (1990e) found that risk estimates from inhalation of resuspended dust was insignificant. We anticipate exposure through direct inhalation of vapor and particle phase COPCs and incidental ingestion of soil to be much more significant. Inhalation of resuspended dust is further discussed in Section 6.2.3.3.
>
> ***Dermal Exposure to Surface Water, Soil, or Air*** - Available data indicate that the contribution of dermal exposure to soils to overall risk is typically small (U.S. EPA 1996g; 1995h). For example, the risk assessment conducted for the Waste Technologies Industries, Inc., hazardous waste incinerator in East Liverpool, Ohio, indicated that—for an adult farmer in a subarea with high exposures—the risk resulting from soil ingestion and dermal contact was 50-fold less than

the risk from any other exposure pathway and 300-fold less than the total estimated risk (U.S. EPA 1996g; 1995h). Also, there are significant uncertainties associated with estimating potential COPC exposure via the dermal exposure pathway. The most significant of these uncertainties are associated with determining the impact of soil characteristics and the extent of exposure (e.g., the amount of soil on the skin and the length of exposure) on estimating compound-specific absorption fractions (ABS).

We don't generally recommend evaluating the dermal exposure to soil pathway as part of the recommended exposure scenarios. However, if either a facility or a permitting authority feel that site-specific conditions indicate dermal exposure to soil may contribute significantly to total soil-related exposures, we recommend following the relevant methods described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (U.S. EPA 1998c). Dermal exposure is further discussed in Section 6.2.3.2 of this guidance.

Inhalation of COPCs and Ingestion of Water by Animals - We don't recommend these animal exposure pathways in calculating animal tissue concentration because we expect their contributions to total risk to be negligible compared to the contributions of the recommended animal exposure pathways. However, you might need to evaluate these exposure pathways on a case-by-case basis considering site-specific exposure setting characteristics.

Our recommended exposure scenarios are further discussed in the following subsections.

4.2.1 Farmer

The Farmer exposure scenario is made up of the exposure pathways through which an adult member of a farming or ranching family could be exposed. We recommend including this scenario when farming or ranching takes place, or may reasonably take place some time in the future, in the study area. As indicated in Table 4-1, we recommend assuming the Farmer is exposed to COPCs emitted from the facility through the following exposure pathways:

- Direct inhalation of vapors and particles
- Incidental ingestion of soil
- Ingestion of drinking water from surface water sources
- Ingestion of homegrown produce (i.e. fruits and vegetables)
- Ingestion of homegrown beef
- Ingestion of milk from homegrown cows
- Ingestion of homegrown chicken
- Ingestion of eggs from homegrown chickens
- Ingestion of homegrown pork
- Ingestion of breast milk (evaluated separately, for an infant of the Farmer; see Chapter 2)

While on the farm property, the Farmer inhales air containing COPC-impacted vapors and suspended particles. Through daily activities, the Farmer ingests incidental amounts of soil. If site characterization suggests that impacted surface waterbodies are used as direct drinking water sources (see Section 4.1.2), the farm family receives its water from a surface waterbody. The farm family raises and consumes beef and milk cattle, pigs, and free-range chickens (including eggs). Cattle ingest soil while foraging on a grazing field, as well as being fed silage and grain grown on the farm. Pigs are contained within a yard or small field, where they are assumed not to forage, but ingest soil while being fed a combination of silage and grain grown on the farm. Free-range chickens are contained within a yard or field, where they ingest soil while being fed grain grown on the farm. The Farmer grows enough fruits and vegetables to supply the family with produce.

The scenario assumes that a portion of the Farmer's diet comes from each homegrown food type listed above (see Table 6-1 and Appendix C for consumption rates). All of these portions are impacted by emissions from the facility being assessed. The recommended consumption rates don't represent the Farmer's entire intake of each food type, but rather only the homegrown portion of the Farmer's diet. It is therefore reasonable to assume that 100% of this subset of each food type (i.e. the homegrown portion) is contaminated. Also, because the portions represent only the homegrown portion of the Farmer diet, assuming ingestion of all meat groups by the Farmer does not grossly overestimate the total amount of meat a farmer or rancher could reasonably consume. Breaking out consumption by food type is an important step in estimating the relative contributions to COPC-specific risk from ingestion of each food type.

Previous Agency guidance (for example, U.S. EPA 1993f and U.S. EPA 1994f) didn't include the ingestion of chicken and eggs exposure pathways. NC DEHNR (1997) considers chicken and egg ingestion pathways only for exposure to dioxins and furans, because biotransfer factors were only available for dioxins and furans when that guidance was published. U.S. EPA (1998c) includes ingestion of both poultry and eggs. Currently, biotransfer factors can be derived from literature data for other organic compounds and metals. Therefore, we generally recommend including the chicken and egg ingestion exposure pathways for all COPCs with available biotransfer factors. Further discussion of these exposure pathways, including numeric equations, parameters values, and COPC-specific inputs, can be found in Chapter 5 and Appendices A, B, and C.

When evaluating the ingestion of drinking water from surface water for the Farmer scenario, we generally recommend also considering the potential for ingestion of cistern water at farm or ranch locations, in addition to surface water sources. If site-specific information (e.g. interviews with local health departments) suggests that cistern water is likely used, or could be used for a drinking water source, you could evaluate ingestion of cistern water in a manner similar to that used to evaluate ingestion of water from a surface water body (see Chapter 5 and Appendix B). Site-specific information (e.g. do cisterns in the study area tend to be covered or uncovered?) can educate decision makers as to appropriate equations and parameter values to use in assessing the ingestion of drinking water from cisterns.

We don't usually recommend the ingestion of fish exposure pathway for the Farmer exposure scenario. However, as indicated in the notes to Table 4-1, we do recommend that you consider evaluating the fish ingestion pathway if regional or site-specific exposure setting characteristics (e.g., presence of ponds on farms or ranches that support fish for human consumption) are identified that warrant consideration. You can use the applicable estimating media concentration equations for ingestion of fish as presented in Chapter 5 and Appendix B. Also, evaluating the Fisher and Fisher Child exposure scenarios (see Sections 4.2.5 and 4.2.6) at farm or ranch locations may be appropriate where on-site ponds are used as sources of fish for human consumption.

We recommend evaluating the exposure of an infant to PCDDs, PCDFs, and dioxin-like PCBs via the ingestion of breast milk as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

If site-specific information is available indicating that farmers in the study area don't raise a type of livestock, nor is raising that type of livestock likely to occur in the future, then you could reasonably consider eliminating the related exposure pathway (or pathways, in the case of chicken and egg ingestion). However, if one meat source is not used, its place in the diet is often taken by one or more of the remaining exposure pathways. Take care, therefore, to consider the intake rates of the remaining exposure pathways, to ensure that the total amount consumed (summed fraction from each food group) is representative. See Chapter 6 (Quantifying Exposure) for further discussion of the implications of modifying our recommended exposure pathways.

4.2.2 Farmer Child

The Farmer Child exposure scenario is made up of the exposure pathways through which a child member of a farming or ranching family may reasonably be expected to be exposed. Agency policy recommends consistently and explicitly evaluating environmental health risks to infants and children in all risk assessments (U.S. EPA 1995j). As indicated in Table 4-1, the scenario assumes the Farmer Child is exposed to COPCs emitted from the facility through the same exposure pathways as the Farmer. The primary differences between the Farmer and Farmer Child are in exposure duration (6 years for the child vs . 40 years for the adult), and consumption rates (e.g. 1.4 homegrown produce servings per week for child vs. 2.8 homegrown produce servings per week for adult, see Table 6-1).

4.2.3 Resident

The Resident exposure scenario is made up of the exposure pathways through which an adult receptor may be exposed in an urban or nonfarm rural setting. We recommend including the adult Resident scenario, because potential exposure to COPCs through ingesting homegrown produce has been shown to be potentially significant. This exposure scenario equates with the "Home Gardener" scenario recommended by U.S. EPA (1994g) and NC DEHNR (1997). As indicated in Table 4-1, the scenario assumes the adult Resident is exposed to COPCs from the emission source through the following exposure pathways:

- Direct inhalation of vapors and particles
- Incidental ingestion of soil
- Ingestion of drinking water from surface water sources
- Ingestion of homegrown produce
- Ingestion of breast milk (evaluated separately, for an infant of the Resident; see Chapter 2)

While on their property, the Resident inhales air containing COPC-impacted vapors and suspended particles. Through daily activities, the Resident ingests incidental amounts of soil. If site characterization suggests that impacted surface waterbodies are used as direct drinking water sources (see Section 4.1.2), the resident family receives its water from a surface waterbody. The Resident grows fruits and vegetables for home consumption (NC DEHNR 1997). Breaking out consumption by exposure pathway is an important step in estimating the relative contributions to COPC-specific risk from ingestion of each food type. Further discussion of these exposure pathways, including equations, parameter values, and COPC-specific inputs, can be found in Chapter 5 and Appendices A, B, and C.

We don't usually recommend evaluating the ingestion of fish exposure pathway for the Resident exposure scenario. However, as indicated in the notes to Table 4-1, we do recommend that you consider evaluating the fish ingestion pathway if exposure setting characteristics (e.g., presence of ponds within semi-rural residential areas that support fish for human consumption) are identified that warrant consideration. It may be appropriate to evaluate the Fisher and Fisher Child exposure scenarios (see Sections 4.2.5 and 4.2.6) at residential locations where ponds or surface water bodies are used as a potential source of fish for human consumption.

We recommend evaluating exposure of an infant to PCDDs, PCDFs, and dioxin-like PCBs via the ingestion of breast milk as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

4.2.4 Resident Child

The Resident Child exposure scenario is made up of the exposure pathways through which a child receptor may be exposed in an urban or nonfarm rural setting. This exposure scenario equates with the "Child of the Home Gardener" scenario recommended by U.S. EPA (1994g) and NC DEHNR (1997). Agency policy recommends consistently and explicitly evaluating environmental health risks to infants and children in all risk assessments (U.S. EPA 1995j). As indicated in Table 4-1, the scenario assumes the Resident Child is exposed to COPCs emitted from the facility through the same exposure pathways as the Resident adult. The primary differences between the Resident and Resident Child are in exposure duration (6 years for the child vs . 30 years for the adult), and consumption rates (e.g. 1.2 homegrown produce servings per week for the child vs. 2.3 homegrown produce servings per week for the adult, see Table 6-1).

4.2.5 Fisher

The Fisher exposure scenario is made up of the exposure pathways through which an adult receptor may be exposed in an urban or nonfarm rural setting where fish is the main source of protein in the receptor diet. We recommend including the Fisher scenario, because food-related ingestion routes may represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1994l; 1994g; 1998c; NC DEHNR 1997). The potential exposure is due primarily to the potential for COPCs to bioaccumulate up the food chain. Breaking out consumption by exposure pathway is an important step in estimating the relative contributions to COPC-specific risk from ingestion of each food type. As indicated in Table 4-1, the scenario assumes the Fisher is exposed to COPCs emitted from the facility through the following exposure pathways:

- Direct inhalation of vapors and particles
- Incidental ingestion of soil
- Ingestion of drinking water from surface water sources
- Ingestion of homegrown produce
- Ingestion of fish
- Ingestion of breast milk (evaluated separately, for an infant of the Fisher; see Chapter 2)

While on their property (i.e. where they reside), the Fisher inhales air containing COPC-impacted vapors and suspended particles. Through daily activities, the Fisher ingests incidental amounts of soil. If site characterization suggests that impacted surface waterbodies are used as direct drinking water sources (see Section 4.1.2), the fisher family receives its water from a surface waterbody. The Fisher grows fruits and vegetables for home consumption (NC DEHNR 1997). The Fisher harvests enough fish from waterbodies in the study area impacted by facility emissions to supply the family with a significant portion of their protein. Further discussion of these exposure pathways, including numeric equations, parameters values, and COPC specific inputs, can be found in Chapter 5 and Appendices A, B, and C.

We recommend evaluating the exposure of an infant to PCDDs, PCDFs, and dioxin-like PCBs via the ingestion of breast milk as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

4.2.6 Fisher Child

The Fisher Child exposure scenario is made up of the exposure pathways through which a child receptor may be exposed in an urban or nonfarm rural setting where fish is the main source of protein in the receptor diet. Evaluating this exposure scenario is the same as the adult/child pairings recommended for the Farmer and Resident scenarios. In addition, Agency policy recommends consistently and explicitly evaluating environmental health risks to infants and children in all risk assessments (U.S. EPA 1995j). As indicated in Table 4-1, the scenario assumes the Fisher Child is exposed to COPCs emitted from the facility through the same exposure pathways as the Fisher. The primary differences between the Fisher and Fisher Child are in exposure duration (6 years for child vs. 30 years for the adult), and consumption rates (e.g. 1.2 homegrown produce servings per week for the child vs. 2.3 homegrown produce servings per week for the adult, see Table 6-1).

4.2.7 Acute Receptor Scenario

In addition to long-term chronic effects evaluated in the other recommended exposure scenarios, we generally recommend evaluating the acute exposure scenario. The acute receptor scenario accounts for short-term effects of exposure to maximum 1-hour concentrations of COPCs in emissions from the facility (see Chapter 3) through direct inhalation of vapors and particles (see Table 4-1 and Chapter 7). A receptor could be exposed in an urban or rural setting where human activity or land use supports any of the recommended exposure scenarios. The receptor could also be exposed in commercial and industrial land use areas (excluding workers from the facility) not typically covered by the other recommended exposure scenarios. As mentioned in Section 4.2 above, we assume that workers from the facility being assessed in the risk assessment are protected by OSHA programs, and therefore aren’t generally included in hazardous waste combustion risk assessments.

We discuss further this recommended exposure scenario and associated exposure pathway, including numeric equations, parameters values, and COPC-specific inputs, in Chapter 7 and Appendices A, B, and C.

4.3 SELECTING EXPOSURE SCENARIO LOCATIONS

Exposure scenario locations are the physical places within the study area selected for evaluating one or more of the recommended exposure scenarios. We generally recommend choosing exposure scenario locations based on COPC air concentrations and deposition rates from ISCST3 (see Chapter 3) specific to land use areas defined during exposure setting characterization (see Section 4.1). Location-specific air concentrations and deposition rates are then used as inputs to the equations which estimate media concentrations.

We would like to emphasize that the method and resulting selection of exposure scenario locations is one of the most critical steps of the risk assessment process, with huge impacts on standardization across all facilities evaluated, and reproducibility of results. This is, at least partly, because ISCST3-modeled air parameter values (and the resulting media concentration estimates) can vary significantly, even within individual land use areas.

To ensure consistent and reproducible risk assessments, we recommend using the following procedures to select your exposure scenario locations. These procedures also reduce the chances that the location(s) you select to evaluate a land use area overlook locations within that same land use area that would result in higher risk estimates. This can be important given the complexity of multiple modeled air parameters and phases per location, possibly multiple facility emission sources, each with multiple source-specific COPCs. This approach also provides a more complete risk evaluation of areas surrounding the facility. This information often becomes relevant later in the permitting process and in risk communication to the surrounding public.

As detailed in Chapter 3, ISCST3 estimates COPC concentrations in the air above, and deposition rates onto, specific locations (i.e. receptor grid nodes) around a central point (e.g. combustor facility stack). If all the locations modeled by ISCST3 are viewed as a group, they form a grid of horizontal and vertical lines on a map, with each location a node, or intersection between vertical and horizontal lines; hence the name "grid nodes" for modeled locations. Also, Section 4.1 of this chapter explained the steps and issues involved in characterizing the various uses of the land in the study area. Figure 4-1 is a graphic representation of these two sets of information, and demonstrates some of the relationship between them. For example, a single land use area can have multiple grid nodes associated with it, each node with its own air concentration and deposition levels. Choosing exposure scenario location(s) for a land use area is a matter of choosing which grid node(s) will provide the data used to generate media concentrations used in the exposure scenario. We recommend the following steps:

FIGURE 4-1

ISCST3 GRID NODES AND LAND USE DESIGNATIONS

Step 1: Define Land Use Areas To Evaluate **-** To avoid confusion and misidentification, land use areas, water bodies, and watersheds identified during the exposure setting characterization step, are best defined and mapped using UTM coordinates in a format consistent with that used to define locations of facility emission sources and the ISCST3 receptor grid nodes. Formats include NAD27 or NAD83 UTM.

Step 2: Identify Receptor Grid Node(s) Within Each Defined Land Use Area **-** For each defined land use area, identify the receptor grid nodes within or on the boundary of that area (defined in Step 1) that represent the location of highest ***yearly average*** concentration for each ISCST3 air parameter output (i.e., air concentration, dry deposition, wet deposition) for each phase (i.e., vapor, particle, particle-bound). We recommend choosing concentrations specific to each facility emission source (e.g., stacks, fugitives), as well as all emission sources at the facility combined. This results in selecting one or more receptor grid nodes (and therefore the exposure scenario locations for that land use area), with the following attributes:

- Highest vapor phase air concentration
- Highest vapor phase dry deposition rate

- Highest vapor phase wet deposition rate
- Highest particle phase air concentration
- Highest particle phase wet deposition rate
- Highest particle phase dry deposition rate
- Highest particle-bound phase air concentration
- Highest particle-bound phase wet deposition rate
- Highest particle-bound phase dry deposition rate

With the exception of water bodies and watersheds (discussed in Step 4 below), we recommend using only air parameters for a single receptor grid node as inputs into the media equations for each exposure scenario location. We also recommend using actual parameter values, without averaging or other statistical manipulation. However, based generally on the number and location of facility emission sources, you might select multiple exposure scenario locations for a specific land use area.

U.S. EPA Region 6 applied these criteria to actual sites, using actual modeled air parameters, and found that only 1 to 3 receptor grid nodes were typically selected per land use area. This was because, in most cases, the highest air concentration and deposition rate occurred at the same receptor grid node.

> ***Please note:*** while these criteria tend to minimize the chances of overlooking maximum risk within a land use area, they do not preclude you from selecting additional exposure scenario locations within that same land use area based on site-specific risk considerations (see Step 3 below).

***Step 3: Identify Receptor Grid Nodes For Acute Risk and Site-Specific Risk Considerations* -** In addition to the receptor grid nodes selected in Steps 1 and 2 above, you might consider additional receptor grid nodes to evaluate acute risk or site-specific risk considerations (e.g., special populations).

To evaluate a land use area (including commercial and industrial land use areas) for acute risk, choose location(s) from receptor grid nodes with the highest modeled ***hourly*** vapor phase air concentration and highest ***hourly*** particle phase air concentration (see Chapter 3) specific to each emission source, as well as all emission sources combined. For site-specific risk considerations, we recommend considering the receptor grid node closest to the exposure point being evaluated (e.g. school, hospital). However, in some cases, a more protective approach might select the closest receptor grid node or nodes with the highest modeled air parameter values.

***Step 4: Identify Receptor Grid Nodes For Water Bodies and Watersheds* -** For recommended exposure scenarios that include evaluating water bodies and their associated watersheds, we recommend considering the receptor grid nodes within their area extent or "effective" areas (defined and mapped in Step 1). For water bodies, you could select the receptor grid node with the highest modeled air parameter values. You could also average the air parameter values for all receptor grid nodes within the area of the water body. For watersheds, you could average the modeled air parameter values of all receptor grid nodes within the drainage basin (excluding the area of the water body). Media concentration equations for water bodies and watersheds need the same air parameter values as found in Step 2 above; yearly averages for each ISCST3 modeled air parameter (e.g., air concentration, dry deposition, wet deposition) for each phase (e.g., vapor, particle, particle-bound); specific to each emission source (e.g., stacks, fugitives) as well as all emission sources at the facility combined.

For evaluating potential exposure routes other than ingestion of fish, we consider it reasonable to assume that the Fisher and Fisher Child reside at the same exposure scenario locations as the Resident scenario. You can similarly assume that the Fisher and Fisher Child exposure scenarios are exposed through ingestion of fish from the water body with the highest modeled combined deposition, that can or does support fish populations. As a result of some site specific conditions, it may be appropriate to evaluate the Fisher and Fisher Child assuming exposure through ingestion of fish calculated using COPC water concentrations from one water body, and exposure from ingestion of drinking water calculated using COPC water concentrations from a different water body.

To reiterate, we recommend initially evaluating current and reasonable potential future land use areas defined during the exposure setting characterization, using the most representative recommended exposure scenario(s), at actual receptor grid nodes selected using the four-step process explained above.

Chapter 5
Estimating Media Concentrations

What's Covered in Chapter 5:

5.1 Calculating COPC Concentrations in Air for Direct Inhalation
5.2 Calculating COPC Concentrations in Soil
5.3 Calculating COPC Concentrations in Produce
5.4 Calculating COPC Concentrations in Beef and Dairy Products
5.5 Calculating COPC Concentrations in Pork
5.6 Calculating COPC Concentrations in Chicken and Eggs
5.7 Calculating COPC Concentrations in Drinking Water and Fish
5.8 Using Site-Specific vs. Default Parameter Values

The purpose of this chapter is to describe the equations (and associated parameters) for estimating media concentration that we recommend using to evaluate the exposure scenarios presented in Chapter 4. In most cases, we include the origin and development of each of these equations, and describe the associated parameters. We also present the equations in Appendix B in a more condensed form (i.e. without derivation), and organize them according to exposure pathway. Discussions of ISCST3-modeled unitized air parameters are presented in Chapter 3. Appendix B also includes equations for modeling phase allocation and speciation of mercury concentrations. Appendix A-2 lists compound-specific parameters the equations need to estimate media concentrations, as well as our recommended hierarchies of sources. The HHRAP companion database provides recommended values for compound-specific parameters.

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak in this chapter is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

Section 5.1 describes the equations that estimate air concentrations for evaluating direct inhalation of COPCs. Section 5.2 describes equations for estimating COPC concentrations in soils. Section 5.3

describes equations for estimating COPC concentrations in produce. Sections 5.4 through 5.6 describe equations for estimating COPC concentrations in animal products (such as milk, beef, pork, poultry, and eggs) resulting from animals ingesting contaminated feed and soil. Section 5.7 describes equations for estimating COPC concentrations in fish through bioaccumulation (or, for some compounds, bioconcentration) from the water column, dissolved water concentration, or bed sediment—depending on the COPC.

> ***Please Note***: references made throughout Chapter 5 to particle phase are generic and made without distinction between particle and particle-bound.

5.1 CALCULATING COPC CONCENTRATIONS IN AIR FOR DIRECT INHALATION

We recommend calculating COPC concentrations in air by summing the vapor phase and particle phase air concentrations of COPCs. To evaluate long-term or chronic exposure via direct inhalation, we generally recommend using unitized ***yearly*** air parameter values to calculate air concentrations, as specified in Appendix B, Table B-5-1. To evaluate short-term or acute exposure via direct inhalation, we recommend using unitized ***hourly*** air parameter values to calculate air concentrations, as specified in Appendix B, Table B-6-1.

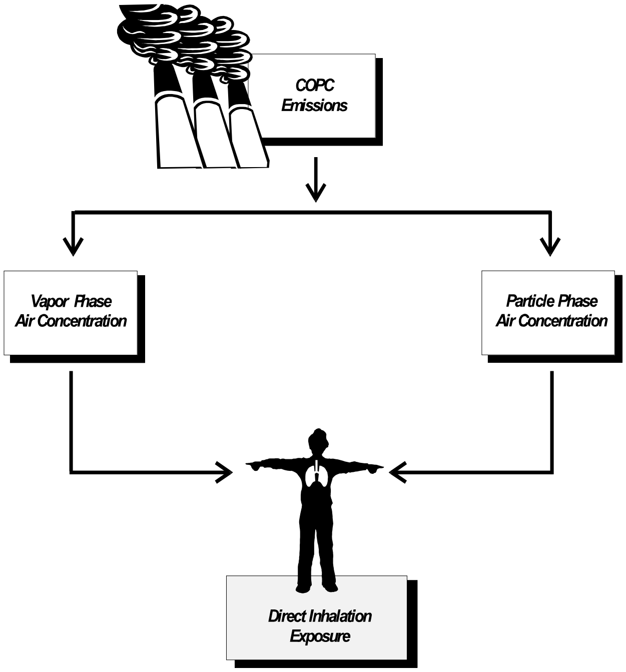
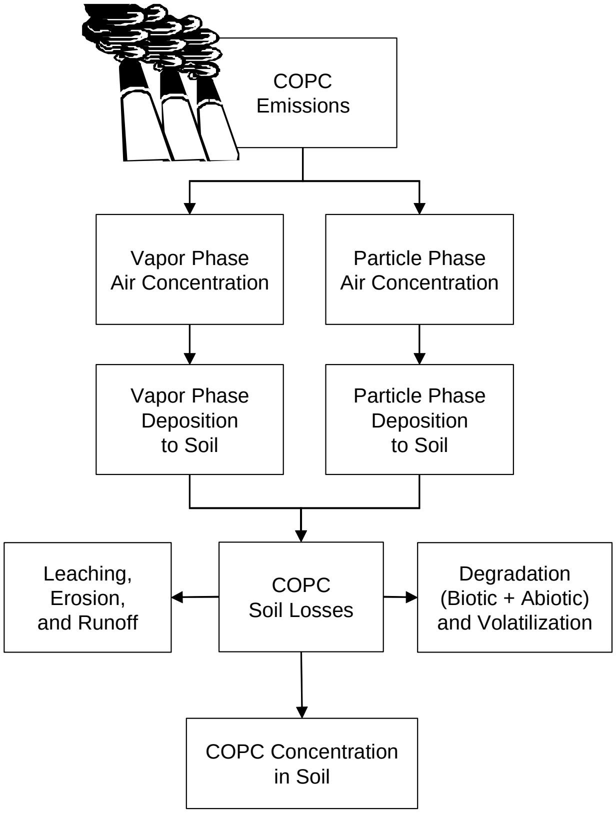


Figure 5-1 - COPC Concentration in Air for Direct Inhalation

5.2 CALCULATING COPC CONCENTRATIONS IN SOIL

We recommend estimating COPC concentrations in soil by summing the vapor phase and particle phase deposition of COPCs to the soil. We generally recommend considering wet and dry deposition of particles and vapors. Calculate dry deposition of vapors from the vapor air concentration and the dry deposition velocity. We consider it appropriate for soil concentration calculations to account for loss of COPCs by several mechanisms, including leaching, erosion, runoff, degradation (biotic and abiotic), and volatilization. These loss mechanisms all lower the soil concentration associated with the deposition rate. We present our recommended equations for calculating soil concentration and soil losses of COPCs in Appendix B, Tables B-1 for land use areas, and Tables B-4 for watersheds (see Section 5.7).

FIGURE 5-2 -
COPC CONCENTRATION IN SOIL



Soil concentrations might require many years to reach steady state. As a result, the equations we suggest to calculate the average soil concentration over the period of deposition were derived by integrating the instantaneous soil concentration equation over the period of deposition. For carcinogenic COPCs, we recommend using two variations of the equation (average soil concentration over exposure duration):

1. one variation if the exposure duration (*T2*) is ***greater than or equal to*** the operating lifetime of the emission source or time period of combustion, and

2. the other form if the exposure duration is ***less*** than the operating lifetime of the emission source or time period of combustion.

For noncarcinogenic COPCs, we recommend using the second form of the carcinogenic equation. This equation calculates the highest annual average COPC soil concentration occurring during the exposure duration. We describe these equations in more detail in Section 5.2.1.

Soil conditions—such as pH, structure, organic matter content, and moisture content—affect the distribution and mobility of COPCs. Modeling the loss of COPCs from soil uses rates specific to the physical and chemical characteristics of the soil. We describe these variables and their use in the following subsections, along with the recommended equations.

5.2.1 Calculating Cumulative Soil Concentration (*Cs*)

U.S. EPA (1990e) recommended using Equation 5-1—adapted from Travis, et al. (1983)—to calculate cumulative soil concentration:

$$Cs = \frac{100 \cdot (Dydp + Dywv) \cdot [1.0 - \exp\ (- ks \cdot tD)]}{Z_s \cdot BD \cdot ks} \qquad \text{Equation 5-1}$$

where

Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
100 = Units conversion factor (mg-m^2/kg-cm^2)
$Dydp$ = Unitized yearly dry deposition from particle phase (s/m^2-yr)
$Dywv$ = Unitized yearly wet deposition from vapor phase (s/m^2-yr)
ks = COPC soil loss constant due to all processes (yr^{-1})
tD = Time period over which deposition occurs (time period of combustion) (yr)
Z_s = Soil mixing zone depth (cm)
BD = Soil bulk density (g soil/cm^3 soil)

U.S. EPA (1993f) stated that Equation 5-1 evaluated deposition of particle phase COPCs, but failed to consider vapor phase deposition or diffusion. To account for vapor phase diffusion, U.S. EPA (1998c) recommended using the following equation:

$$Cs = \frac{100 \cdot (Dydp + Dywv + L_{dif}) \cdot [1.0 - \exp\ (-ks \cdot tD)]}{Z_s \cdot BD \cdot ks}$$ Equation 5-1A

where

Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
100 = Units conversion factor (mg-m^2/kg-cm^2)
$Dydp$ = Unitized yearly dry deposition from particle phase (s/m^2-yr)
$Dywv$ = Unitized yearly wet deposition from vapor phase (s/m^2-yr)
L_{dif} = Dry vapor phase diffusion load to soil (g/m^2-yr)
ks = COPC soil loss constant due to all processes (yr^{-1})
tD = Time period over which deposition occurs (time period of combustion) (yr)

Z_s = Soil mixing zone depth (cm)
BD = Soil bulk density (g soil/cm^3 soil)

Other guidance (U.S. EPA 1994g) recommended the original Equation 5-1, but only for calculating *Cs* for 2,3,7,8-TCDD. U.S. EPA (1994g) also recommended setting the COPC soil loss constant (*ks*) equal to 0 for all other COPCs. For COPCs other than 2,3,7,8-TCDD, U.S. EPA (1994g) recommended Equation 5-1B—which eliminates the COPC soil loss constant:

$$Cs = 100 \cdot \frac{Dyd + Dyw}{Z_s \cdot BD} \cdot tD$$ Equation 5-1B

where

Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
100 = Units conversion factor (mg-m^2/kg-cm^2)
Dyd = Yearly dry deposition rate of pollutant (g/m^2-yr)
Dyw = Yearly wet deposition rate of pollutant (g/m^2-yr)
tD = Time period over which deposition occurs (time period of combustion) (yr)
Z_s = Soil mixing zone depth (cm)
BD = Soil bulk density (g soil/cm^3 soil)

More recent guidance documents—U.S. EPA (1994r) and NC DEHNR (1997)—recommended using two different equations (Equations 5-1C and 5-1D) with carcinogenic COPCs. Equation 5-1C was recommended for $T_2 \leq tD$ and Equation 5-1D was recommended for $T_1 < tD < T_2$. For noncarcinogenic COPCs, Equation 5-1E was recommended.

We recommend using Equations 5-1C, 5-1D, and 5-1E to calculate *Cs*. Appendix B, Table B-1-1 discusses further how to use these equations.

Recommended Equations for Calculating:
Cumulative Soil Concentration (*Cs*)

Carcinogens:

For $T_2 \leq tD$

$$Cs = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left[\left(tD + \frac{\exp\ (-ks \cdot tD)}{ks} \right) - \left(T_1 + \frac{\exp(-ks \cdot T_1)}{ks} \right) \right] \qquad \text{Equation 5-1C}$$

For $T_1 < tD < T_2$

$$Cs = \frac{\left(\frac{Ds \cdot tD - Cs_{tD}}{ks}\right) + \left(\frac{Cs_{tD}}{ks}\right) \cdot \left(1 - \exp\ [-ks \cdot (T_2 - tD)]\right)}{(T_2 - T_1)} \qquad \text{Equation 5-1D}$$

Noncarcinogens:

$$Cs_{tD} = \frac{Ds \cdot [1 - \exp(-ks \cdot tD)]}{ks} \qquad \text{Equation 5-1E}$$

where

Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Ds = Deposition term (mg COPC/kg soil/yr)
ks = COPC soil loss constant due to all processes (yr^{-1})
tD = Time period over which deposition occurs (time period of combustion) (yr)
T_1 = Time period at the beginning of combustion (yr)
Cs_{tD} = Soil concentration at time tD (mg/kg)
T_2 = Length of exposure duration (yr)

We discuss the deposition term further in this Section, as well as Section 5.2.3. Section 5.2.2 discusses the COPC-specific soil loss constant (*ks*). Chapter 2 discusses how the period of time at the beginning of combustion (T_1) relates to characterizing site conditions immediately preceding the study period. Chapter 2 also addresses the time period during which burning - and therefore deposition - occurs (*tD*), as it relates to setting emission rates. Chapter 3 addresses air dispersion modeling aspects of *tD*. Chapter 6 further discusses how the duration of exposure (T_2) relates to characterizing exposure.

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-1C when you model an exposure duration that is ***less than or equal to*** the operating lifetime of the emission source or hazardous waste combustor ($T_2 \leq tD$). We recommend using Equation 5-1D when you model an exposure duration ***greater than*** the operating lifetime of the hazardous waste combustor ($T_1 < tD < T_2$). For noncarcinogenic COPCs, we recommend Equation 5-1E.

We generally recommend using the COPC soil concentration averaged over the exposure duration (represented by *Cs*) for carcinogenic compounds. Carcinogenic risk is averaged over the lifetime of an individual. Because the hazard quotient associated with noncarcinogenic COPCs is based on a threshold dose rather than a lifetime exposure, we recommend using the highest annual average COPC soil concentration (Cs_{tD}) occurring during the exposure duration period for noncarcinogenic COPCs. Cs_{tD} typically occurs at the end of the operating life of the emission source or the time period of combustion.

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using the highest 1-year annual average soil concentration, determined using Equation 5-1E, to evaluate risk from noncarcinogenic COPCs (see Chapter 7).

5.2.2 Calculating the COPC Soil Loss Constant (*ks*)

Organic and inorganic COPCs can be lost from the soil by several processes that may or may not occur simultaneously. The rate at which a COPC is lost from the soil is known as the soil loss constant (*ks*). We recommend determining *ks* by using the soil's physical, chemical, and biological characteristics, to estimate the COPC-specific loss resulting from:

(1) leaching,
(2) runoff,
(3) erosion,
(4) biotic and abiotic degradation, and
(5) volatilization.

U.S. EPA (1990e) recommended Equation 5-2 to calculate *ks*:

$$ks = ksl + ksg + ksv \qquad \text{Equation 5-2}$$

where

ks = COPC soil loss constant due to all processes (yr^{-1})
ksl = COPC loss constant due to leaching (yr^{-1})
ksg = COPC loss constant due to biotic and abiotic degradation (yr^{-1})
ksv = COPC loss constant due to volatilization (yr^{-1})

We recommend using Equation 5-2A to calculate *ks*. We describe this equation further in Appendix B, Table B-1-2. Using Equation 5-2A is consistent with U.S. EPA (1994g), U.S. EPA (1994r), U.S. EPA (1998c) and NC DEHNR (1997).

Recommended Equation for Calculating: COPC Soil Loss Constant (*ks*)

$$ks = ksg + kse + ksr + ksl + ksv \qquad \text{Equation 5-2A}$$

where

ks = COPC soil loss constant due to all processes (yr^{-1})
ksg = COPC loss constant due to biotic and abiotic degradation (yr^{-1})
kse = COPC loss constant due to soil erosion (yr^{-1})
ksr = COPC loss constant due to surface runoff (yr^{-1})
ksl = COPC loss constant due to leaching (yr^{-1})
ksv = COPC loss constant due to volatilization (yr^{-1})

Section 5.2.2.1 discusses loss due to biotic and abiotic degradation (*ksg*). Section 5.2.2.2 discusses loss due to erosion (*kse*). Section 5.2.2.3 discusses loss due to surface runoff (*ksr*). Section 5.2.2.4 discusses Loss due to leaching (*ksl*). Section 5.2.2.5 discusses loss due to volatilization (*ksv*).

As highlighted in Section 5.2.1, using Equation 5-2A in Equations 5-1C and 5-1D assumes that you can define COPC loss using first-order reaction kinetics. First-order reaction rates depend on the concentration of one reactant (Bohn et al. 1985). The loss of a COPC by a first-order process depends only on the concentration of the COPC in the soil, and a constant fraction of the COPC is removed from the soil over time. Those processes that apparently exhibit first-order reaction kinetics without implying a mechanistic dependence on a first-order loss rate are termed "apparent first-order" loss rates (Sparks 1989). The assumption that COPC loss follows first-order reaction kinetics may be an oversimplification because—at various concentrations or under various environmental conditions—the loss rates from soil systems will resemble different kinetic expressions. However, at low concentrations, a first-order loss constant may be adequate to describe the loss of the COPC from soil (U.S. EPA 1998c).

COPC loss in soil can also follow zero or second-order reaction kinetics. Zero-order reaction kinetics are independent of reactant concentrations (Bohn et al. 1985). Zero-order loss rates describe processes in which the reactants are present at very high concentrations. Under zero-order kinetics, a constant amount of a COPC is lost from the soil over time, independent of its concentration. Processes that follow second-order reaction kinetics depend on the concentrations of two reactants or the concentration of one reactant squared (Bohn et al. 1985). The loss constant of a COPC following a second-order process can be contingent on its own concentration, or on both its concentration and the concentration of another reactant, such as an enzyme or catalyst.

Because COPC loss from soil depends on many complex factors, it may be difficult to model the overall rate of loss. In addition, because the physical phenomena that cause COPC loss can occur simultaneously, using Equation 5-2A might also overestimate loss rates for each process (Valentine 1986). We recommend, when possible, taking into account the common occurrence of all loss processes. It’s possible to derive combined rates of soil loss by these processes experimentally. U.S. EPA (1986c) presents values for some COPCs.

5.2.2.1 COPC Loss Constant Due to Biotic and Abiotic Degradation *(ksg)*

Soil losses resulting from biotic and abiotic degradation (*ksg*) are determined empirically from field studies and should be available in the literature (U.S. EPA 1998c). According to Lyman et al. (1982), it’s reasonable to assume that degradation rates follow first order kinetics in a homogenous media. You’re therefore able to relate the half-life of a compound to the degradation rate constant. Ideally, *ksg* is the sum of all biotic and abiotic rate constants in the soil media. Therefore, if the half-life of a compound (for all of the mechanisms of transformation) is known, you can calculate the degradation rate. However, literature sources don’t provide sufficient data for all such mechanisms, especially for soil. Earlier Agency guidance (U.S. EPA 1994g) recommended setting *ksg* for all COPCs other than 2,3,7,8-TCDD equal to zero. The HHRAP companion database presents our recommended values for this COPC-specific variable.

Recommended Values for:
COPC Loss Constant Due to Biotic and Abiotic Degradation (ksg)

COPC-Specific
(See the HHRAP companion database)

The rate of biological degradation in soils depends on the concentration and activity levels of the microbial populations in the soil, the soil conditions, and the COPC concentration (Jury and Valentine 1986). First-order loss rates often fail to account for the high variability of these features in a single soil system. However, using simple rate expressions may be appropriate at low chemical concentrations (e.g., nanogram per kilogram soil). A first-order dependence on chemical concentration may be reasonable at low chemical concentrations. The rate of biological degradation is COPC-specific, and depends on the complexity of the COPC and the usefulness of the COPC to the microorganisms. Some substances, rather than being used by the organisms as a nutrient or energy source, are simply degraded with other similar COPCs, which can be further utilized. Environmental- and COPC-specific factors that can limit the biodegradation of COPCs in the soil environment (Valentine and Schnoor 1986) include:

- availability of the COPC;
- nutrient limitations;
- toxicity of the COPC; and
- inactivation or nonexistence of enzymes capable of degrading the COPC.

Chemical degradation of organic compounds can be a significant mechanism for removing COPCs from soil (U.S. EPA 1998c). Hydrolysis and oxidation-reduction reactions are the primary chemical transformation processes occurring in the upper layers of soils (Valentine 1986). General rate expressions describing the transformation of some COPCs by all non-biological processes are available. These expressions are helpful when division into component reactions isn’t possible.

Hydrolysis in aqueous systems is characterized by three processes: acid-catalyzed, base-catalyzed, and neutral reactions. The overall rate of hydrolysis is the sum of the first-order rates of these processes (Valentine 1986). In soil systems, sorption of the COPC can increase, decrease, or not affect the rate of hydrolysis, as numerous studies cited in Valentine (1986) have shown. We recommend predicting the overall (i.e. total) rate of hydrolysis in soil by adding the rates in the soil and water phases. We recommend assuming that these rates are first-order reactions at a fixed pH (Valentine 1986). Lyman et al. (1982) describes methods for estimating these hydrolysis constants.

Organic and inorganic compounds also undergo oxidation-reduction (redox) reactions in the soil (Valentine 1986). Organic redox reactions involve the reacting molecules exchanging oxygen and hydrogen atoms. Inorganic redox reactions may involve the reactants exchanging atoms or electrons. In soil systems where the identities of oxidant and reductant species aren't known, you can acquire a first-order rate constant for describing loss by redox reactions (Valentine 1986). Redox reactions involving metals may promote losses from surface soils by making metals more mobile (e.g., leaching to subsurface soils).

5.2.2.2 COPC Loss Constant Due to Soil Erosion (*kse*)

U.S. EPA (1998c) recommended using Equation 5-3 to calculate the constant for soil loss resulting from erosion (*kse*).

$$kse = \frac{0.1 \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s} \cdot \frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)} \qquad \text{Equation 5-3}$$

where

kse = COPC soil loss constant due to soil erosion
0.1 = Units conversion factor (1,000 g-kg/10,000 cm^2-m^2)
X_e = Unit soil loss (kg/m^2-yr)
SD = Sediment delivery ratio (unitless)
ER = Soil enrichment ratio (unitless)
BD = Soil bulk density (g soil/cm^3 soil)
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (ml water/g soil)
θ_{sw} = Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3

We recommend using the Universal Soil Loss Equation (USLE) to calculate unit soil loss (X_e)(See Section 5.7.2). We describe soil bulk density (BD) in Section 5.2.4.2. We describe Soil mixing depth (Z_s) in Section 5.2.4.1. We describe soil volumetric water content (θ_{sw}) in Section 5.2.4.4. We discuss site-specific variables associated with Equation 5-3 further in Appendix B.

U.S. EPA (1994g and 1994r) recommended setting all *kse* values equal to zero. U.S. EPA (1994r) recommended setting *kse* equal to zero because contaminated soil erodes both onto and off of the site.

As in U.S. EPA (1994g and 1994r), we recommend setting *kse* equal to zero.

Recommended Value for: COPC Loss Constant Due to Erosion (*kse*)
0

For additional information on addressing *kse*, we recommend consulting U.S. EPA (1998c). We also further describe using *kse* values in Appendix B, Table B-1-3.

5.2.2.3 COPC Loss Constant Due to Runoff (*ksr*)

Earlier U.S. EPA guidance (1994g) recommended setting all *ksr* values equal to zero.

As in U.S. EPA (1994r; 1998c) and NC DEHNR (1997), we recommend using Equation 5-4 to calculate *ksr*. We further discuss using Equation 5-4 in Appendix B, Table B-1-4.

Recommended Equation for Calculating:
COPC Loss Constant Due to Runoff (*ksr*)

$$ksr = \frac{RO}{\theta_{sw} \cdot Z_s} \cdot \left(\frac{1}{1 + \left(Kd_s \cdot BD / \theta_{sw} \right)} \right) \qquad \text{Equation 5-4}$$

where

ksr = COPC loss constant due to runoff (yr^{-1})
RO = Average annual surface runoff from pervious areas (cm/yr)
θ_{sw} = Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (ml water/g soil)
BD = Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3

The average annual surface runoff from pervious surfaces (RO) is a site-specific water loss term discussed in Section 5.2.4.3. Section 5.2.4.4 describes soil volumetric water content (θ_{sw}). Section 5.2.4.1 discusses the depth of soil mixing (Z_s). Appendix A-2 explains how we recommend calculating the COPC-specific soil/water partition coefficient (Kd_s). Section 5.2.4.2 describes soil bulk density (*BD*).

5.2.2.4 COPC Loss Constant Due to Leaching (*ksl*)

Losses of soil COPCs due to leaching (*ksl*) depend on the amount of water available to generate leachate and soil properties such as bulk density, soil moisture, soil porosity, and soil sorption properties.

U.S. EPA (1990e) recommended using Equation 5-5 to calculate *ksl*.

$$ksl = \frac{P + I - E_v}{\theta_{sw} \cdot Z_s \cdot [1.0 + (Kd_s \cdot BD / \theta_{sw})]} \qquad \text{Equation 5-5}$$

where

ksl = COPC loss constant due to leaching (yr^{-1})
P = Average annual precipitation (cm/yr)
I = Average annual irrigation (cm/yr)
E_v = Average annual evapotranspiration (cm/yr)
θ_{sw} = Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (ml water/g soil)
BD = Soil bulk density (g soil/cm^3 soil)

U.S. EPA (1993f) determined that Equation 5-5 does not properly account for surface runoff. U.S. EPA (1994g) recommended setting all *ksl* values to zero.

More recent guidance (U.S. EPA 1994r; 1998c; NC DEHNR 1997) have recommended using Equation 5-5A to calculate *ksl*. As with U.S. EPA (1994r), U.S. EPA (1998c), and NC DEHNR (1997), we recommend using Equation 5-5A to account for runoff while calculating *ksl*. We further discuss the use of this equation in Appendix B, Table B-1-5.

Recommended Equation for Calculating:
COPC Loss Constant Due to Leaching (*ksl*)

$$ksl = \frac{P + I - RO - E_v}{\theta_{sw} \cdot Z_s \cdot [1.0 + (BD \cdot Kd_s / \theta_{sw})]} \qquad \text{Equation 5-5A}$$

where

ksl	=	COPC loss constant due to leaching (yr^{-1})
P	=	Average annual precipitation (cm/yr)
I	=	Average annual irrigation (cm/yr)
RO	=	Average annual surface runoff from pervious areas (cm/yr)
E_v	=	Average annual evapotranspiration (cm/yr)
θ_{sw}	=	Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3
Z_s	=	Soil mixing zone depth (cm)
BD	=	Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3
Kd_s	=	Soil/water partition coefficient (cm^3 water/g soil)

Appendix B describes how we suggest acquiring site-specific variables associated with Equation 5-5A. The average annual volume of water available to generate leachate is the mass balance of all water inputs and outputs from the area under consideration ($P + I - RO - E_v$). These variables are described in Section 5.2.4.3. Section 5.2.4.4 describes soil volumetric water content (θ_{sw}). Section 5.2.4.1 describes the soil mixing depth (Z_s). Section 5.2.4.2 soil bulk density (BD). Appendix A-2 describes how we recommend calculating the COPC-specific soil/water partition coefficient (Kd_s).

5.2.2.5 COPC Loss Constant Due to Volatilization (*ksv*)

Semi-volatile and volatile COPCs emitted in high concentrations may become adsorbed to soil particles and exhibit volatilization losses from soil. The loss of a COPC from the soil by volatilization depends on the rate of movement of the COPC to the soil surface, the chemical vapor concentration at the soil surface, and the rate at which vapor is carried away by the atmosphere (Jury 1986).

U.S. EPA (1990e; 1993f; 1998c) recommended using Equation 5-6 to calculate *ksv*.

$$ksv = Ke \cdot Kt \qquad \text{Equation 5-6}$$

where

ksv	=	COPC loss constant due to volatilization (yr^{-1})
Ke	=	Equilibrium coefficient (s/cm-yr)
Kt	=	Gas phase mass transfer coefficient (cm/s)

U.S. EPA (1990e; 1993f; 1998c) don't identify a reference for Equation 5-6. U.S. EPA (1993f) stated that Equation 5-6 had been independently verified as accurately representing volatilization loss, but that the equation for *Kt* (Equation 5-8) appeared to fit to data empirically. U.S. EPA (1993f) also stated that *ksv* is modeled as a means of limiting soil concentration. Because this mass flux never experiences rain out, or washout and subsequent re-deposit, soil COPC concentrations are underestimated for soluble volatile COPCs. U.S. EPA (1993f) further recommended that additional research be conducted to determine the magnitude of the uncertainty introduced for volatile COPCs. U.S. EPA (1998c) recommended not considering the volatilized residues of semi-volatile COPCs (such as dioxin). U.S. EPA (1994g) recommended setting all *ksv* values to zero.

U.S. EPA guidance (1994r) and NC DEHNR (1997) recommended using Equation 5-6A to calculate *ksv*. Equation 5-6A appears to incorporate equations that U.S. EPA (1990e) recommended for calculating *Ke* (equilibrium coefficient) and *Kt* (gas phase mass transfer coefficient).

$$ksv = \left(\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD} \right) \cdot \left(0.482 \cdot W^{0.78} \cdot \left[\frac{\mu_a}{\rho_a \cdot D_a} \right]^{-0.67} \cdot \left[\sqrt{\frac{4 \cdot A}{\pi}} \right]^{-0.11} \right) \qquad \text{Equation 5-6A}$$

where

ksv = COPC loss constant due to volatilization (yr^{-1})
3.1536×10^7 = Units conversion factor (s/yr)
H = Henry's Law constant (atm-m^3/mol)
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (cm^3 water/g soil)
R = Universal gas constant (atm-m^3/mol-K)
T_a = Ambient air temperature (K) = 298.1 K
BD = Soil bulk density (g soil/cm^3 soil)
0.482 = Empirical constant (unitless) Units conversion factor $[(3600 \text{ s/hr})^{0.78}(100 \text{ cm/m})/(3600 \text{ s/hr})] \cdot$ (empirical constant 0.0292)
W = Average annual wind speed (m/s)
0.78 = Empirical constant (unitless)
μ_a = Viscosity of air (g/cm-s)
ρ_a = Density of air (g/cm^3)
D_a = Diffusivity of COPC in air (cm^2/s)
-0.67 = Empirical constant (unitless)
A = Surface area of contaminated area (m^2)
-0.11 = Empirical constant (unitless)

U.S. EPA (1990e) recommended using Equation 5-7 to calculate *Ke* and Equation 5-8 to calculate *Kt*.

$$Ke = \frac{3.1536 \times 10^7 \cdot (H \times 10^3)}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD} \qquad \text{Equation 5-7}$$

$$Kt = 0.482 \cdot W^{0.78} \cdot Sc_a^{-0.67} \cdot d_e^{-0.11} \qquad \text{Equation 5-8}$$

where

Ke = Equilibrium coefficient (s/cm-yr)
3.1536×10^7 = Units conversion factor (s/yr)
H = Henry's Law constant (atm-L/mol)
10^3 = Units conversion factor (L/m^3)
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (cm^3 water/g soil)
R = Universal gas constant (atm-m^3/mol-K)
T_a = Ambient air temperature (K) = 298.1 K
BD = Soil bulk density (g soil/cm^3 soil)
Kt = Gas phase mass transfer coefficient (cm/s)
0.482 = Units conversion factor [(3600 s/hr)$^{0.78}$(100 cm/m)/(3600 s/hr)] · (empirical constant 0.0292)
W = Average annual wind speed (m/s)
Sc_a = Schmidt number for gas phase (unitless)
d_e = Effective diameter of contaminated media (m)

U.S. EPA (1990e) also recommended using Equation 5-9 to calculate the Schmidt number for gas phase (Sc_a), and Equation 5-10 to calculate the effective diameter of contaminated media (d_e).

$$Sc_a = \frac{\mu_a}{\rho_a \cdot D_a} \qquad \text{Equation 5-9}$$

$$d_e = \sqrt{\frac{4 \cdot A}{\pi}} \qquad \text{Equation 5-10}$$

where

Sc_a = Schmidt number for gas phase (unitless)
μ_a = Viscosity of air (g/cm-s)
ρ_a = Density of air (g/cm^3)
D_a = Diffusivity of COPC in air (cm^2/s)
d_e = Effective diameter of contaminated media (m)
A = Surface area of contaminated area (m^2)

As in U.S. EPA (1998c), we recommend using Equation 5-7A to calculate *ksv*, in cases where high concentrations of volatile organic compounds are expected to be present in the soil.

Recommended Equation for Calculating:
COPC Loss Constant Due to Volatilization (*ksv*)

$$ksv = \left(\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right) \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \left(\frac{BD}{\rho_{soil}}\right) - \Theta_{sw}\right] \qquad \text{Equation 5-7A}$$

where

ksv = COPC loss constant due to volatilization (yr^{-1})
3.1536×10^{7} = Units conversion factor (s/yr)
H = Henry's Law constant ($\text{atm-m}^3/\text{mol}$)
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil/water partition coefficient (ml/g)
R = Universal gas constant ($\text{atm-m}^3/\text{mol-K}$)
T_a = Ambient air temperature (K) = 298.1 K
BD = Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3
D_a = Diffusivity of COPC in air (cm^2/s)
ρ_{soil} = Solids particle density (g/cm^3) = 2.7 g/cm^3
θ_{sw} = Soil volumetric water content (ml/cm^3 soil) = 0.2 ml/cm^3

Henry's Law constants are compound-specific, and we supply recommended values in the HHRAP companion database. We describe the soil mixing depth (Z_s) in. Appendix A-2 describes how we recommend calculating the COPC-specific soil/water partition coefficient (Kd_s). The Universal gas constant (R) and ambient air temperature (T_a) are discussed further in Appendix B, Table B-1-6. Soil bulk density (*BD*) is described below, as well as in Section 5.2.4.2. Appendix A-2 discusses the diffusivity of a COPC in air (D_a). Solids particle density (ρ_{soil}) is discussed in this Section, below. Soil volumetric water content (θ_{sw}) is further described below, as well as in Section 5.2.4.4.

Equation 5-7A is based on gas equilibrium coefficients and gas phase mass transfer, and combines Equations 5-7, 5-7B, and 5-7C. You can derive *ksv* by adapting the Hwang and Falco (1986) equation for soil vapor phase diffusion, to obtain Equation 5-6, as previously reported by U.S. EPA (1990e). Based on general soil properties, you can also write the gas-phase mass transfer coefficient, K_t, as follows (Hillel 1980; Miller and Gardiner 1998):

$$K_t = \frac{D_a \, \theta_v}{Z_s} \qquad \text{Equation 5-7B}$$

where

K_t = Gas phase mass transfer coefficient (cm/s)
Z_s = Soil mixing zone depth (cm)
D_a = Diffusivity of COPC in air (cm^2/s)
θ_v = Soil void fraction (cm^3/cm^3)

We describe Soil mixing depth (Z_s) in Section 5.2.4.1. The soil void fraction (θ_v) is the volumetric fraction of a soil that does not contain solids or water, and can be expressed as:

$$\theta_v = 1 - \left(\frac{BD}{\rho_s}\right) - \theta_{SW} \qquad \text{Equation 5-7C}$$

where
θ_v = Soil void fraction (cm^3/cm^3)
BD = Soil bulk density (g/cm^3) = 1.5 g/cm^3
ρ_{soil} = Solids particle density (g/cm^3) = 2.7 g/cm^3
θ_{sw} = Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3

The expression containing bulk density (*BD*) divided by solids particle density (ρ_{soil}) gives the volume of soil occupied by pore space or voids (Miller and Gardiner 1998). Soil bulk density is affected by the soil structure, such as looseness or compactness of the soil, and depends on the water and clay content of the soil (Hillel 1980). A range for bulk density of 0.83 to 1.84 was originally cited in Hoffman and Baes (1979). Blake and Hartge (1996) and Hillel (1980) both suggest that the mean density of solid particles is about 2.7 g/cm^3. We recommend a default soil bulk density of 1.5 g/cm^3, based on a mean value for loam soil from Carsel et al. (1988).

The soil water content (θ_{sw}) depends on both the available water and the soil structure of a particular soil. Values for θ_{sw} range from 0.03 to 0.40 ml/cm^3 depending on soil type (Hoffman and Baes 1979). The lower values are typical of sandy soils, which can't retain much water; the higher values are typical of soils such as clay or loam soils which can retain water. If site-specific information isn't available, we recommend a mid-point default value of 0.2 ml water/cm^3 soil. However, since θ_{sw} is unique for each soil type, we highly recommend using site-specific information.

We discuss *ksv* further in Appendix B, Table B-1-6.

5.2.3 Calculating the Deposition Term (*Ds*)

We recommend using Equation 5-11 to calculate the deposition term *(Ds)*. This equation is further described in Appendix B, Table B-1-1. Using Equation 5-11 to calculate *Ds* is consistent with U.S. EPA (1994r) and NC DEHNR (1997), which both incorporate *Ds* into Equation 5-1C.

Recommended Equation for Calculating:
Deposition Term (*Ds*)

$$Ds = \left[\frac{100 \cdot Q}{Z_s \cdot BD}\right] \cdot [F_v \cdot (Dydv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)] \qquad \text{Equation 5-11}$$

where

Ds = Deposition term (mg COPC/kg soil/yr)
100 = Units conversion factor (mg-m^2/kg-cm^2)
Q = COPC emission rate (g/s)
Z_s = Soil mixing zone depth (cm)
BD = Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3
F_v = Fraction of COPC air concentration in vapor phase (unitless)
$Dydv$ = Unitized yearly average dry deposition from vapor phase (s/m^2-yr)
$Dywv$ = Unitized yearly average wet deposition from vapor phase (s/m^2-yr)
$Dydp$ = Unitized yearly average dry deposition from particle phase (s/m^2-yr)
$Dywp$ = Unitized yearly average wet deposition from particle phase (s/m^2-yr)

Chapters 2 and 3 explain how we suggest quantifying the COPC emission rate (*Q*). Chapter 3 describes generating modeled air parameters *Cyv*, *Dydv, Dywv*, *Dydp*, and *Dywp*. We describe the soil mixing depth (Z_s) in Section 5.2.4.1. Soil bulk density (*BD*) is described in Sections 5.2.2.5 and 5.2.4.2, as well as Appendix B. Appendix A-2 describes how we suggest determining the COPC-specific parameter *Fv*.

5.2.4 Site-Specific Parameters for Calculating Cumulative Soil Concentration

Calculating *Cs* requires the following site-specific parameters:

- Soil mixing zone depth (Z_s)
- Soil bulk density (*BD*)
- Available water ($P + I - RO - E_v$)
- Soil volumetric water content (θ_{sw})

We discuss these parameters further in the following subsections, and in Appendix B.

5.2.4.1 Soil Mixing Zone Depth (Z_s)

When modeling exposures to COPCs in soils, the depth of contaminated soils is important in calculating the appropriate soil concentration. Tilling might mix deposited COPCs deeper into the soil, whether manually in a garden or mechanically in a large field. Increasing the volume of soil through which COPCs are mixed will tend to decrease (i.e. dilute) concentrations. The value of Z_s you choose may affect the outcome of the risk assessment, because soil concentrations that are based on soil depth are used to calculate exposure via several pathways:

- ingestion of plants contaminated by root uptake;
- direct ingestion of soil by humans, cattle, swine, or chickens; and
- surface runoff into water bodies.

For example, in calculations of exposures resulting from uptake through plant roots, the average concentration of COPCs over the depth of the plant root determines plant uptake.

In general, U.S. EPA (1992d, 1998c) estimated that if the area under consideration is likely to be tilled, soil depth is about 10 to 20 centimeters, depending on local conditions and the equipment used. If soil is not moved, COPCs were assumed to be retained in the shallower, upper soil layer. In this case, earlier Agency guidance (U.S. EPA 1990e; U.S. EPA 1998c) typically recommended a value of 1 centimeter.

U.S. EPA (1998c) recommended selecting Z_s as follows:

Soil Depth (Z_s)	Exposure	Description
1 cm	Direct ingestion of soil	Human exposure: in gardens, lawns, landscaped areas, parks, and recreational areas. Animal exposure: in pastures, lawns, and parks (untilled soils).
1 cm	Surface water runoff in nonagricultural areas	These areas are typically assumed to be untilled.
20 cm	Plant uptake for agricultural soils	The root depth is assumed to equal the tilling depth of 20 centimeters. In untilled soils, the root zone does not directly reflect tilling depth, although it is assumed that tilling depth is an adequate substitute for root zone depth.
20 cm	Surface water runoff in agricultural areas	These areas are typically assumed to be tilled.

We recommend the following values for Z_s:

Recommended Values for: Soil Mixing Zone Depth (Z_s)
2 cm - untilled 20 cm - tilled

We recommend a default Z_s of 2 cm for estimating surface soil concentrations in untilled soils, based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). We recommend a default Z_s of 20 cm for estimating surface soil concentrations in tilled soils, as in U.S. EPA (1998c).

5.2.4.2 Soil Dry Bulk Density (*BD*)

BD is the ratio of the mass of soil to its total volume. This variable is affected by the soil structure, type, and moisture content (Hillel 1980).

U.S. EPA (1994r) recommended deriving wet soil bulk density by weighing a thin-walled, tube soil sample (e.g., a Shelby tube) of known volume and subtracting the tube weight (ASTM Method D2937). Moisture content can then be calculated (ASTM Method 2216) to convert wet soil bulk density to dry soil bulk density.

As in U.S. EPA (1994g; 1998c) and presented in Hoffman and Baes (1979), we recommend the following value for *BD*:

Recommended Value for: Soil Dry Bulk Density (*BD*)
1.50 g/cm^3

5.2.4.3 Available Water ($P + I - RO - E_v$)

The average annual volume of water available ($P + I - RO - E_v$) for generating leachate is the mass balance of all water inputs and outputs from the area under consideration. A wide range of values for these site-specific parameters could apply in the various Agency regions.

The average annual precipitation (P), irrigation (I), runoff (RO), and evapotranspiration (E_v) rates and other climatological data are available from either data recorded on site or from the Station Climatic Summary for a nearby airport.

Meteorological variables—such as the evapotranspiration rate (E_v) and the runoff rate (RO)—might also be found in resources such as Geraghty et al. (1973). You could also estimate surface runoff by using the Curve Number Equation developed by the U.S. Soil Conservation Service (NC DEHNR 1997). U.S. EPA (1985b) cited isopleths of mean annual cropland runoff corresponding to various curve numbers developed by Stewart et al. (1975). Curve numbers were assigned to an area on the basis of soil type, land use or cover, and the hydrologic conditions of the soil (NC DEHNR 1997).

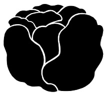
The wide range of available values, however, demonstrates the uncertainties and limitations in our ability to estimate these parameters. For example, Geraghty et al. (1973) presented isopleths for annual surface water contributions that include interflow and ground water recharge. U.S. EPA (1994g) recommended reducing these values by 50 percent, to represent surface runoff only.

5.2.4.4 Soil Volumetric Water Content (θ_{sw})

The soil volumetric water content (θ_{sw}) depends on the available water and the soil structure. A wide range of values for these variables may apply in the various Agency regions. As in earlier guidance documents, (U.S. EPA 1993i; U.S. EPA 1994g; NC DEHNR 1997), we recommend using a default value of 0.2 ml/cm^3 for θ_{sw}.

Recommended Value for: **Soil Volumetric Water Content (θ_{sw})**
0.2 ml/cm^3

5.3 CALCULATING COPC CONCENTRATIONS IN PRODUCE



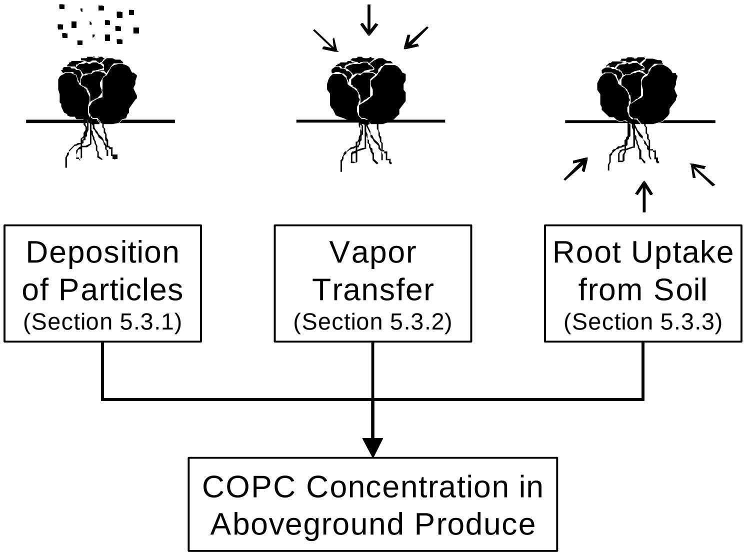
Indirect exposure resulting from ingestion of produce depends on the total concentration of COPCs in the leafy, fruit, and tuber portions of the plant. Because of general differences in contamination mechanisms, we recommend separating produce into two broad categories—aboveground produce and belowground produce. In addition, aboveground produce can be further subdivided into exposed and protected aboveground produce.

Aboveground Produce

Aboveground exposed produce is typically assumed to be contaminated by three possible mechanisms:

- **Direct deposition of particles—**wet and dry deposition of particle phase COPCs on the leaves and fruits of plants (Section 5.3.1).
- **Vapor transfer—**uptake of vapor phase COPCs by plants through their foliage (Section 5.3.2).
- **Root uptake—**root uptake of COPCs available from the soil and their transfer to the aboveground portions of the plant (Section 5.3.3).

FIGURE 5-3
COPC CONCENTRATION IN PRODUCE



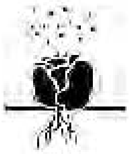
As in U.S.EPA (1998c), we recommend calculating the total COPC concentration in aboveground ***exposed*** produce as a sum of contamination occurring through all three of these mechanisms. However, edible portions of aboveground ***protected*** produce, such as peas, corn, and melons, are covered by a protective covering. They are therefore protected from contamination from deposition and vapor transfer. Root uptake of COPCs is the primary mechanism through which aboveground protected produce becomes contaminated (Section 5.3.3). Appendix B further describes the equations and parameters we recommend to calculate COPC concentrations in exposed and protected aboveground produce.

Belowground Produce

For belowground produce, we recommend assuming contamination occurs only through one mechanism—root uptake of COPCs available from soil (Section 5.3.3). The HHRAP doesn't address contamination of belowground produce via direct deposition of particles and vapor transfer because we assume that the root or tuber is protected from contact with contaminants in the vapor phase. Appendix B further describes the equations and parameters we recommend to calculate COPC concentrations in belowground produce.

Generally, we don't consider risks associated with exposure to VOCs via food-chain pathways significant. This is primarily because VOCs are typically low-molecular-weight COPCs that do not persist in the environment and do not bioaccumulate (U.S. EPA 1994r; U.S. EPA 1996g). However, as discussed in Chapter 2, we recommend evaluating all COPCs, including VOCs, for each exposure pathway.

5.3.1 Aboveground Produce Concentration Due to Direct Deposition (*Pd*)



Some earlier guidance documents (U.S. EPA 1990e; 1998c) recommended using Equation 5-13 to calculate COPC concentrations in aboveground vegetation resulting from wet and dry deposition onto plant surfaces of leafy plants and exposed produce (*Pd*):

Equation 5-13

where

Pd_i = Concentration of pollutant due to direct deposition in the *i*th plant group (µg COPC/g plant tissue DW))

1,000 = Units conversion factor (kg/10^3 g and 10^6 µg/g pollutant)

Dyd = Yearly dry deposition from particle phase (g/m^2-yr)

Fw = Fraction of COPC wet deposition that adheres to plant surfaces (unitless)

*Dyw*v = Yearly wet deposition from vapor phase (g/m^2-yr)

Rp_i = Interception fraction of the edible portion of plant tissue for the *i*th plant group (unitless)

kp = Plant surface loss coefficient (yr^{-1})

Tp_i = Length of plant's exposure to deposition per harvest of the edible portion of the *i*th plant group (yr)
Yp_i = Yield or standing crop biomass of edible portion of the *i*th plant group (kg DW/m^2)

U.S. EPA (1994r) modified Equation 5-13 to include stack emissions adjusted to remove the fraction of air concentration in vapor phase [Q (1 - F_v)] (Equation 5-14).

We recommend using Equation 5-14 to calculate *Pd*. We further discuss the use of this equation in Appendix B, Table B-2-7.

Recommended Equation for Calculating:
Aboveground Produce Concentration Due to Direct Deposition (*Pd*)

$$Pd = \frac{1{,}000 \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - \exp(-kp \cdot Tp)]}{Yp \cdot kp} \quad \text{Equation 5-14}$$

where
Pd = Plant (aboveground produce) concentration due to direct (wet and dry) deposition (mg COPC/kg DW)
1,000 = Units conversion factor (mg/g)
Q = COPC emission rate (g/s)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
$Dydp$ = Unitized yearly average dry deposition from particle phase (s/m^2-yr)
Fw = 0.2 for anions, 0.6 for cations & most organics (unitless)
$Dywp$ = Unitized yearly wet deposition from particle phase (s/m^2-yr)
Rp = Interception fraction of the edible portion of plant (unitless)
kp = Plant surface loss coefficient (yr^{-1})
Tp = Length of plant exposure to deposition per harvest of the edible portion of the *i*th plant group (yr)
Yp = Yield or standing crop biomass of the edible portion of the plant (productivity) (kg DW/m^2)

Chapters 2 and 3 explain how we recommend quantifying the COPC emission rate (*Q*). Appendix A-2 describes how we recommend determining the COPC-specific parameter *Fv*. Chapter 3 describes how the modeled air parameters *Dydp* and *Dywp* are generated. Appendix B explains our recommendations for *Fw*. *Rp*, *kp*, *Tp*, and *Yp* are neither site- nor COPC-specific, and are described in Sections 5.3.1.1 through 5.3.1.4.

5.3.1.1 Interception Fraction of the Edible Portion of Plant (*Rp*)

U.S. EPA (1998c) stated that NRC models assumed a constant of 0.2 for *Rp* for dry and wet deposition of particles (Boone et al. 1981). However, Shor et al. (1982) suggested that diversity of plant growth necessitated vegetation-specific *Rp* values.

As summarized in Baes et al. (1984), experimental studies of pasture grasses identified a correlation between initial *Rp* values and productivity (standing crop biomass [*Yp*]) (Chamberlain 1970):

$$Rp = 1 - e^{-\gamma Yp} \qquad \text{Equation 5-14A}$$

where

Rp = Interception fraction of the edible portion of plant (unitless)
γ = Empirical constant (Chamberlain [1970] gives the range as 2.3 to 3.3 for pasture grasses; Baes et al. [1984] used the midpoint, 2.88, for pasture grasses.)
Yp = Standing crop biomass (productivity) (kg DW/m^2 for silage; kg WW/m^2 for exposed produce)

Baes et al. (1984) also developed methods for estimating *Rp* values for leafy vegetables, silage, and exposed produce. However, these vegetation class-specific calculations produced *Rp* values that were independent of productivity measurements. This independence led to potentially unreasonable estimates of surface plant concentrations. Therefore, Baes et al. (1984) proposed using the same empirical relationship developed by Chamberlain (1970) for other vegetation classes. Baes et al. (1984) developed class-specific estimates of the empirical constant (γ) by forcing an exponential regression equation through several points. Points included average and theoretical maximum estimates of *Rp* and *Yp*. The following class-specific empirical constants (γ) were developed:

- Exposed produce = 0.0324
- Leafy vegetables = 0.0846
- Silage = 0.769

U.S. EPA (1994r) and U.S. EPA (1995e) proposed a default aboveground produce *Rp* value of 0.05, based on a weighted average of class-specific *Rp* values. Specifically, class-specific *Rp* values were calculated using the equation developed by Chamberlain (1970) and the following empirical constants:

- Leafy vegetables were assigned the same empirical constant (0.0846) developed by Baes et al. (1984).

- Fruits, fruiting vegetables, and legumes were assigned the empirical constant (0.0324) originally developed by Baes et al. (1984) for "exposed produce."

Vegetables and fruits included in each class are as follows:

- Fruits—apple, apricot, berry, cherry, cranberry, grape, peach, pear, plum/prune, and strawberry
- Fruiting Vegetables—asparagus, cucumber, eggplant, sweet pepper, and tomato
- Legumes—snap beans
- Leafy Vegetables—broccoli, brussel sprouts, cauliflower, celery, lettuce, and spinach

The class-specific *Rp* values were then weighted by relative ingestion (by humans) of each class, to determine a weighted average *Rp* value of 0.05. However, the produce classes and relative ingestion values used by U.S. EPA (1994r) and U.S. EPA (1995e) to calculate and weight the *Rp* values are not current with the U.S. EPA *1997 Exposure Factors Handbook* (U.S. EPA 1997b). In addition, the overall *Rp* value presented in U.S. EPA (1994r; 1995e) was based on limited information; subsequent revision to U.S. EPA (1994r; 1995e) resulted in an overall *Rp* value of 0.2 (RTI 1997).

For purposes of consistency, we combined the produce classes into two groups—exposed fruit and exposed vegetables. We used the exposed produce empirical constant (γ) to calculate *Rp*. Since the exposed vegetable category includes leafy and fruiting vegetables, we calculated *Rp* for leafy and fruiting vegetables. We then calculated the exposed vegetable *Rp* by a weighted average based on productivity (*Yp*) of leafy and fruiting vegetables, respectively. The relative ingestion rates used to determine an average weighted *Rp* value we derived from the intake of homegrown produce discussion presented in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997b). We recommend using the weighted average *Rp* value of 0.39 as a default *Rp* value, because it represents the most current parameters, including standing crop biomass and relative ingestion rates.

Recommended Value for: **Interception Fraction of the Edible Portion of Plant (*Rp*)**
0.39

Unweighted *Rp* and ingestion rates used for the weighting are as follows:

Aboveground Produce Class	*Rp*	Ingestion Rate (g DW/kg-day)
Exposed fruits	0.053	0.19
Exposed vegetables	0.982	0.11

One of the primary uncertainties associated with this variable is whether the algorithm developed by Chamberlain (1970) and the empirical constants developed by Baes et al. (1984) for use in this algorithm accurately represent aboveground produce. Specifically, Chamberlain (1970) based his algorithm on studies of pasture grass rather than aboveground produce. Baes et al. (1984) noted that their approach to developing class-specific *Rp* values is "at best *ad hoc,"* but stated that this approach was justified, because the consequences of using *Rp* estimates that are independent of productivity are "serious."

5.3.1.2 Plant Surface Loss Coefficient (*kp*)

U.S. EPA (1998c) identified several processes—including wind removal, water removal, and growth dilution—that reduce the amount of contaminant that has deposited on plant surfaces. The term *kp* is a measure of the amount of contaminant that is lost to these physical processes over time. U.S. EPA (1998c) cited Miller and Hoffman (1983) for the following equation:

$$kp = \left(\frac{\ln 2}{t_{1/2}} \right) \cdot 365 \qquad \text{Equation 5-15}$$

where

kp = Plant surface loss coefficient (yr^{-1})
$t_{1/2}$ = Half-life (days)
365 = Units conversion factor (days/yr)

Miller and Hoffman (1983) reported half-life values ranging from 2.8 to 34 days for a variety of contaminants on herbaceous vegetation. These half-life values converted to *kp* values of 7.44 to 90.36 (yr^{-1}). U.S. EPA (1994r; 1998c) recommended a *kp* value of 18, based on a generic 14-day half-life corresponding to physical processes only. The 14-day half-life is approximately the midpoint of the range (2.8 to 34 days) estimated by Miller and Hoffman (1983).

Lacking experimental data supporting chemical- and/or site-specific values, we recommend using a default *kp* value of 18. This *kp* value is the midpoint of Miller and Hoffman's (1983) range of values. Based on this range (7.44 to 90.36), plant concentrations could range from about 1.8 times higher to about 48 times lower than the plant concentrations, based on a *kp* value of 18. If chemical- or site-specific data is available, you could also calculate site- and chemical-specific *kp* values using the equation in Miller and Hoffman (1983).

Recommended Value for: **Plant Surface Loss Coefficient (*kp*)**
18 yr^{-1}

The primary uncertainty associated with *kp* relates to its position as the sole surface loss term in Equation 5-14. As defined by Miller and Hoffman (1983) and U.S. EPA (1998c), *kp* only represents potential losses from the physical processes listed above, not ***all*** potential losses (e.g. chemical degradation). However, information regarding chemical degradation of contaminants on plant surfaces is limited. Including chemical degradation processes would decrease half-life values and thereby increase *kp* values. Note that effective plant concentration decreases as *kp* increases. Therefore, using a *kp* value that does not consider chemical degradation processes is protective.

In addition, there are uncertainties associated with the half-life values reported by Miller and Hoffman (1983) with regard to how accurately these values represent the behavior of risk assessment COPCs on aboveground produce. However, the relative impact of this second uncertainty is less than the omission of chemical degradation processes.

5.3.1.3 Length of Plant Exposure to Deposition per Harvest of Edible Portion of Plant (*Tp*)

U.S. EPA (1990e; 1993f; 1994r; 1998c), and NC DEHNR (1997) recommended treating *Tp* as a constant, based on the average period between successive hay harvests. Belcher and Travis (1989) estimated this period at 60 days (0.164 years), which represents the length of time that aboveground vegetation (in this case, hay) is exposed to contaminant deposition before being harvested. Calculate *Tp* as follows:

$$Tp = \frac{60\ days}{365\ days/yr} = 0.164\ yr \qquad \text{Equation 5-16}$$

where

Tp = Length of plant exposure to deposition per harvest of the edible portion of plant (yr)
60 = Average period between successive hay harvests (days)
365 = Units conversion factor (days/yr)

As in previous guidance, we recommend using a *Tp* value of 0.164 year.

Recommended Value for: **Length of Plant Exposure to Deposition per Harvest of Edible Portion of Plant (*Tp*)**
0.164 years

The primary uncertainty associated with using this value is that it is based on the growing season for hay rather than aboveground produce. The average period between successive hay harvests (60 days) may not reflect the length of the growing season or the period between successive harvests for aboveground produce at specific sites. To the extent that information documenting the growing season or period between successive harvests for aboveground produce is available, this information could be appropriate to estimate a site-specific *Tp* value. The greater the difference between site-specific *Tp* and our recommended value, the greater the effect on plant concentration estimates.

5.3.1.4 Standing Crop Biomass (Productivity) (*Yp*)

U.S. EPA (1998c) recommended that the best estimate of *Yp* is productivity, which Baes et al. (1984) and Shor et al. (1982) define as follows:

$$Yp = \frac{Yh_i}{Ah_i} \qquad \text{Equation 5-17}$$

where

Yh_i = Harvest yield of the *i*th crop (kg DW)
Ah_i = area planted to the *i*th crop (m^2)

U.S. EPA (1994r) and NC DEHNR (1997) recommended using this equation and calculated a *Yp* value of 1.6 for aboveground produce, based on weighted average *Yh* and *Ah* values for four aboveground produce classes (fruits, fruiting vegetables, legumes, and leafy vegetables). Vegetables and fruits included in each class were as follows:

- Fruits—apple, apricot, berry, cherry, cranberry, grape, peach, pear, plum/prune, and strawberry
- Fruiting Vegetables—asparagus, cucumber, eggplant, sweet pepper, and tomato
- Legumes—snap beans
- Leafy Vegetables—broccoli, brussel sprouts, cauliflower, celery, lettuce, and spinach.

Class-specific *Yp* values were estimated using U.S. average *Yh* and *Ah* values for a variety of fruits and vegetables for 1993 (USDA 1994a; USDA 1994b). *Yh* values were converted to dry weight using average class-specific conversion factors (Baes et al. 1984). U.S. EPA (1994r and 1995e) calculated class-specific *Yp* values and then used relative ingestion rates of each group to calculate the weighted average *Yp* value of 1.6. However, the produce classes and relative ingestion values used by U.S. EPA (1994r and1995e) to calculate and weight the *Yp* values are not current with the U.S. EPA 1997 *Exposure Factors Handbook*. In addition, overall *Yp* value presented in U.S. EPA (1994r and U.S. EPA 1995e) was based on limited information; subsequent revision to U.S. EPA (1994r and 1995e) has resulted in an overall *Yp* value of 1.7 (RTI 1997).

For consistency, we combined the produce classes into two groups—exposed fruit and exposed vegetables. We derived the exposed vegetable *Yp* summing *Yh* values for leafy and fruiting vegetables and dividing by the sum of *Ah* values for leafy and fruiting vegetables. We derived the relative ingestion rates used to calculate an overall average weighted *Yp* value from the homegrown produce discussions presented in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997b). We recommend using the weighted average *Yp* value of 2.24 as a default *Yp* value, because this value represents the most complete and thorough information available.

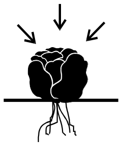
Recommended Value for: Standing Crop Biomass (Productivity) (*Yp*)
2.24

Unweighted *Yp* and ingestion rates used for the weighting are as follows:

Aboveground Produce Class	***Yp***	**Ingestion Rate (g DW/kg-day)**
Exposed fruits	0.25	0.19
Exposed vegetables	5.66	0.11

The primary uncertainty associated with this variable is that the harvest yield (*Yh*) and area planted (*Ah*) may not reflect site-specific conditions. To the extent to which site-specific information is available, you can estimate the magnitude of the uncertainty introduced by the default *Yp* value.

5.3.2 Aboveground Produce Concentration Due to Air-to-Plant Transfer (*Pv*)



The method we recommend for estimating COPC concentrations in exposed and aboveground produce due to air-to-plant transfer (*Pv*) was developed with consideration of items which might limit the transfer of COPC concentrations from plant surfaces to the inner portions of the plant. These limitations result from mechanisms responsible for

- inhibiting the transfer of lipophilic COPCs (e.g., the shape of the produce); and
- removing COPCs from the edible portion of the produce (e.g., washing, peeling, and cooking).

We recommend using Equation 5-18 to calculate *Pv*. We further discuss the use of this equation in Appendix B, Table B-2-8.

Recommended Equation for Calculating:
Aboveground Produce Concentration Due to Air-to-Plant Transfer (*Pv*)

$$Pv = Q \cdot F_v \cdot \frac{Cyv \cdot Bv_{ag} \cdot VG_{ag}}{\rho_a} \qquad \text{Equation 5-18}$$

where

Pv = Concentration of COPC in the plant resulting from air-to-plant transfer (µg COPC/g DW)
Q = COPC emission rate (g/s)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
Cyv = Unitized yearly average air concentration from vapor phase (µg-s/g-m^3)
Bv_{ag} = COPC air-to-plant biotransfer factor ([mg COPC/g DW plant]/[mg COPC/g air]) (unitless)
VG_{ag} = Empirical correction factor for aboveground produce (unitless)
ρ_a = Density of air (g/m^3)

Chapters 2 and 3 explain how we recommend quantifying the COPC emission rate (Q). Appendix A-2 describes how we recommend determining the COPC-specific parameters Fv and Bv_{ag}. Chapter 3 describes generating the modeled air parameter Cyv. As discussed below in Section 5.3.2.1, the parameter VG_{ag} depends on the lipophilicity of the COPC. Appendix B further describes how we recommend using Equation 5-18, including calculating ρ_a.

5.3.2.1 Empirical Correction Factor for Aboveground Produce (VG_{ag})

The parameter VG_{ag} was incorporated into Equation 5-18 to address the potential to overestimate the transfer of lipophilic COPCs to the inner portions of bulky produce, such as apples. Because of the protective outer skin, size, and shape of bulky produce, transfer of lipophilic COPCs (log K_{ow} greater than 4) to the center of the produce is not as likely as for non-lipophilic COPCs. As a result, the inner portions will be less affected.

To address this issue, U.S. EPA (1994m) recommended an empirical correction factor (VG_{ag}) of 0.01 for lipophilic COPCs to reduce estimated vegetable concentrations. The factor of 0.01 is based on a similar correction factor ($VG_{rootveg}$) for belowground produce. $VG_{rootveg}$ was estimated for unspecified vegetables as follows:

$$VG_{rootveg} = \frac{M_{skin}}{M_{vegetable}}$$ Equation 5-19

where

$VG_{rootveg}$ = Correction factor for belowground produce (g/g)
M_{skin} = Mass of a thin (skin) layer of belowground vegetable (g)
$M_{vegetable}$ = Mass of the entire vegetable (g)

Assuming that the density of the skin and the whole vegetable are the same, this equation becomes a ratio of the volume of the skin to that of the whole vegetable. U.S. EPA (1994m) assumed that the vegetable skin is 0.03 centimeters, which is the leaf thickness of a broad-leaf tree, as was used in experiments conducted by Riederer (1990). Using this assumption, U.S. EPA (1994m) calculated $VG_{rootveg}$ values of 0.09 and 0.03 for carrots and potatoes, respectively.

Based on the work by Wipf et al. (1982), U.S. EPA (1994m) identified other processes—such as peeling, cooking, and cleaning—that further reduce the vegetable concentration. U.S. EPA (1994m) recommended a $VG_{rootveg}$ value of 0.01 for lipophilic COPCs. These are less than the estimates of 0.09 and 0.03 for the carrots and potatoes mentioned earlier, but greater than the estimate would be if the correction factor was adjusted for cleaning, washing, and peeling, as described by Wipf et al. (1982). Following this line of reasoning, U.S. EPA (1994m) recommended a lipophilic COPC VG_{ag} value of 0.01 for all aboveground produce except leafy vegetables. As with $VG_{rootveg}$, U.S. EPA (1994m) noted that assignment of this value is based on the consideration that it "should be less than estimated just based on surface volume to whole fruit volume ratios."

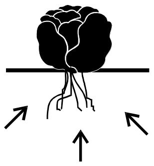
U.S. EPA (1994m) recommended a lipophilic COPC VG_{ag} of 1.0 for pasture grass because of a direct analogy to exposed azalea and grass leaves (for which data were available). Pasture grass is described as "leafy vegetation." However, the leafy vegetable group, as defined in Section 5.3.1.1, is composed of bulkier produce such as broccoli, brussel sprouts, cauliflower, celery, lettuce, and spinach. In addition, the outer leaves of most of the produce in this category are removed during preparation. Therefore, the VG_{ag} value of 1.0 for leafy vegetables is inappropriate and may overestimate COPC concentrations. A default lipophilic COPC VG_{ag} value of 0.01 for leafy vegetables is more appropriate for leafy vegetables, because the leafy vegetable category represents bulkier, more protected plants as compared to single leaves of grass blades. U.S. EPA (1994r) and NC DEHNR (1997) recommended a lipophilic COPC VG_{ag} value of 0.01, for all classes of aboveground produce.

For COPCs with a log K_{ow} greater than 4, we recommend using a lipophilic VG_{ag} value of 0.01 for all aboveground exposed produce. For COPCs with a log K_{ow} less than 4, we recommend using a VG_{ag} value of 1.0, because we assume these COPCs pass more easily through the skin of produce.

Recommended Values for: **Empirical Correction Factor for Aboveground Produce (VG_{ag})**
0.01 for COPCs with a log K_{ow} greater than 4
1.0 for COPCs with a log K_{ow} less than 4

Uncertainty may be introduced by assuming VG_{ag} values for leafy vegetables (such as lettuce) and for legumes (such as snap beans). Assuming a VG_{ag} value of 0.01 for legumes and leafy vegetables may underestimate concentrations because these species often have a higher ratio of surface area to mass than other bulkier fruits and fruiting vegetables, such as tomatoes.

5.3.3 Produce Concentration Due to Root Uptake (*Pr*)



Root uptake of contaminants from soil may contribute to COPC concentrations in aboveground exposed produce, aboveground protected produce, and belowground produce. As in previous guidance (U.S. EPA 1994m; U.S. EPA 1994r; and U.S. EPA 1995e), we recommend using Equations 5-20A and 5-20B to calculate *Pr*. We discuss the use of these equations further in Appendix B.

Recommended Equation for Calculating: Produce Concentration Due to Root Uptake (*Pr*)

Exposed and protected aboveground produce:

$$Pr = Cs \cdot Br \qquad \text{Equation 5-20A}$$

Belowground produce:

$$Pr = \frac{Cs \cdot RCF \cdot VG_{rootveg}}{Kd_s \cdot 1\ kg/L} \qquad \text{Equation 5-20B}$$

where

Pr = Concentration of COPC in produce due to root uptake (mg/kg)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Br = Plant-soil bioconcentration factor for produce (unitless)
RCF = Root concentration factor (unitless)
$VG_{rootveg}$ = Empirical correction factor for belowground produce (unitless)
Kd_s = Soil/water partition coefficient (L/kg)

Appendix B and Section 5.2 explain how we recommend calculating Cs. Appendix A-2 describes how we recommend calculating the COPC-specific parameters Br, RCF, and Kd_s. Similar to VG_{ag} and as discussed in Section 5.3.2.1, $VG_{rootveg}$ is based on the lipophilicity of the COPC.

Equation 5-20A is based on the soil-to-aboveground plant transfer approach developed by Travis and Arms (1988). This approach is appropriate for evaluating exposed and protected aboveground produce; however, it might not be appropriate for soil-to-belowground plant transfers. For belowground produce, U.S. EPA (1994m) and U.S. EPA (1995e) recommended Equation 5-20B, which includes a root concentration factor (RCF) developed by Briggs et al. (1982). RCF is the ratio of COPC concentration in the edible root to the COPC concentration in the soil water. Since Briggs et al. (1982) conducted their experiments in a growth solution, in order to use this equation you must divide the COPC soil concentration (Cs) by the COPC-specific soil/water partition coefficient (Kd_s) (U.S. EPA 1994m).

As in U.S. EPA (1994m), we recommend using a $VG_{rootveg}$ value of 0.01 for lipophilic COPCs (log K_{ow} greater than 4) based on root vegetables like carrots and potatoes. A value of 0.01 appears to represent the most complete and thorough information available. For COPCs with a log K_{ow} less than 4, we recommend a $VG_{rootveg}$ value of 1.0.

Recommended Values for: **Empirical Correction Factor for Belowground Produce ($VG_{rootveg}$)**
0.01 for COPCs with a log K_{ow} greater than 4
1.0 for COPCs with a log K_{ow} less than 4

5.4 CALCULATING COPC CONCENTRATIONS IN BEEF AND DAIRY PRODUCTS

We generally recommend that you estimate COPC concentrations in beef tissue and milk products on the basis of the amount of COPCs that cattle are assumed to consume through their diet. The HHRAP assumes the cattle's diet consists of:

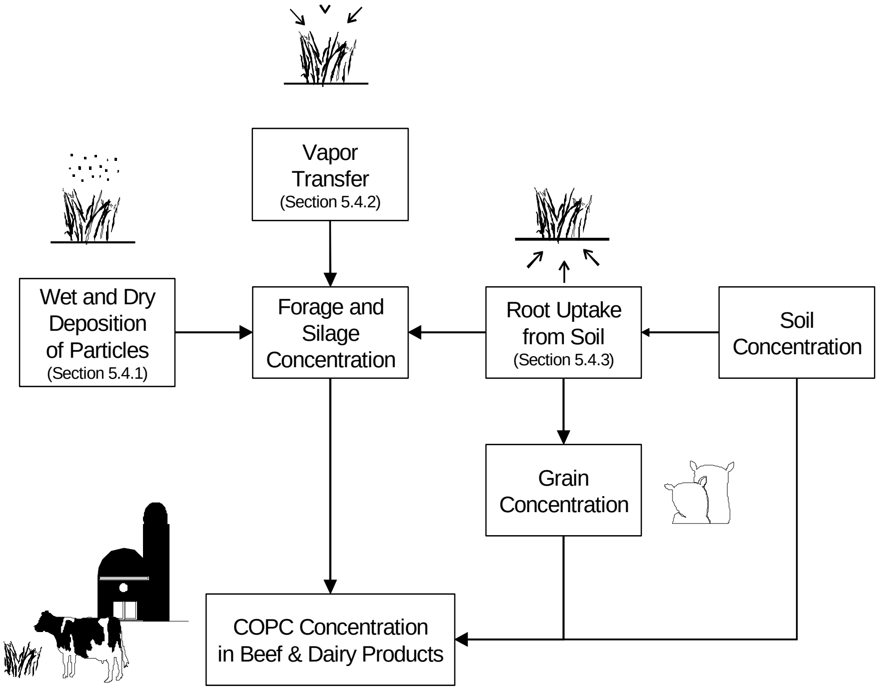
- forage (primarily pasture grass and hay),
- silage (forage that has been stored and fermented), and
- grain.

Additional contamination may occur through the cattle ingesting soil. The HHRAP calculates the total COPC concentration in the feed items (e.g., forage, silage, and grain) as a sum of contamination occurring through the following mechanisms:

- **Direct deposition of particles**—wet and dry deposition of particle phase COPCs onto forage and silage (Section 5.4.1).
- **Vapor transfer**—uptake of vapor phase COPCs by forage and silage through foliage (Section 5.4.2).
- **Root uptake**—root uptake of COPCs available from the soil and their transfer to the aboveground portions of forage, silage, and grain (Section 5.4.3).

Feed items consumed by animals can be classified as exposed or protected, depending on whether they have a protective outer covering. Because the outer covering on protected feed acts as a barrier, we assume that there is negligible contamination of protected feed through deposition of particles and vapor transfer. In the HHRAP, grain is classified as protected feed. As a result, we recommend that you assume grain contamination occurs only through root uptake. We also recommend assuming that contamination of exposed feed items, including forage and silage, occurs through all three mechanisms.

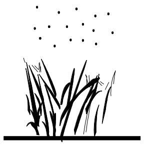
FIGURE 5-4
COPC CONCENTRATION IN BEEF AND DAIRY PRODUCTS



The HHRAP assumes that the amount of grain, silage, forage, and soil consumed varies between dairy and beef cattle. Sections 5.4.4 (beef) and 5.4.5 (dairy) describe the methods we recommend to estimate consumption rates and subsequent COPC concentrations in cattle. As in previous guidance (U.S. EPA 1990e and 1994a; NC DEHNR 1997), we recommend assuming that 100 percent of the plant materials eaten by cattle were grown on soil contaminated by emission sources. Therefore, we recommend assuming that 100 percent of the feed items are contaminated.

Appendix B, Tables B-3-1 through B-3-11, describe how we recommend calculating (1) the COPC concentrations in soil and feed items (forage, silage, and grain) consumed by beef and dairy cattle, and (2) the resulting COPC concentrations in beef and milk.

5.4.1 Forage and Silage Concentrations Due to Direct Deposition (*Pd)*



COPC concentrations in forage and silage result from wet and dry deposition onto exposed plant surfaces; similar to aboveground produce (Section 5.3.1). Therefore, we recommend also using Equation 5-14 to calculate *Pd* for forage and silage. We discuss calculating *Pd* for Forage and silage further in Appendix B. Appendix A-2 explains how we recommend calculating COPC-specific *Fv* values for forage and silage (i.e. exactly as they are calculated for aboveground produce). Sections 5.4.1.1 through 5.4.1.4 describe how we recommend calculating *Rp*, *kp*, *Tp*, and *Yp* for use in calculating forage and silage concentrations.

5.4.1.1 Interception Fraction of the Edible Portion of Plant (*Rp*)

As discussed in Section 5.3.1.1, Chamberlain (1970) found a correlation between *Rp* and productivity, *Yp* (standing crop biomass). This correlation is expressed in Equation 5-14A.

Based on U.S. EPA (1994r and 1995b) and NC DEHNR (1997), we recommend using Equation 5-14 to calculate *Rp* values for forage and silage.

Substituting the Baes et al. (1984) empirical constant (**(**) value of 2.88 for pasture grass, and the standing crop biomass value of 0.24 kg DW/m^2 (these variables are discussed in Section 5.3.1.1) into Equation 5-14, the forage-specific *Rp* is 0.5. Substituting the Baes et al. (1984) empirical constant (**(**) value of 0.769 for silage, and the standing crop biomass value of 0.8 kg DW/m^2 into Equation 5-14, the silage-specific *Rp* value is 0.46.

Recommended Value for: **Interception Fraction of the Edible Portion of Plant (*Rp*)**
Forage = 0.5 Silage = 0.46

Several uncertainties are associated with the *Rp* variable:

- The empirical relationship developed by Chamberlain (1970) is based on a study of pasture grass, and therefore may not accurately represent site-specific silage types.
- The empirical constant for silage developed by Baes et al. (1984) used in Chamberlain's empirical relationship may also fail to accurately represent site-specific silage types.
- The range of empirical constants recommended by Baes et al. (1984) for pasture grass does not result in a significant range of estimated *Rp* values for forage (the calculated *Rp* range is 0.42 to 0.54). Therefore, using the empirical constant midpoint (2.88 for pasture grass) does not significantly affect the *Rp* value and the resulting estimate of plant COPC concentration.

5.4.1.2 Plant Surface Loss Coefficient (*kp*)

We recommend using Equation 5-15 (Section 5.3.1.2) to calculate the plant surface loss coefficient *kp* for aboveground produce. The *kp* factor is derived in the same manner for cattle forage and silage. The uncertainties of *kp* for cattle forage and silage are similar to the uncertainties for aboveground produce.

5.4.1.3 Length of Plant Exposure to Deposition per Harvest of the Edible Portion of Plant (*Tp*)

As discussed in Section 5.3.1.3, the HHRAP treats *Tp* as a constant, based on the average period between successive hay harvests. This period, which Belcher and Travis (1989) estimated at 60 days, represents the length of time that aboveground vegetation (in this case, hay) would be exposed to particle deposition before being harvested. We used Equation 5-16 (Section 5.3.1.3), to calculate a *Tp* of 0.16 year for cattle silage.

For cattle forage, we modified Equation 5-16 to consider the average of :

1. the average period between successive hay harvests, and
2. the average period between successive grazing.

Based on Belcher and Travis (1989), the we assumed the average period between hay harvests is 60 days, and the average period between successive grazing is 30 days. We therefore calculated *Tp* as follows:

$$Tp = \frac{0.5 \cdot (60\ days + 30\ days)}{365\ days/yr} = 0.12\ yr \qquad \text{Equation 5-21}$$

Recommended Value for: **Plant Exposure Length to Deposition per Harvest of the Edible Portion of Plant (*Tp*)**
Forage = 0.12 yr Silage = 0.16 yr

The primary uncertainties associated with *Tp* are similar to those for aboveground produce, as discussed in Section 5.3.1.3.

5.4.1.4 Standing Crop Biomass (Productivity) (*Yp*)

As discussed in Section 5.3.1.4, U.S. EPA (1998c) stated that the best estimate of *Yp* is productivity, as defined in Equation 5-17. Consequently, under this approach, you would consider dry harvest yield (*Yh*) and area harvested (*Ah*).

We calculated forage *Yp* as a weighted average of the calculated pasture grass and hay *Yp* values. We assumed weightings of 0.75 for forage and 0.25 for hay. The weightings are based on the fraction of a year that cattle are assumed to be pastured and eating grass (9 months per year) or not pastured and fed hay (3 months per year). We assumed an unweighted pasture grass *Yp* of 0.15 kg DW/m^2 (U.S. EPA 1994r; U.S. EPA 1994m). We then calculated an unweighted hay *Yp* of 0.5 kg DW/m^2 using Equation 5-17 and the following *Yh* and *Ah* values:

Yh = 1.22×10^{11} kg DW, calculated from the 1993 U.S. average wet weight *Yh* of 1.35×10^{11} kg (USDA 1994b) and a conversion factor of 0.9 (Fries 1994).

Ah = 2.45×10^{11} m^2, the 1993 U.S. average for hay (USDA 1994b).

The unweighted pasture grass and hay *Yp* values were multiplied by their weighting factors (0.75 and 0.25, respectively), and summed to calculate the recommended weighted forage *Yp* of 0.24 kg DW/m^2.

We recommend assuming a production-weighted U.S. average *Yp* of 0.8 kg DW/m^2 for silage (Shor, et al. 1982).

Recommended Values for:
Standing Crop Biomass (Productivity) (*Yp*)

Forage = 0.24 kg DW/m^2
Silage = 0.8 kg DW/m^2

The primary uncertainty associated with this variable is that the harvest yield (*Yh*) and area planted (*Ah*) may not reflect site-specific conditions. To the extent that site-specific information is available, it's feasible to estimate the magnitude of the uncertainty introduced by the default *Yp* value. In addition, the weightings assumed in this discussion for the amount of time that cattle are pastured (and foraging) or stabled (and being fed silage) could be adjusted to reflect site-specific conditions, as appropriate.

5.4.2 Forage and Silage Concentrations Due to Air-to-Plant Transfer (*Pv*)

We recommend using Equation 5-18 (Section 5.3.2) to calculate the COPC concentration in aboveground produce resulting from air-to-plant transfer (*Pv*). *Pv* is calculated for cattle forage and silage similarly to the way that it's calculated for aboveground produce. We provide a detailed discussion of *Pv* in Section 5.3.2. We present differences in VG_{ag} values for forage and silage, as compared to the values for aboveground produce described in Section 5.3.2.1, in Section 5.4.2.1. We discuss calculating *Pv* further in Appendix B. Appendix A-2 explains how we recommend calculating COPC-specific *Bv* values for forage and silage (i.e. the same as they are calculated for aboveground produce).

5.4.2.1 Empirical Correction Factor for Forage and Silage (VG_{ag})

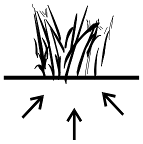
Please Section 5.3.2.1 for a detailed, general introduction to VG_{ag}. Using such a factor while estimating COPC concentrations specifically for forage and silage assumes that there is insignificant translocation of COPCs deposited on the surface of bulky silage to the inner parts of the vegetation. Applying a silage VG_{ag} would be relevant if the silage can't be characterized as leafy (e.g., if grain is used as silage). As a point of clarification, forage and silage are considered vegetative plant parts, and grains are considered reproductive plant parts.

U.S. EPA (1994m) didn't recommend a VG_{ag} value for silage. NC DEHNR (1997) recommended a VG_{ag} factor of 0.5 for bulky silage but didn't present a specific rationale for this value. U.S. EPA (1995b) noted that a volume ratio of outer whole surface area to volume of vegetation could be used to assign a silage VG_{ag} value, if one knew the proportions of each type of vegetation of which silage consisted. In the absence of specific data concerning the quantities of different silage material (e.g., hay and grain), U.S. EPA (1995b) recommended assuming a VG_{ag} of 0.5 for silage without rigorous justification.

We recommend using VG_{ag} values of 1.0 for forage and 0.5 for silage. As discussed, the primary uncertainty associated with this variable is the lack of specific information on the proportions of each vegetation type of which silage may consist, leading to the default assumption of 0.5.

Recommended Values for: **Empirical Correction Factor for Forage and Silage (VG_{ag})**
Forage = 1 Silage = 0.5

5.4.3 Forage, Silage, and Grain Concentrations Due to Root Uptake (*Pr*)



We recommend using Equations 5-20A and 5-20B (Section 5.3.3) to calculate the COPC concentration in aboveground and belowground produce resulting from root uptake. *Pr* is calculated for cattle forage, silage, and grain in the same way that it is calculated for aboveground produce, except that we recommend using forage/silage- and grain-specific bioconcentration factors (Br_{forage} and Br_{grain} respectively). Appendix A-2 explains how we recommend calculating COPC-specific Br values for forage and silage (i.e. exactly as it's calculated for aboveground produce). We provide a detailed discussion on how we recommend calculating Pr in Section 5.3.3. We further discuss the calculation of Pr in Appendix B.

5.4.4 Beef Concentration Resulting from Plant and Soil Ingestion (A_{beef})

As in U.S. EPA (1995h), we recommend using Equation 5-22 to calculate COPC concentration in beef tissue (A_{beef}). The equation was modified from an equation presented in U.S. EPA (1990c), U.S. EPA (1994r), U.S. EPA (1995b), and NC DEHNR (1996) by introducing a metabolism factor (MF). Equation 5-22 calculates the daily amount of a COPC that is consumed by cattle through the ingestion of contaminated feed items (plant) and soil. The equation includes biotransfer and metabolism factors to transform the daily animal intake of a COPC (mg/day) into an animal COPC tissue concentration (mg COPC/kg tissue). We further discuss using this equation in Appendix B, Table 3-10.

Recommended Equation for Calculating:
Concentration of COPC in Beef (A_{beef})

Equation 5-22

where

A_{beef} = Concentration of COPC in beef (mg COPC/kg FW tissue)
F_i = Fraction of plant type i grown on contaminated soil and ingested by the animal (cattle) (unitless)
Qp_i = Quantity of plant type i eaten by the animal (cattle) per day (kg DW plant/day)
P_i = Concentration of COPC in each plant type i eaten by the animal (cattle) (mg/kg DW)
Qs = Quantity of soil eaten by the animal (cattle) each day (kg/day)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Bs = Soil bioavailability factor (unitless)
Ba_{beef} = COPC biotransfer factor for beef (day/kg FW tissue)
MF = Metabolism factor (unitless)

Sections 5.4.4.1 through 5.4.4.7 describe the parameters F_i, Qp_i, P_i, Qs, Cs, Bs, and MF, respectively. Appendix A-2 explains how we recommend calculating the COPC-specific parameter Ba_{beef}.

5.4.4.1 Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Cattle)(F_i)

As in U.S. EPA (1990e and 1994r), and NC DEHNR (1997), we recommend assuming that 100 percent of the plant materials eaten by cattle were grown on soil contaminated by the emission sources being evaluated. This assumption translates to a default value of 1.0 for F_i.

Recommended Value for: **Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Cattle) (F_i)**
1

5.4.4.2 Quantity of Plant Type *i* Eaten by the Animal (Cattle) Each Day (Qp_i)

The daily quantity of plants eaten by cattle can be estimated (kg DW/day) for each category of plant feed. U.S. EPA (1994r and 1998c) and NC DEHNR (1997) recommended including forage, silage, and grain feeds in this estimate.

NC DEHNR (1997) recommended plant ingestion rates for the cattle of either subsistence beef farmers or typical beef farmers. Subsistence beef farmers rely on a higher percentage of forage and silage to feed cattle, whereas typical beef farmers rely on greater amounts of grain to feed cattle. U.S. EPA (1990e) and U.S. EPA (1994r) identified plant ingestion rates only for subsistence farmers. The following daily quantities of forage, grain, and silage eaten by cattle were recommended by NC DEHNR (1997), U.S. EPA (1994r and 1990e), and Boone et al. (1981):

Source	Forage (kg DW/day)	Grain (kg DW/day)	Silage (kg DW/day)	References
NC DEHNR (1997) Subsistence Farmer Beef Cattle	8.8	0.47	2.5	Boone et al. (1981) NAS (1987)
NC DEHNR (1997) Typical Farmer Beef Cattle	3.8	3.8	1.0	Rice (1994)
U.S. EPA (1994r) Subsistence Farmer Beef Cattle	8.8	Not reported	Not reported	Boone et al. (1981) NAS (1987)
U.S. EPA (1990e) Subsistence Farmer Beef Cattle	8.8	0.47	2.5	Boone et al. (1981) McKone and Ryan (1989)
Boone et al. (1981)	8.87	1.9	2.5	Boone et al. (1981)

With the exception of a higher grain ingestion rate, Boone et al. (1981) rates are consistent with those recommended by U.S. EPA (1990e and 1994r), and NC DEHNR (1997). For typical farmer beef cattle, NC DEHNR (1997) cites Rice (1994) as a reference for the Qp_i variables and notes that the values include grain supplemented during the growing phase for beef cattle.

U.S. EPA (1990e) noted that McKone and Ryan (1989) reported an average total ingestion rate of 12 kg DW/day for the three plant feeds, which is consistent with the total recommended by U.S. EPA (1990e) and NC DEHNR (1997) (forage, grain, and silage total of 11.8 kg DW/day). U.S. EPA (1994r) and NC DEHNR (1997) also noted that NAS (1987) reported a daily dry matter intake that is 2 percent of an average beef cattle body weight of 590 kilograms. This results in a daily total intake rate of 11.8 kg DW/day. NAS (1987) reported that a nonlactating cow eats dry matter equivalent to 2 percent of its body weight.

We recommend using the following beef cattle ingestion rates of forage, silage, and grain. These values are based on the total daily intake rate of about 12 kg DW/day.

Recommended Values for:
Quantity of Plant Type *i* Eaten by the Animal (Cattle) Each Day (Qp_i)

Forage = 8.8 kg DW/day

Silage = 2.5 kg DW/day

Grain = 0.47 kg DW/day

The principal uncertainty associated with Qp_i is the variability between forage, silage, and grain ingestion rates for cattle.

5.4.4.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Cattle) (P_i)

We generally recommend using Equation 5-23 to calculate the total COPC concentration in forage, silage, and grain. We recommend deriving values for *Pd*, *Pv*, and *Pr* for each type of feed by using Equations 5-14, 5-18, and 5-20, respectively.

Recommended Equation for Calculating:
Concentration of COPC in Plant Type *i* Eaten by the Animal (Cattle) (P_i)

$$P_i = \sum_i (Pd + Pv + Pr) \qquad \text{Equation 5-23}$$

where

P_i = Concentration of COPC in each plant type *i* eaten by the animal (mg COPC/kg DW)
Pd = Plant concentration due to direct deposition (mg COPC/kg DW)
Pv = Plant concentration due to air-to-plant transfer (mg COPC/kg DW)
Pr = Plant concentration due to root uptake (mg COPC/kg DW)

This equation is further described in Appendix B.

5.4.4.4 Quantity of Soil Eaten by the Animal (Cattle) Per Day (Qs)

Additional cattle contamination occurs through ingestion of soil.

NC DEHNR (1997) and U.S. EPA (1994r) recommended a soil ingestion rate for subsistence beef cattle of 0.5 kg/day. This rate is based on Fries (1994). U.S. EPA (1994r) and NC DEHNR (1997) noted that

Fries (1994) reported soil ingestion to be 4 percent of the total dry matter intake. NAS (1987) was also referenced. NAS (1987) cited an average beef cattle weight of 590 kg, and a daily dry matter intake rate (nonlactating cows) of 2 percent of body weight. This results in a daily dry matter intake rate of 11.8 kg DW/day and a daily soil ingestion rate of about 0.5 kg/day. U.S. EPA (1990e) reported a soil ingestion rate that is 3 percent of the forage intake rate of 8.8 kg DW/day, resulting in a daily soil ingestion rate of approximately 0.3 kg/day. Simmonds and Linsley (1981) and Thornton and Abrams (1983) were cited as the references for this assumption.

We recommend using 0.5 kg/day for the quantity of soil ingested by the animal (cattle).

Recommended Value for: **Quantity of Soil Ingested by the Animal (Cattle) Per Day (*Qs*)**
0.5 kg/day

5.4.4.5 Average Soil Concentration Over Exposure Duration (*Cs*)

We recommend using Equations 5-1C, 5-1D, and 5-1E to calculate the COPC concentration in soil as discussed in Section 5.2.1. Also, Appendix B further describes how we recommend calculating the soil concentration.

> ***Please Note:*** You might need to generate soil concentration estimates for grain separate from those for forage and silage. Currently, the HHRAP assumes that forage and silage are grown on untilled land, and grain is grown on tilled land. We highly recommend that your C_s calculations include the appropriate Z_s (1 for untilled land, 20 for tilled land).

5.4.4.6 Soil Bioavailability Factor (*Bs*)

The efficiency of transfer from soil may differ from the efficiency of transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, then the ratio would be less than 1.0. If it is equal to or greater than that of vegetation, the *Bs* value would be equal to or greater than 1.0.

Until more COPC-specific data becomes available for this parameter, we recommend a default value of 1 for *Bs*.

Recommended Values for: **Soil Bioavailability Factor (*Bs*)**
1.0

5.4.4.7 Metabolism Factor (*MF*)

The metabolism factor (*MF*) estimates the amount of COPC that remains in fat and muscle. Based on a study by Ikeda et al. (1980), U.S. EPA (1995h) used a COPC-specific *MF* to account for metabolism in animals and humans. Evidence indicates BEHP is more readily metabolized and excreted by mammalian species than other contaminants (ATSDR 1987). As in U.S. EPA (1995h), we recommend a *MF* of 0.01 for bis(2-ethylhexyl)phthalate (BEHP). Lacking data to support derivation of other chemical-specific *MF*s, we recommend using a *MF* of 1.0 for all chemicals other than BEHP. Using the recommended values for this variable, *MF* has no quantitative effect on A_{beef} (with the exception of BEHP).

Recommended Values for: **Metabolism Factor (*MF*)**
bis(2-ethylhexyl)phthalate (BEHP) = 0.01 All other COPCs = 1.0

The *MF* presented above for BEHP applies only to mammalian species, including beef cattle, dairy cattle, and pigs. It does not relate to metabolism in produce, chicken, or fish. In addition, since exposures evaluated in this guidance are intake driven, using an MF applies only to estimating COPC concentrations in food sources used in evaluating indirect human exposure, including ingestion of beef, milk, and pork. In summary, an *MF* is not applicable for direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish.

5.4.5 COPC Concentration In Milk Due to Plant and Soil Ingestion (A_{milk})

We recommend modifying Equation 5-22 (Section 5.4.4) to calculate COPC milk concentrations (A_{milk}), as follows:

Recommended Equation for Calculating:
Concentration of COPC in Milk (A_{milk})

$$A_{milk} = \left(\Sigma(F_i \cdot Qp_i \cdot P_i) + Qs \cdot Cs \cdot Bs\right) \cdot Ba_{milk} \cdot MF \qquad \text{Equation 5-24}$$

where

A_{milk} = Concentration of COPC in milk (mg COPC/kg milk)
F_i = Fraction of plant type *i* grown on contaminated soil and ingested by the animal (dairy cattle) (unitless)
Qp_i = Quantity of plant type *i* eaten by the animal (dairy cattle) each day (kg DW plant/day)
P_i = Concentration of COPC in plant type *i* eaten by the animal (dairy cattle) (mg/kg DW)
Qs = Quantity of soil eaten by the animal (dairy cattle) each day (kg soil/day)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Bs = Soil bioavailability factor (unitless)
Ba_{milk} = COPC biotransfer factor for milk (day/kg WW tissue)
MF = Metabolism factor (unitless)

Appendix A-2 explains how we recommend calculating the COPC-specific parameter Ba_{milk}. The discussion in Section 5.4.4 of the variables F_i, Qp_i, P_i, Qs, Cs, and MF for beef cattle generally applies to the corresponding variables for dairy cattle. However, there are some differences in assumptions made for dairy cattle; these differences are summarized in the following subsections.

We recommend using Equation 5-24 to estimate A_{milk}. Using Equation 5-24 is described further in Appendix B, Table B-3-11.

5.4.5.1 Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Dairy Cattle) (F_i)

The calculation of F_i for dairy cattle is identical to that for beef cattle (Section 5.4.4.1).

5.4.5.2 Quantity of Plant Type *i* Eaten by the Animal (Dairy Cattle) Per Day (Qp_i)

As discussed in Section 5.4.4.2, the daily quantity of forage, silage, and grain feed consumed by cattle is estimated for each category of feed material. However, daily ingestion rates for dairy cattle are estimated differently than for beef cattle. We generally recommend estimating the daily quantity of feed consumed by cattle on a dry weight basis for each category of plant feed.

NC DEHNR (1997) recommended using plant ingestion rates for either subsistence dairy farmer or typical dairy farmer cattle. In addition, subsistence dairy farmers rely on a higher percentage of forage and silage to feed cattle, whereas typical dairy farmers rely on greater amounts of grain to feed cattle. U.S. EPA (1990e and 1994r) identified plant ingestion rates only for subsistence farmers.

The following daily quantities of forage, grain, and silage eaten by dairy cattle were recommended by NC DEHNR (1997), U.S. EPA (1994r), U.S. EPA (1990e), and Boone et al. (1981):

Source	Forage (kg/day DW)	Grain (kg/day DW)	Silage (kg/day DW)	References
NC DEHNR (1997) Subsistence Dairy Farmer Cattle	13.2	3.0	4.1	Boone et al. (1981) NAS (1987)
NC DEHNR (1997) Typical Dairy Farmer Cattle	6.2	12.2	1.9	Rice (1994)
U.S. EPA (1994r) Subsistence Dairy Farmer Cattle	13.2	Not reported	Not reported	Boone et al. (1981) NAS (1987)
U.S. EPA (1990e) Subsistence Dairy Farmer Cattle	11.0	2.6	3.3	Boone et al. (1981) McKone and Ryan (1989)
Boone et al. (1981)	11.0	2.6	3.3	Boone et al. (1981)

U.S. EPA (1990e) noted that McKone and Ryan (1989) reports an average total ingestion rate of 17 kg/day DW for the three plant feeds, which is consistent with the total ingestion rate recommended by U.S. EPA (1990e). U.S. EPA (1994r) and NC DEHNR (1997) noted that NAS (1987) reports a daily dry matter intake that is 3.2 percent of an average dairy cattle body weight of 630 kilograms. This results in a daily total intake rate of approximately 20 kg/day DW, which is consistent with the average total ingestion rates for the three plant feeds recommended by U.S. EPA (1994r) and NC DEHNR (1997) . NAS (1987) reported that dairy cows eat dry matter equivalent to 3.2 percent of their body weight; the

630-kilogram average dairy cow body weight was not confirmed. U.S. EPA (1995b) also cited a feed ingestion rate of 20 kg/day DW, citing U.S. EPA (1993c).

Based on more recent references (NAS 1987; U.S. EPA 1993c) which recommend a feed ingestion rate of 20 kg/day DW, we recommend a default total ingestion rate of 20 kg DW/day for dairy cattle.

Recommended Values for: Quantity of Plant Type ***i*** Eaten by the Animal (Dairy Cattle) Per Day (***Qp_i***)
Forage = 13.2 kg DW/day
Silage = 4.1 kg DW/day
Grain = 3.0 kg DW/day

Uncertainties associated with estimating Qp_i include estimating forage, grain, and silage ingestion rates, which will vary from site to site. Assuming uniform contamination of plant materials consumed by cattle also introduces uncertainty.

5.4.5.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Dairy Cattle) (P_i)

The estimation of P_i for dairy cattle is identical to that for beef cattle (Section 5.4.4.3).

5.4.5.4 Quantity of Soil Eaten by the Animal (Dairy Cattle) Per Day (*Qs*)

As discussed in Section 5.4.4.4, contamination of dairy cattle also results from the ingestion of soil. We generally recommend the following soil ingestion rate for dairy cattle:

Recommended Values for: Quantity of Soil Eaten by the Animal (Dairy Cattle) Per Day (***Qs***)
0.4 kg/day

U.S. EPA (1994r) and NC DEHNR (1997) recommended a soil ingestion rate of 0.4 kg/day for dairy cattle, based on Fries (1994). U.S. EPA (1994r) and NC DEHNR (1997) noted that Fries (1994) reported soil ingestion rates as 2 percent of the total dry matter intake. NAS (1987) was also referenced, which reported an average dairy cattle weight of 630 kilograms and a daily dry matter intake rate (nonlactating cows) of 3.2 percent of body weight. This resulted in a daily dry matter intake rate of 20 kg/day DW, and a daily soil ingestion rate of approximately 0.4 kg/day. NC DEHNR (1997) recommended a soil ingestion rate of 0.2 kg/day for the cattle of typical dairy farmers, citing Rice (1994). U.S. EPA (1990e) reported soil ingestion rates as 3 percent of the forage intake rate. U.S. EPA (1990e) assumed that the more protective forage intake rate of 13.2 kg/day DW results in a daily soil ingestion rate of about 0.4 kg/day. Simmonds and Linsley (1981) and Thornton and Abrams (1983) were cited as the references for this assumption.

Uncertainties associated with *Qs* include the lack of current empirical data to support soil ingestion rates for dairy cattle. Assuming uniform contamination of soil ingested by cattle also adds uncertainty.

5.4.5.5 Average Soil Concentration Over Exposure Duration (*Cs*)

The calculation of *Cs* for dairy cattle is the same as for beef cattle (Section 5.4.4.5).

> ***Please Note:*** You might need to generate soil concentration estimates for grain separate from those for forage and silage. Currently, the HHRAP assumes that forage and silage are grown on untilled land, and grain is grown on tilled land. We highly recommend making sure that your C_s calculations include the appropriate Z_s (2 for untilled land, 20 for tilled land).

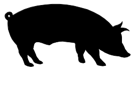
5.4.5.6 Soil Bioavailability Factor (*Bs*)

The calculation of *Bs* for dairy cattle is the same as for beef cattle (Section 5.4.4.6).

5.4.5.7 Metabolism Factor (*MF*)

The recommended values for *MF* are identical to those we recommend for beef cattle (Section 5.4.4.7).

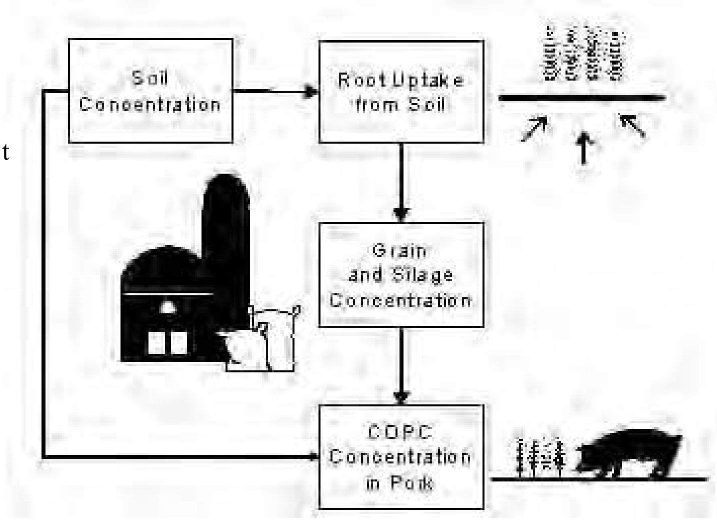
5.5 CALCULATING COPC CONCENTRATIONS IN PORK



Under the approach recommended in this guidance, COPC concentrations in pork tissue are estimated on the basis of the amount of COPCs that swine consume through a diet consisting of silage and grain. Additional COPC contamination of pork tissue may occur through their ingestion of soil.

FIGURE 5-5

COPC CONCENTRATION IN PORK



Equation 5-22 (Section 5.4.4) describes how we recommend calculating COPC concentration in beef cattle (A_{beef}). We suggest modifying Equation 5-22 to calculate COPC concentrations in swine (A_{pork}), as follows:

Recommended Equation for Calculating:

Concentration of COPC in Pork (A_{pork})

Equation 5-25

where

A_{pork} = Concentration of COPC in pork (mg COPC/kg FW tissue)
F_i = Fraction of plant type *i* grown on contaminated soil and ingested by the animal (swine)(unitless)
Qp_i = Quantity of plant type *i* eaten by the animal (swine) each day (kg DW plant/day)
P_i = Concentration of COPC in plant type *i* eaten by the animal (swine) (mg/kg DW)
Qs = Quantity of soil eaten by the animal (swine) (kg/day)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Bs = Soil bioavailability factor (unitless)
Ba_{pork} = COPC biotransfer factor for pork (day/kg FW tissue)
MF = Metabolism factor (unitless)

Appendix A-2 explains how we recommend calculating the COPC-specific parameter Ba_{pork}. The discussions in Section 5.4.5 of the variables F_i, Qp_i, P_i, Qs, Cs and MF for beef cattle generally apply to the corresponding variables for pork. However, some different assumptions are made for pork. These differences are summarized in the following subsections.

We generally recommend using Equation 5-25 to calculate COPC pork concentrations (A_{pork}). This equation is further described in Appendix B, Table B-3-12.

5.5.1 Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Swine) (F_i)

The calculation of F_i for pork is identical to that for beef cattle (Section 5.4.4.1).

5.5.2 Quantity of Plant Type *i* Eaten by the Animal (Swine) Each Day (Qp_i)

Section 5.4.4.2 discusses estimating the daily quantity of forage, silage, and grain feed consumed by beef cattle for each feed category. However, daily ingestion rates for pork are estimated differently than for beef cattle. U.S. EPA (1994r and 1998c), and NC DEHNR (1997) recommended only including silage and grain feeds to estimate daily plant quantity eaten by swine. Because swine are not grazing animals, they are assumed not to eat forage (U.S. EPA 1998c). We therefore generally recommend estimating the daily quantity of plant feeds (kilograms of DW) consumed by swine for each category of plant feed.

U.S. EPA (1990e) and NC DEHNR (1997) recommended grain and silage ingestion rates for swine of 3.0 and 1.3 kg DW/day, respectively. NC DEHNR (1997) references U.S. EPA (1990e) as the source of these ingestion rates. U.S. EPA (1990e) reported total dry matter ingestion rates for hogs and lactating sows as 3.4 and 5.2 kg DW/day, respectively. U.S. EPA (1990e) cites Boone et al. (1981) as the source of the ingestion rate for hogs, and NAS (1987) as the source of the ingestion rate for a lactating sow. Boone et al. (1981) reported a grain ingestion rate of 3.4 kg DW/day for a hog. NAS (1987) reported an average ingestion rate of 5.2 kg DW/day for a lactating sow. U.S. EPA (1990e) recommended using the average of these two rates (4.3 kg DW/day).

U.S. EPA (1990e) assumed that 70 percent of the swine diet is grain and 30 percent silage to obtain the grain ingestion rate of 3.0 kg DW/day and the silage ingestion rate of 1.3 kg DW/day. U.S. EPA (1990e) cited U.S. EPA (1982b) as the source of the grain and silage dietary fractions. U.S. EPA (1995b) recommended an ingestion rate of 4.7 kg DW/day for a swine, referencing NAS (1987). NAS (1987) reported an average daily intake of 4.36 kg DW/day for a gilt (young sow) and a average daily intake of 5.17 kg DW/day for a sow, which averages out to 4.7 kg/DW/day. Assuming the 70 percent grain to 30 percent silage diet noted above, estimated ingestion rates of 3.3 kg DW/day (grain) and 1.4 kg DW/day (silage) are derived.

Recommended Values for: Quantity of Plant Type *i* Eaten by the Animal (Swine) Each Day (Qp_i)
Grain = 3.3 kg DW/day Silage = 1.4 kg DW/day

Uncertainties associated with this variable include the variability of actual grain and silage ingestion rates from site to site. You could use site-specific data to mitigate this uncertainty. In addition, assuming uniform contamination of the plant materials consumed by swine produces some uncertainty.

5.5.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Swine) (P_i)

The suggested calculation of P_i for pork is identical to that for beef cattle (Section 5.4.4.3).

5.5.4 Quantity of Soil Eaten by the Animal (Swine) Each Day (Q_s)

As discussed in Section 5.4.4.4, additional contamination of swine results from ingestion of soil. The following Q_s values were recommended by earlier guidance:

Guidance	Quantity of Soil Eaten by Swine Each Day (Q_s)
U.S. EPA (1990e)	Stated that sufficient data are not available to estimate swine soil ingestion rates.
NC DEHNR (1997)	0.37 kg/day Estimated by assuming a soil intake that is 8% of the plant ingestion rate of 4.3 kg DW/day). U.S. EPA (1993f) was cited as the reference for the soil ingestion rate of 8 percent of dry matter intake.
U.S. EPA (1998c)	Cites a companion “Parameters Guidance Document” for detailed recommendations on Q. The “Parameters” document has not been published.

As in NC DEHNR (1997), we recommend the following soil ingestion rate for swine:

Recommended Value for:
Quantity of Soil Eaten by the Animal (Swine) Each Day (Q_s)

0.37 kg DW/day

Uncertainties associated with this variable include the lack of current empirical data to support soil ingestion rates for swine, and assuming uniform contamination of the soil ingested by swine.

5.5.5 Average Soil Concentration Over Exposure Duration (*Cs*)

Our suggested calculation of *Cs* for pork is the same as for beef cattle (Section 5.4.4.5).

> ***Please Note:*** You might need to generate soil concentration estimates for grain separate from those for silage. We recommend assuming that silage is grown on untilled land, and grain is grown on tilled land. We highly recommend that you make sure that your C_s calculations include the appropriate Z_s (2 for untilled land, 20 for tilled land).

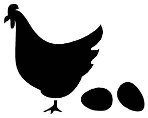
5.5.6 Soil Bioavailability Factor (*Bs*)

Our suggested calculation of *Bs* for pork is the same as for beef cattle (Section 5.4.4.6)

5.5.7 Metabolism Factor (*MF*)

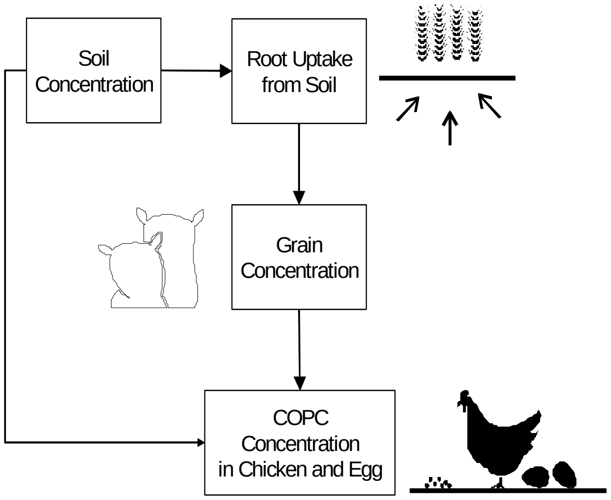
Our recommended values for MF are identical to those we recommended for beef cattle (Section 5.4.4.7).

5.6 CALCULATING COPC CONCENTRATIONS IN CHICKEN AND EGGS



Under the approach outlined in this guidance document, estimates of the COPC concentrations in chicken and eggs are based on the amount of COPCs that chickens consume through ingestion of grain and soil. We recommend assuming that the uptake of COPCs via inhalation and via ingestion of water are insignificant relative to other pathways. The HHRAP assumes that chickens are housed in a typical manner that allows contact with soil. Because of this, chickens are assumed to consume 10 percent of their diet as soil. Assuming 10 percent is consistent with the study from which the biotransfer factors were obtained (Stephens et al. 1995). We recommend assuming that the remainder of the diet (90 percent) consists of grain grown at the exposure scenario location. Therefore, it’s appropriate to assume 100 percent of the grain consumed is contaminated. The equations don’t account for the uptake of COPCs via ingestion of contaminated insects and other organisms (e.g., worms, etc.), which may also contribute to the ingestion of COPCs. This may be a limitation, depending on the site-specific conditions under which the chickens are raised.

FIGURE 5-6
COPC CONCENTRATION IN CHICKEN & EGGS



We generally recommend using the algorithm for aboveground produce described in Section 5.3 to estimate the COPC concentration in grain. Grain is considered to be protected from direct deposition of particles, and vapor transfer. This approach considers only contamination due to root uptake of COPCs in calculating COPC concentrations in grain. Our recommended equations for calculating concentrations in chicken and eggs are presented in Appendix B. The method we used to derive biotransfer factors, and the COPC-specific values for chicken and eggs are presented in Appendix A-2.

As in NC DEHNR (1997), we recommend using Equation 5-26 to calculate COPC concentrations in chicken and eggs (Stephens et al. 1995). We generally recommend calculating COPC concentrations in chicken and eggs separately. Parameters and variables in Equation 5-26 are further described in Appendix B, Tables B-3-13 and B-3-14.

Recommended Equation for Calculating:
Concentration of COPC in Chicken and Eggs ($A_{chicken}$ or A_{egg})

$$A_{chicken} \text{ or } A_{egg} = (\Sigma[F_i \cdot Qp_i \cdot Pi] + Qs \cdot Cs \cdot Bs) \cdot (Ba_{egg} \text{ or } Ba_{chicken}) \qquad \text{Equation 5-26}$$

where

$A_{chicken}$ = Concentration of COPC in chicken (mg COPC/kg FW tissue)
A_{egg} = Concentration of COPC in eggs (mg COPC/kg FW tissue)
F_i = Fraction of plant type *i* (grain) grown on contaminated soil and ingested by the animal (chicken)(unitless)
Qp_i = Quantity of plant type *i* (grain) eaten by the animal (chicken) each day (kg DW plant/day)
P_i = Concentration of COPC in plant type *i* (grain) eaten by the animal (chicken) (mg/kg DW)
Qs = Quantity of soil eaten by the animal (chicken) (kg/day)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
Bs = Soil bioavailability factor (unitless)
$Ba_{chicken}$ = COPC biotransfer factor for chicken (day/kg FW tissue)
Ba_{egg} = COPC biotransfer factor for eggs (day/kg FW tissue)

Appendix A-2 explains how we recommend determining the COPC-specific parameters $Ba_{chicken}$ and Ba_{egg}. The remaining parameters are discussed in Appendix B and in the following subsections.

5.6.1 Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Chicken)(F_i)

The calculation of F_i for chicken is identical to that for beef cattle (Section 5.4.4.1).

5.6.2 Quantity of Plant Type *i* Eaten by the Animal (Chicken) Each Day (Qp_i)

Section 5.4.4.2 discusses estimating the daily quantity of forage, silage, and grain feed consumed by beef cattle for each feed category. However, daily ingestion rates for chicken are estimated differently than for beef cattle. NC DEHNR (1997) recommended that only grain feeds be included in this estimate. Because chickens are not grazing animals, they are assumed not to eat forage (U.S. EPA 1998c). Chickens are similarly assumed not to consume any silage. We recommend only estimating the daily quantity of plant feeds (kilograms of DW) consumed by chicken (Qp) for grain feed.

As in Ensminger (1980), Fries (1982), and NAS (1987), we recommend using the following ingestion rate:

Recommended Value for:
Quantity of Plant Type *i* Eaten by the Animal (Chicken) Each Day (Qp_i)

Grain = 0.2 kg DW/day

Uncertainties associated with this variable include the variability of actual grain ingestion rates from site to site. In addition, assuming uniform contamination of plant materials consumed by chicken produces some uncertainty.

5.6.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Chicken) (P_i)

The total COPC concentration is the COPC concentration in grain. We recommend using Equation 5-27 to calculate P_i. This equation is further described in Appendix B.

Recommended Equation for Calculating:
Concentration of COPC in Plant Type *i* Eaten by the Animal (Chicken) (P_i)

$$P_i = \sum_i (Pr) \qquad \text{Equation 5-27}$$

where

P_i = Concentration of COPC in each plant type *i* eaten by the animal (mg COPC/kg DW)
Pr = Plant concentration due to root uptake (mg COPC/kg DW)

We generally recommend calculating plant concentration due to root uptake (*Pr*) using Equation 5-20, as discussed in Section 5.3.3.

5.6.4 Quantity of Soil Eaten by the Animal (Chicken) Each Day (Qs)

COPC concentration in chickens also results from intake of soil. As discussed earlier, The HHRAP assumes that chickens consume 10 percent of their total diet as soil, a percentage that is consistent with the study from Stephens et al. (1995). We recommend the following soil ingestion rate for chicken:

Recommended Value for:
Quantity of Soil Eaten by the Animal (Chicken) Each Day (*Qs*)

0.022 kg DW/day

Uncertainties associated with this variable include the lack of current empirical data to support soil ingestion rates for chicken, and assuming uniform contamination of soil ingested by chicken.

5.6.5 Average Soil Concentration Over Exposure Duration (*Cs*)

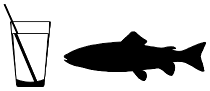
The calculation of *Cs* for chicken is the same as for beef cattle (Section 5.4.4.5).

> ***Please Note:*** We recommend assuming that forage and silage are grown on untilled land, and grain is grown on tilled land. We highly recommend making sure that your C_s calculations include the appropriate Z_s (20 for tilled land).

5.6.6 Soil Bioavailability Factor (*Bs*)

The calculation of *Bs* for chicken is the same as for beef cattle (Section 5.4.4.6)

5.7 CALCULATING COPC CONCENTRATIONS IN DRINKING WATER AND FISH

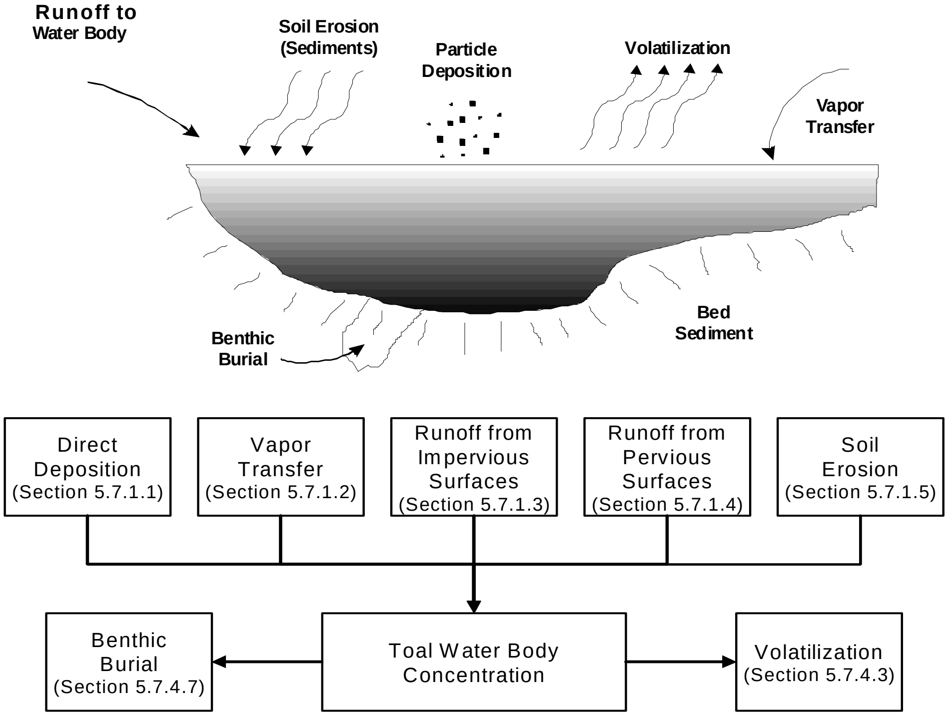


We generally recommend calculating COPC concentrations in surface water for all water bodies you selected to evaluate in the risk assessment. Specifically, those waterbodies selected as potential sources for the drinking water and/or fish ingestion exposure pathways. Mechanisms we suggest considering in determining COPC loading of the water column include:

- Direct deposition,
- Runoff from impervious surfaces within the watershed,
- Runoff from pervious surfaces within the watershed,
- Soil erosion over the total watershed,
- Direct diffusion of vapor phase COPCs into the surface water, and
- Internal transformation of compounds chemically or biologically.

Considering other potential mechanisms may be appropriate, due to site-specific conditions (e.g., tidal influences). Typically, though, we assume that contributions from other potential mechanisms are negligible compared to those evaluated in the HHRAP.

FIGURE 5-7
COPC LOADING TO THE WATER BODY



The total concentration of each COPC partitions between the sediment and the water column. Partitioning between water and sediment varies with the COPC. The HHRAP uses the Universal Soil Loss Equation (USLE) and a sediment delivery ratio to estimate the rate of soil erosion from the watershed. The equations we recommend for estimating surface water concentrations include a sediment mass balance, in which the amount of sediment assumed to be buried and lost from the water body is equal to the difference between the amount of soil introduced to the water body by erosion and the amount of suspended solids lost in downstream flow. As a result, we typically assume that sediments do not accumulate in the water body over time, and an equilibrium is maintained between the surficial layer of sediments and the water column. The total water column COPC concentration is the sum of the COPC concentration dissolved in water and the COPC concentration associated with suspended solids. Appendix B-4 presents the equations we recommend using to estimate surface water concentrations.

To evaluate the COPC loading to a water body from its associated watershed, we generally recommend calculating watershed soil-specific COPC concentrations. The equation in Section 5.2 for estimating COPC concentration in soil includes a loss term that considers the loss of contaminants from the soil after deposition. These loss mechanisms all lower the soil concentration associated with a specific deposition rate. Appendix B (Tables B-4-1 through B-4-28) provides the equations we recommend for calculating COPC concentrations in watershed soils and in the water body.

The equations presented in Appendix B for modeling COPC loading to a water body represent a simple steady-state model to solve for a water column in equilibrium with the upper sediment layer. These equations (Appendix B) predict the steady-state mass of contaminants in the water column and underlying sediments, and don’t address the dynamic exchange of contaminants between the water body and the sediments following changes in external loadings. While appropriate for calculating risk under long-term average conditions, evaluating complex water bodies or shorter term loading scenarios might be improved by using a dynamic modeling framework [e.g., Exposure Analysis Modeling System (EXAMS), or Water Quality Analysis Simulation Program (WASP), both of which can be downloaded from the EPA Center for Exposure Assessment Modeling]. Although typically more resource intensive, such analysis may be able to refine modeling of contaminant loading to a water body. Also, the computations may better represent the exposure scenario you are evaluating.

For example, EXAMS allows performing computations for each defined segment or compartment of a water body or stream. These compartments are considered physically homogeneous and are connected via advective and dispersive fluxes. Compartments can be defined as littoral, epilimnion, hypolimnion, or benthic. Such resolution also makes it possible to assign receptor locations specific to certain portions of a water body where evaluating exposure is of greatest interest.

The following are some considerations regarding the selection and use of a dynamic modeling framework or simulation model to evaluate water bodies:

- Will a complex surface water modeling effort provide enhanced results over the use of the more simplistic steady-state equations presented in Appendix B?
- Are the resources needed to conduct, as well as review, a more complex modeling effort justified compared to the more refined results?
- Has the model been used previously for regulatory purposes, and therefore, already has available documentation to support such uses?

- Can the model conduct steady-state and dynamic analysis? and
- Does the model require calibration with field data, and if so, are there sufficient quantity and quality of site-specific data available to support calibration?

As mentioned previously in Chapter 2 (Section 2.3.5.3 - "Mercury"), the SERAFM model offers a dynamic modeling framework for mercury that enables the user to model specific water body mercury transformation processes in lieu of applying default speciation assumptions.

5.7.1 Total COPC Load to the Water Body (L_T)

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-28 to calculate the total COPC load to a water body (L_T). This equation is described in detail in Appendix B, Table B-4-7.

Recommended Equation for Calculating:
Total COPC Load to the Water Body (L_T)

$$L_T = L_{DEP} + L_{dif} + L_{RI} + L_R + L_E + L_I \qquad \text{Equation 5-28}$$

where

L_T = Total COPC load to the water body (including deposition, runoff, and erosion) (g/yr)
L_{DEP} = Total (wet and dry) particle phase and vapor phase COPC direct deposition load to water body (g/yr)
L_{dif} = Vapor phase COPC diffusion load to water body (g/yr)
L_{RI} = Runoff load from impervious surfaces (g/yr)
L_R = Runoff load from pervious surfaces (g/yr)
L_E = Soil erosion load (g/yr)
L_I = Internal transfer (g/yr)

Due to the limited data and uncertainty associated with the chemical or biological internal transfer, L_I, of compounds into degradation products, we generally recommend a default value for this variable of zero. However, if a permitting authority determines that site-specific conditions indicate calculating internal transfer may need to be considered, we recommend following the methods described in U.S. EPA (1998c). The remaining variables (L_{DEP}, L_{dif}, L_{RI}, L_R, and L_E) are discussed in the following subsections.

5.7.1.1 Total (Wet and Dry) Particle Phase and Vapor Phase COPC Direct Deposition Load to Water Body (L_{DEP})

As in U.S. EPA (1994r) and NC DEHNR (1997), with the inclusion of the direct deposition of total vapor, we recommend using Equation 5-29 to calculate the load to the water body from the direct deposition of wet and dry particles and vapors onto the surface of the water body (L_{DEP}). The equation is described in detail in Appendix B, Table B-4-8.

Recommended Equation for Calculating:
Total Particle Phase and Vapor Phase Direct Deposition Load to Water Body (L_{DEP})

$$L_{DEP} = Q \cdot [\, F_v \cdot Dytwv + (1 - F_v) \cdot Dytwp \,] \cdot A_W \qquad \text{Equation 5-29}$$

where

L_{DEP} = Total (wet and dry) particle phase and vapor phase COPC direct deposition load to water body (g/yr)
Q = COPC emission rate (g/s)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
$Dytwv$ = Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase (s/m^2-yr)
$Dytwp$ = Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase (s/m^2-yr)
A_W = Water body surface area (m^2)

Appendix A-2 describes how we recommend determining the COPC-specific parameter F_v. Chapter 3 describes generating the modeled air parameters, *Dytwv* and *Dytwp*. Methods for determining the water body surface area, A_w, are described in Chapter 4 and Appendix B.

5.7.1.2 Vapor Phase COPC Diffusion Load to Water Body (L_{dif})

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-30 to calculate L_{dif}. The equation is described in detail in Appendix B, Table B-4-12.

Recommended Equation for Calculating:
Vapor Phase COPC Diffusion Load to Water Body (L_{Dif})

$$L_{dif} = \frac{K_v \cdot Q \cdot F_v \cdot Cywv \cdot A_W \cdot 1 \times 10^{-6}}{\frac{H}{R \cdot T_{wk}}} \qquad \text{Equation 5-30}$$

where

L_{dif} = Vapor phase COPC diffusion load to water body (g/yr)
K_v = Overall COPC transfer rate coefficient (m/yr)
Q = COPC emission rate (g/s)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
$Cywv$ = Unitized yearly (water body or watershed) average air concentration from vapor phase (µg-s/g-m^3)
A_W = Water body surface area (m^2)
10^{-6} = Units conversion factor (g/µg)
H = Henry's Law constant (atm-m^3/mol)
R = Universal gas constant (atm-m^3/mol-K)
T_{wk} = Water body temperature (K)

Calculating the overall COPC transfer rate coefficient (K_v) is described in Section 5.7.4.4, as well as in Appendix B, Table B-4-19. Chapters 2 and 3 explain how we recommend quantifying the COPC emission rate (Q). Appendix A-2 describes how we recommend determining the COPC-specific parameters F_v, H, and R. Chapter 3 describes generating the modeled air parameter, *Cywv*. Methods for determining the water body surface area, A_w, are described in Chapter 4 and Appendix B. Consistent with U.S. EPA (1994r) and U.S. EPA (1998c), we recommend a default water body temperature (T_{wk}) of 298 K (or 25°C).

5.7.1.3 Runoff Load from Impervious Surfaces (L_{RI})

In some watershed soils, a portion of the total (wet and dry) deposition in the watershed will be to impervious surfaces. This deposition may accumulate and be washed off during rain events. As in U.S. EPA (1994r) and NC DEHNR (1997), with the inclusion of total (wet and dry) vapor phase deposition, we recommend using Equation 5-31 to calculate impervious runoff load to a water body (L_{RI}). The equation is also presented in Appendix B, Table B-4-9.

Recommended Equation for Calculating:
Runoff Load from Impervious Surfaces (L_{RI})

$$L_{RI} = Q \cdot [\, F_v \cdot Dytwv + (1.0 - F_v) \cdot Dytwp \,] \cdot A_I \qquad \text{Equation 5-31}$$

where

L_{RI} = Runoff load from impervious surfaces (g/yr)
Q = COPC emission rate (g/s)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
$Dytwv$ = Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase (s/m^2-yr)
$Dytwp$ = Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase (s/m^2-yr)
A_I = Impervious watershed area receiving COPC deposition (m^2)

Chapters 2 and 3 explain how we recommend quantifying the COPC emission rate (Q). Appendix A-2 describes how we recommend determining the COPC-specific parameter F_v. Chapter 3 describes a method for generating the modeled air parameters, *Dytwv* and *Dytwp*. Impervious watershed area receiving COPC deposition (A_I) is the portion of the total effective watershed area that is impervious to rainfall (such as roofs, driveways, streets, and parking lots) and drains to the water body. Our recommended method for determining A_I is described in Chapter 4 and Appendix B.

5.7.1.4 Runoff Load from Pervious Surfaces (L_R)

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-32 to calculate the runoff dissolved COPC load to the water body from pervious soil surfaces in the watershed (L_R). The equation is also presented in Appendix B, Table B-4-10.

Recommended Equation for Calculating:
Runoff Load from Pervious Surfaces (L_R)

$$L_R = RO \cdot (A_L - A_I) \cdot \frac{Cs \cdot BD}{\theta_{sw} + Kd_s \cdot BD} \cdot 0.01 \qquad \text{Equation 5-32}$$

where

L_R	=	Runoff load from pervious surfaces (g/yr)
RO	=	Average annual surface runoff from pervious areas (cm/yr)
A_L	=	Total watershed area receiving COPC deposition (m^2)
A_I	=	Impervious watershed area receiving COPC deposition (m^2)
Cs	=	Average soil concentration over exposure duration (in watershed soils) (mg COPC/kg soil)
BD	=	Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3
θ_{sw}	=	Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3
Kd_s	=	Soil-water partition coefficient (cm^3 water/g soil)
0.01	=	Units conversion factor (kg-cm^2/mg-m^2)

Appendix B describes how we recommend determining the site-specific parameters RO, A_L, A_I, BD, and θ_{sw}. We also address soil bulk density (BD) in Section 5.2.4.2. We also address soil water content (θ_{sw}) in Section 5.2.4.4. Our recommended method for calculating the COPC concentration in watershed soils (Cs) is discussed in Section 5.2.1 and Appendix B, Table B-4-1. Appendix A-2 describes how we recommend calculating the COPC-specific soil/water partition coefficient (Kd_s).

5.7.1.5 Soil Erosion Load (L_E)

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-33 to calculate soil erosion load (L_E). The equation is also presented in Appendix B, Table B-4-11.

Recommended Equation for Calculating:
Soil Erosion Load (L_E)

$$L_E = X_e \cdot (A_L - A_I) \cdot SD \cdot ER \cdot \frac{Cs \cdot Kd_s \cdot BD}{\theta_{sw} + Kd_s \cdot BD} \cdot 0.001 \qquad \text{Equation 5-33}$$

where

L_E = Soil erosion load (g/yr)
X_e = Unit soil loss (kg/m^2-yr)
A_L = Total watershed area (evaluated) receiving COPC deposition (m^2)
A_I = Impervious watershed area receiving COPC deposition (m^2)
SD = Sediment delivery ratio (watershed) (unitless)
ER = Soil enrichment ratio (unitless)
Cs = Average soil concentration over exposure duration (in watershed soils) (mg COPC/kg soil)
BD = Soil bulk density (g soil/cm^3 soil) = 1.5 g/cm^3
θ_{sw} = Soil volumetric water content (ml water/cm^3 soil) = 0.2 ml/cm^3
Kd_s = Soil-water partition coefficient (ml water/g soil)
0.001 = Units conversion factor (k-cm^2/mg-m^2)

Section 5.7.2 describes unit soil loss (X_e). Chapter 4 and Appendix B describe how we recommend determining the site-specific parameters A_L and A_I. We generally recommend calculating the watershed sediment delivery ratio (SD) as described in Section 5.7.3 and in Appendix B, Table B-4-14. COPC concentration in soils (Cs) is described in Section 5.2.1, and Appendix B, Table B-4-1. Soil bulk density (BD) is described in Section 5.2.4.2. Soil water content (θ_{sw}) is described in Section 5.2.4.4. Appendix B, Table B-4-11 describes how we recommend determining the COPC-specific soil enrichment ration (ER).

5.7.2 Universal Soil Loss Equation - USLE

As in U.S. EPA (1994g and 1994r), we generally recommend using the universal soil loss equation (USLE), Equation 5-33A, to calculate the unit soil loss (X_e) specific to each watershed. This equation is further described in Appendix B, Table B-4-13. Appendix B also describes how we suggest determining the site- and watershed-specific values for each of the variables associated with Equation 5-33A.

Recommended Equation for Calculating:
Unit Soil Loss (X_e)

$$X_e = RF \cdot K \cdot LS \cdot C \cdot PF \cdot \frac{907.18}{4047} \qquad \text{Equation 5-33A}$$

where

X_e = Unit soil loss (kg/m^2-yr)
RF = USLE rainfall (or erosivity) factor (yr^{-1})
K = USLE erodibility factor (ton/acre)
LS = USLE length-slope factor (unitless)
C = USLE cover management factor (unitless)
PF = USLE supporting practice factor (unitless)
907.18 = Units conversion factor (kg/ton)
4047 = Units conversion factor (m^2/acre)

The USLE *RF* variable, which represents the influence of precipitation on erosion, is derived from data on the frequency and intensity of storms. This value is typically derived on a storm-by-storm basis, but average annual values have been compiled (U.S. Department of Agriculture 1982). Information on determining site-specific values for variables used in calculating X_e is provided in U.S. Department of Agriculture (U.S. Department of Agriculture 1997) and U.S. EPA guidance (U.S. EPA 1985b). Refer to Appendix B, Table B-4-13 for additional discussion of the USLE.

5.7.3 Sediment Delivery Ratio (*SD*)

We recommend using Equation 5-34 to calculate the sediment delivery ratio (*SD*). The use of this equation is further described in Appendix B, Table B-4-14.

Recommended Equation for Calculating:
Sediment Delivery Ratio *(SD)*

$$SD = a \cdot (A_L)^{-b} \qquad \text{Equation 5-34}$$

where

SD = Sediment delivery ratio (watershed) (unitless)
a = Empirical intercept coefficient (unitless)
b = Empirical slope coefficient (unitless)
A_L = Total watershed area (evaluated) receiving COPC deposition (m^2)

The sediment delivery ratio (*SD*) for a large land area (i.e. a watershed or part of a watershed) can be calculated, on the basis of the area of the watershed, by using an approach proposed by Vanoni (1975). Accordingly, U.S. EPA (1998c) recommended using Equation 5-34 to calculate the *SD*.

According to Vanoni (1975), sediment delivery ratios vary approximately with the -0.125 power of the drainage area. Therefore, the empirical slope coefficient is assumed to be equal to 0.125. An inspection of the data presented by Vanoni (1975) indicates that the empirical intercept coefficient varies with the size of the watershed, as illustrated in Appendix B, Table B-4-14.

A_L is the total watershed surface area evaluated that is affected by deposition and drains to the body of water (see Chapter 4). In assigning values to the watershed surface area affected by deposition, we generally consider the following relevant:

- the distance from the emission source,
- the location of the area affected by deposition fallout with respect to the point at which drinking water is extracted or fishing occurs
- the watershed hydrology.

5.7.4 Total Water Body COPC Concentration (C_{wtot})

We recommend using Equation 5-35 to calculate the total water body COPC concentration (C_{wtot}). C_{wtot} includes both the water column and the bed sediment. The equation is also presented in Appendix B, Table B-4-15.

Recommended Equation for Calculating:
Total Water Body COPC Concentration (C_{wtot})

$$C_{wtot} = \frac{L_T}{Vf_x \cdot f_{wc} + k_{wt} \cdot A_W \cdot (d_{wc} + d_{bs})} \qquad \text{Equation 5-35}$$

where

C_{wtot} = Total water body COPC concentration (including water column and bed sediment) (g COPC/m^3 water body)
L_T = Total COPC load to the water body (including deposition, runoff, and erosion) (g/yr)
Vf_x = Average volumetric flow rate through water body (m^3/yr)
f_{wc} = Fraction of total water body COPC concentration in the water column (unitless)
k_{wt} = Overall total water body COPC dissipation rate constant (yr^{-1})
A_W = Water body surface area (m^2)
d_{wc} = Depth of water column (m)
d_{bs} = Depth of upper benthic sediment layer (m)

The total COPC load to the water body (L_T)—including deposition, runoff, and erosion—is described in Section 5.7.1 and Appendix B, Table B-4-7. Average volumetric flow rate through the water body (Vf_x) and water body surface area (A_w) are discussed in Appendix B. Section 5.7.4.1 discusses the fraction of total COPC concentration in the water column (f_{wc}). Section 5.7.4.2 discusses the COPC dissipation rate constant (k_{wt}). Chapter 4 discusses the water body-specific d_{wc}. We discuss the depth of the upper benthic sediment layer (d_{bs}) below.

The depth of the upper benthic layer (d_{bs}), which represents the portion of the bed that is in equilibrium with the water column, cannot be precisely specified; however, U.S. EPA (1998c) recommended values ranging from 0.01 to 0.05. As in U.S. EPA (1994r), we recommend a default value of 0.03, which represents the midpoint of the specified range. Issues related to the remaining parameters are summarized in the following subsections.

Recommended Default Value for:
Depth of Upper Benthic Sediment Layer (d_{bs})

0.03 m

5.7.4.1 Fraction of Total Water Body COPC Concentration in the Water Column (f_{wc}) and Benthic Sediment (f_{bs})

We generally recommend using Equation 5-36A to calculate the fraction of total water body COPC concentration in the water column (f_{wc}), and Equation 5-36B to calculate the total water body contaminant concentration in benthic sediment (f_{bs}). The equations are also presented in Appendix B, Table B-4-16.

Recommended Equation for Calculating:
Fraction of Total Water Body COPC Concentration in
the Water Column (f_{wc}) and Benthic Sediment (f_{bs})

$$f_{wc} = \frac{(1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}) \cdot d_{wc} / d_z}{(1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}) \cdot d_{wc} / d_z + (\theta_{bs} + Kd_{bs} \cdot C_{BS}) \cdot d_{bs} / d_z} \qquad \text{Equation 5-36A}$$

$$f_{bs} = 1 - f_{wc} \qquad \text{Equation 5-36B}$$

where

f_{wc} = Fraction of total water body COPC concentration in the water column (unitless)
f_{bs} = Fraction of total water body COPC concentration in benthic sediment (unitless)
Kd_{sw} = Suspended sediments/surface water partition coefficient (L water/kg suspended sediment)
TSS = Total suspended solids concentration (mg/L)
1×10^{-6} = Units conversion factor (kg/mg)
d_z = Total water body depth (m)
θ_{bs} = Bed sediment porosity ($L_{water}/L_{sediment}$)
Kd_{bs} = Bed sediment/sediment pore water partition coefficient (L water/kg bottom sediment)
C_{BS} = Bed sediment concentration (g/cm^3 [equivalent to kg/L])
d_{wc} = Depth of water column (m)
d_{bs} = Depth of upper benthic sediment layer (m)

The COPC-specific partition coefficient (Kd_{sw}) describes the partitioning of a contaminant between sorbing material, such as soil, surface water, suspended solids, and bed sediments (see Appendix A-2). Total suspended solids (TSS), total water body depth (d_z), bed sediment porosity (θ_{bs}) and bed sediment concentration (C_{BS}) are addressed below. Bed sediment and sediment pore water partition coefficient

(Kd_{bs}) is discussed in Appendix A-2. Depth of water column (d_{wc}) and depth of upper benthic layer (d_{bs}) are discussed in Section 5.7.4.

U.S. EPA (1998c) and NC DEHNR (1997) recommended using Equations 5-36A and 5-36B to calculate f_{wc} and f_{bs}. NC DEHNR (1997) also recommended adding the depth of the water column to the depth of the upper benthic layer ($d_{wc} + d_{bs}$) to calculate the total water body depth (d_z).

NC DEHNR (1997) recommended a default total suspended solids (*TSS*) concentration of 10 mg/L, which was adapted from U.S. EPA (1993e). However, due to variability in water body specific values for this variable, we recommend using water body-specific measured *TSS* values representative of long-term average annual values. Average annual values for *TSS* are generally expected to be in the range of 2 to 300 mg/L. Additional information on anticipated *TSS* values is available in U.S. EPA (1998c).

If measured data are not available, or of unacceptable quality, it's possible to calculate a *TSS* value for non-flowing water bodies using Equation 5-36C.

$$TSS = \frac{X_e \cdot (A_L - A_I) \cdot SD \cdot 1x10^3}{Vf_x + D_{ss} \cdot A_W} \qquad \text{Equation 5-36C}$$

where

TSS = Total suspended solids concentration (mg/L)
X_e = Unit soil loss (kg/m^2-yr)
A_L = Total watershed area (evaluated) receiving COPC deposition (m^2)
A_I = Impervious watershed area receiving COPC deposition (m^2)
SD = Sediment delivery ratio (watershed) (unitless)
Vf_x = Average volumetric flow rate through water body (value should be 0 for quiescent lakes or ponds) (m^3/yr)
D_{ss} = Suspended solids deposition rate (a default value of 1,825 for quiescent lakes or ponds) (m/yr)
A_W = Water body surface area (m^2)

The default value of 1,825 m/yr provided for D_{ss} is characteristic of Stoke's settling velocity for an intermediate (fine to medium) silt.

Also, it's possible to evaluate the appropriateness of watershed-specific values used in calculating the unit soil loss (X_e), as described in Section 5.7.2 and Appendix B, by comparing the water-body specific measured *TSS* value to the estimated *TSS* value obtained using Equation 5-36C. If the measured and calculated *TSS* values differ significantly, we recommend re-evaluating the parameter values used to calculate X_e. You might also re-evaluate *TSS* and X_e if the calculated *TSS* value is outside of the normal range expected for average annual measured values, as discussed above.

One approach to calculating bed sediment porosity (θ_{bs}) from the bed sediment concentration is by using the following equation (U.S. EPA 1998c):

$$\theta_{bs} = 1 - \frac{C_{BS}}{\rho_s} \qquad \text{Equation 5-37}$$

where

θ_{bs} = Bed sediment porosity ($L_{water}/L_{sediment}$)
ρ_s = Bed sediment density (kg/L)
C_{BS} = Bed sediment concentration (kg/L)

We recommend the following default value for bed sediment porosity (θ_{bs}), adapted from NC DEHNR (1997):

Recommended Value for: **Bed Sediment Porosity (θ_{bs})**
$\theta_{bs} = 0.6$ $L_{water}/L_{sediment}$ assuming ρ_s = 2.65 kg/L [bed sediment density] and C_{BS}= 1.0 kg/L [bed sediment concentration])

U.S. EPA (1994r) and NC DEHNR (1997) recommended a benthic solids concentration (C_{BS}) ranging from 0.5 to 1.5 kg/L, which was adapted from U.S. EPA (1993e). We recommend the following default value for bed sediment concentration (C_{BS}):

Recommended Default Value for: **Bed Sediment Concentration (C_{BS})**
1.0 kg/L

5.7.4.2 Overall Total Water Body COPC Dissipation Rate Constant (k_{wt})

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-38 to calculate the overall dissipation rate of COPCs in surface water, resulting from volatilization and benthic burial. The equation is also presented in Appendix B, Table B-4-17.

Recommended Equation for Calculating:
Overall Total Water Body COPC Dissipation Rate Constant (k_{wt})

$$k_{wt} = f_{wc} \cdot k_v + f_{bs} \cdot k_b \qquad \text{Equation 5-38}$$

where

k_{wt} = Overall total water body dissipation rate constant (yr^{-1})
f_{wc} = Fraction of total water body COPC concentration in the water column (unitless)
k_v = Water column volatilization rate constant (yr^{-1})
f_{bs} = Fraction of total water body COPC concentration in benthic sediment (unitless)
k_b = Benthic burial rate constant (yr^{-1})

The variables f_{wc} and f_{bs} are discussed in Section 5.7.4.1. The water column volatilization rate constant (k_v) is discussed in Section 5.7.4.3. The benthic burial rate constant (k_b)is discussed in Section 5.7.4.7.

5.7.4.3 Water Column Volatilization Rate Constant (k_v)

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-39 to calculate k_v. The equation is also presented in Appendix B, Table B-4-18.

Recommended Equation for Calculating: Water Column Volatilization Rate Constant (k_v)

$$k_v = \frac{K_v}{d_z \cdot (1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6})}$$ Equation 5-39

where

k_v = Water column volatilization rate constant (yr^{-1})
K_v = Overall COPC transfer rate coefficient (m/yr)
d_z = Total water body depth (m)
Kd_{sw} = Suspended sediments/surface water partition coefficient (L water/kg suspended sediments)
TSS = Total suspended solids concentration (mg/L)
1×10^{-6} = Units conversion factor (kg/mg)

The overall transfer rate coefficient (K_v) is discussed in Section 5.7.4.4. Total water body depth (d_z), suspended sediment and surface water partition coefficient (Kd_{sw}), and total suspended solids concentration (TSS), are described in Section 5.7.4.1. Kd_{sw} is also discussed in Appendix A-2.

5.7.4.4 Overall COPC Transfer Rate Coefficient (K_v)

Volatile organic chemicals can move between the water column and the overlying air. The overall transfer rate K_v, or conductivity, is determined by a two-layer resistance model that assumes that two "stagnant films" are bounded on either side by well-mixed compartments. Concentration differences serve as the driving force for the water layer diffusion. Pressure differences drive the diffusion for the air layer. From balance considerations, the same mass must pass through both films; the two resistances thereby combine in series, so that the conductivity is the reciprocal of the total resistance.

As in U.S. EPA (1993f; 1993e; 1998c), and NC DEHNR (1997), we recommend using Equation 5-40 to calculate K_v. The equation is also presented in Appendix B, Table B-4-19.

Recommended Equation for Calculating:
Overall COPC Transfer Rate Coefficient (K_v)

$$K_v = \left(K_L^{-1} + \left(K_G \cdot \frac{H}{R \cdot T_{wk}} \right)^{-1} \right)^{-1} \cdot \theta^{T_{wk} - 293} \qquad \text{Equation 5-40}$$

where

K_v = Overall COPC transfer rate coefficient (m/yr)
K_L = Liquid phase transfer coefficient (m/yr)
K_G = Gas phase transfer coefficient (m/yr)
H = Henry's Law constant (atm-m^3/mol)
R = Universal gas constant (atm-m^3/mol-K)
T_{wk} = Water body temperature (K)
θ = Temperature correction factor (unitless)

The liquid and gas phase transfer coefficients, K_L and K_G, respectively, vary with the type of water body. We discuss the liquid phase transfer coefficient (K_L) in Section 5.7.4.5, and the gas phase transfer coefficient (K_G) in Section 5.7.4.6.

Henry's Law constants generally increase with increasing vapor pressure of a COPC and generally decrease with increasing solubility of a COPC. Henry's Law constants are COPC-specific and we offer recommended default values in the HHRAP companion database. The universal ideal gas constant, R, is 8.205×10^{-5} atm-m^3/mol-K, at 20°C. The temperature correction factor (θ), which is equal to 1.026, adjusts for the actual water temperature. Equation 5-40 assumes that volatilization occurs much less readily in lakes and reservoirs than in moving water bodies.

The value of the conductivity K_v depends on the intensity of turbulence in the water body and the overlying atmosphere. As Henry's Law constant increases, the conductivity tends to be increasingly influenced by the intensity of turbulence in water. Conversely, as Henry's Law constant decreases, the value of the conductivity tends to be increasingly influenced by the intensity of atmospheric turbulence.

5.7.4.5 Liquid Phase Transfer Coefficient (K_L)

We generally recommend using Equations 5-41A and 5-41B to calculate liquid phase transfer coefficient. (K_L). The use of these equations is further described in Appendix B, Table B-4-20.

Recommended Equation for Calculating: Liquid Phase Transfer Coefficient (K_L)

For flowing streams or rivers:

$$K_L = \sqrt{\frac{(1 \times 10^{-4}) \cdot D_w \cdot u}{d_z}} \cdot 3.1536 \times 10^7 \qquad \text{Equation 5-41A}$$

For quiescent lakes or ponds:

$$K_L = (C_d^{0.5} \cdot W) \cdot (\frac{\rho_a}{\rho_w})^{0.5} \cdot \frac{k^{0.33}}{\lambda_z} \cdot (\frac{\mu_w}{\rho_w \cdot D_w})^{-0.67} \cdot 3.1536 \times 10^7 \qquad \text{Equation 5-41B}$$

where

K_L = Liquid phase transfer coefficient (m/yr)
D_w = Diffusivity of COPC in water (cm^2/s)
u = Current velocity (m/s)
1×10^{-4} = Units conversion factor (m^2/cm^2)
d_z = Total water body depth (m)
C_d = Drag coefficient (unitless)
W = Average annual wind speed (m/s)
ρ_a = Density of air (g/cm^3)
ρ_w = Density of water (g/cm^3)
k = von Karman's constant (unitless)
λ_z = Dimensionless viscous sublayer thickness (unitless)
μ_w = Viscosity of water corresponding to water temperature (g/cm-s)
3.1536×10^7 = Units conversion factor (s/yr)

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using the following default values:

- a diffusivity of chemical in water ranging (D_w) from 1.0×10^{-5} to 8.5×10^{-2} cm^2/s,
- a drag coefficient (C_d) of 0.0011 which was adapted from U.S. EPA (1998c),
- a density of air (ρ_a) of 0.0012 g/cm^3 at standard conditions (temperature = 20°C or 293 K, pressure = 1 atm or 760 millimeters of mercury) (Weast 1986),
- a density of water (ρ_w) of 1 g/cm^3 (Weast 1986),
- a von Karman's constant (k) of 0.4,
- a dimensionless viscous sublayer thickness (λ_z) of 4,
- a viscosity of water (μ_w) of a 0.0169 g/cm-s corresponding to water temperature (Weast 1986).

The values above are further discussed in Appendix A-2. Chapter 4 discusses the current velocity (u). Chapter 3 describes methods for determining the average annual wind speed (W). Section 5.7.4.1 discusses the total water body depth (d_z) for liquid phase transfer coefficients.

For a flowing stream or river, the transfer coefficients are controlled by flow-induced turbulence. For these systems, we recommend calculating K_L using Equation 5-41A, which is the O'Connor and Dobbins (1958) formula, as presented in U.S. EPA (1998c).

For a stagnant system (quiescent lake or pond), the transfer coefficient is controlled by wind-induced turbulence. For quiescent lakes or ponds, we recommend calculating K_L using Equation 5-41B (O'Connor 1983; U.S. EPA 1998c).

5.7.4.6 Gas Phase Transfer Coefficient (K_G)

We generally recommend using Equations 5-42A and 5-42B to calculate gas phase transfer coefficient (K_G). The equation is also discussed in Appendix B, Table B-4-21.

Recommended Equation for Calculating:
Gas Phase Transfer Coefficient (K_G)

For flowing streams or rivers:

$$K_G = 36500 \text{ m/yr}$$

Equation 5-42A

For quiescent lakes or ponds:

$$K_G = (C_d^{0.5} \cdot W) \cdot \frac{k^{0.33}}{\lambda_z} \cdot \left(\frac{\mu_a}{\rho_a \cdot D_a}\right)^{-0.67} \cdot 3.1536 \times 10^7$$

Equation 5-42B

where

K_G	=	Gas phase transfer coefficient (m/yr)
C_d	=	Drag coefficient (unitless)
W	=	Average annual wind speed (m/s)
k	=	von Karman's constant (unitless)
λ_z	=	Dimensionless viscous sublayer thickness (unitless)
μ_a	=	Viscosity of air corresponding to air temperature (g/cm-s)
ρ_a	=	Density of air corresponding to water temperature (g/cm^3)
D_a	=	Diffusivity of COPC in air (cm^2/s)
3.1536×10^7	=	Units conversion factor (s/yr)

The following parameters, including default values, are discussed in Section 5.7.4.5, and in Appendix A-2: C_d, k, λ_z, and ρ_a. Chapter 3 describes methods for determining the average annual wind speed (W). As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using a value of 1.81 x 10^{-4} g/cm-s for the viscosity of air corresponding to air temperature (μ_a). Appendix A-2 discusses the COPC-specific parameter D_a.

U.S. EPA (1998c) indicated that the rate of transfer of a COPC from the gas phase for a flowing stream or river is assumed to be constant, in accordance with O'Connor and Dobbins (1958) (Equation 5-42A).

For a stagnant system (quiescent lake or pond), the transfer coefficients are controlled by wind-induced turbulence. For quiescent lakes or ponds, we recommend calculating the gas phase transfer coefficient using the equation presented in O'Connor (1983) (Equation 5-42B).

5.7.4.7 Benthic Burial Rate Constant (k_b)

We generally recommend using Equation 5-43 to calculate benthic burial rate (k_b). The equation is also discussed in Appendix B, Table B-4-22.

Recommended Equation for Calculating:
Benthic Burial Rate Constant (k_b)

$$k_b = \left(\frac{X_e \cdot A_L \cdot SD \cdot 1 \times 10^3 - Vf_x \cdot TSS}{A_W \cdot TSS} \right) \cdot \left(\frac{TSS \cdot 1 \times 10^{-6}}{C_{BS} \cdot d_{bs}} \right) \qquad \text{Equation 5-43}$$

where

k_b = Benthic burial rate constant (yr^{-1})
X_e = Unit soil loss (kg/m^2-yr)
A_L = Total watershed area (evaluated) receiving deposition (m^2)
SD = Sediment delivery ratio (watershed) (unitless)
Vf_x = Average volumetric flow rate through water body (m^3/yr)
TSS = Total suspended solids concentration (mg/L)
A_W = Water body surface area (m^2)
C_{BS} = Bed sediment concentration (g/cm^3)
d_{bs} = Depth of upper benthic sediment layer (m)
1 x 10^{-6} = Units conversion factor (kg/mg)
1 x 10^3 = Units conversion factor (g/kg)

Section 5.7.2 discusses the unit soil loss (X_e). Section 5.7.3 discusses watershed area evaluated receiving COPC deposition (A_L) and sediment delivery ratio (*SD*). Average volumetric flow rate through the water body (Vf_x) and water body surface area (A_w) are discussed in Appendix B. A_w is also discussed in Appendix A-2. Section 5.7.4.1 discusses total suspended solids concentration (*TSS*) and bed sediment concentration (C_{BS}). Section 5.7.4 discusses the depth of the upper benthic sediment layer (d_{bs}).

It's possible to express the benthic burial rate constant (k_b), which is calculated in Equation 5-43, in terms of the rate of burial (*Wb*):

$$Wb = k_b \cdot d_{bs} \qquad \text{Equation 5-44}$$

where

Wb = Rate of burial (m/yr)
k_b = Benthic burial rate constant (yr^{-1})
d_{bs} = Depth of upper benthic sediment layer (m)

According to U.S. EPA (1994r) and NC DEHNR (1997), COPC loss from the water column resulting from burial in benthic sediment can be calculated using Equation 5-43.

We expect k_b values to range from 0 to 1.0: Low k_b values for water bodies with limited or no sedimentation (rivers and fast flowing streams), and k_b values closer to 1.0 for water bodies characteristic of higher sedimentation (lakes). This range of values is based on the relation between the benthic burial rate and rate of burial expressed in Equation 5-44; with the depth of upper benthic sediment layer held constant. If you calculate a negative k_b value (water bodies with high average annual volumetric flow rates in comparison to watershed area evaluated), we recommend using a k_b value of 0 in calculating the total water body COPC concentration (C_{wtot}) in Equation 5-35. If the calculated k_b value exceeds 1.0, we recommend re-evaluating the parameter values used in calculating X_e. Our experience has shown that the value calculated for X_e is the most likely reason for estimating a large and potentially unrealistic benthic burial rate. Information about determining site-specific values and variables for calculating Xe are in the references cited in Section 5.7.2.

5.7.4.8 Total COPC Concentration in Water Column (C_{wctot})

As in U.S. EPA (1994r) and NC DEHNR (1997), we generally recommend using Equation 5-45 to calculate total COPC concentration in water column (C_{wctot}). The equation is also discussed in Appendix B, Table B-4-23.

Recommended Equation for Calculating:
Total COPC Concentration in Water Column (C_{wctot})

$$C_{wctot} = f_{wc} \cdot C_{wtot} \cdot \frac{d_{wc} + d_{bs}}{d_{wc}} \qquad \text{Equation 5-45}$$

where

C_{wctot} = Total COPC concentration in water column (mg COPC/L water column)
f_{wc} = Fraction of total water body COPC concentration in the water column (unitless)
C_{wtot} = Total water body COPC concentration, including water column and bed sediment (mg COPC/L water body)
d_{wc} = Depth of water column (m)
d_{bs} = Depth of upper benthic sediment layer (m)

We discussed the fraction of total water body COPC concentration in the water column (f_{wc}) in Section 5.7.4.1. We discussed the total COPC Concentration in the water column (C_{wctot}), as well as depth of the water column (d_{wc}) and benthic sediment layer (d_{bs}) in Section 5.7.4.

5.7.4.9 Dissolved Phase Water Concentration (C_{dw})

We recommend using Equation 5-46 to calculate the concentration of COPC dissolved in the water column (C_{dw}). The equation is discussed in detail in Appendix B, Table B-4-24.

Recommended Equation for Calculating:
Dissolved Phase Water Concentration (C_{dw})

$$C_{dw} = \frac{C_{wctot}}{1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}} \qquad \text{Equation 5-46}$$

where

C_{dw} = Dissolved phase water concentration (mg COPC/L water)
C_{wctot} = Total COPC concentration in water column (mg COPC/L water column)
Kd_{sw} = Suspended sediments/surface water partition coefficient (L water/kg suspended sediment)
TSS = Total suspended solids concentration (mg/L)
1×10^{-6} = Units conversion factor (kg/mg)

We discussed C_{wctot} in Section 5.7.4.8. We discussed Kd_{sw} and TSS in Section 5.7.4.1.

Using Equation 5-46 to calculate the concentration of COPC dissolved in the water column is consistent with recommendations in U.S. EPA (1994r) and NC DEHNR (1997).

5.7.4.10 COPC Concentration Sorbed to Bed Sediment (C_{sb})

We recommend using Equation 5-47 to calculate COPC concentration sorbed to bed sediment (C_{sb}). The equation is also presented in Appendix B, Table B-4-25.

Recommended Equation for Calculating:
COPC Concentration Sorbed to Bed Sediment (C_{sb})

$$C_{sb} = f_{bs} \cdot C_{wtot} \cdot \left(\frac{Kd_{bs}}{\theta_{bs} + Kd_{bs} \cdot C_{BS}} \right) \cdot \left(\frac{d_{wc} + d_{bs}}{d_{bs}} \right) \qquad \text{Equation 5-47}$$

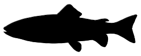
where

C_{sb} = COPC concentration sorbed to bed sediment (mg COPC/kg sediment)
f_{bs} = Fraction of total water body COPC concentration in benthic sediment (unitless)
C_{wtot} = Total water body COPC concentration, including water column and bed sediment (mg COPC/L water body)
Kd_{bs} = Bed sediment/sediment pore water partition coefficient (L COPC/kg water body)
θ_{bs} = Bed sediment porosity ($L_{pore\ water}/L_{sediment}$)
C_{BS} = Bed sediment concentration (g/cm^3)
d_{wc} = Depth of water column (m)
d_{bs} = Depth of upper benthic sediment layer (m)

We discussed f_{bs}, θ_{bs}, and C_{BS} in Section 5.7.4.1. We discussed C_{wtot}, d_{wc}, and d_{bs} in Section 5.7.4. We discuss Kd_{bs} in Appendix A-2.

As in U.S. EPA (1994r) and NC DEHNR (1997), we continue to recommend using Equation 5-47 to calculate the COPC concentration sorbed to bed sediment.

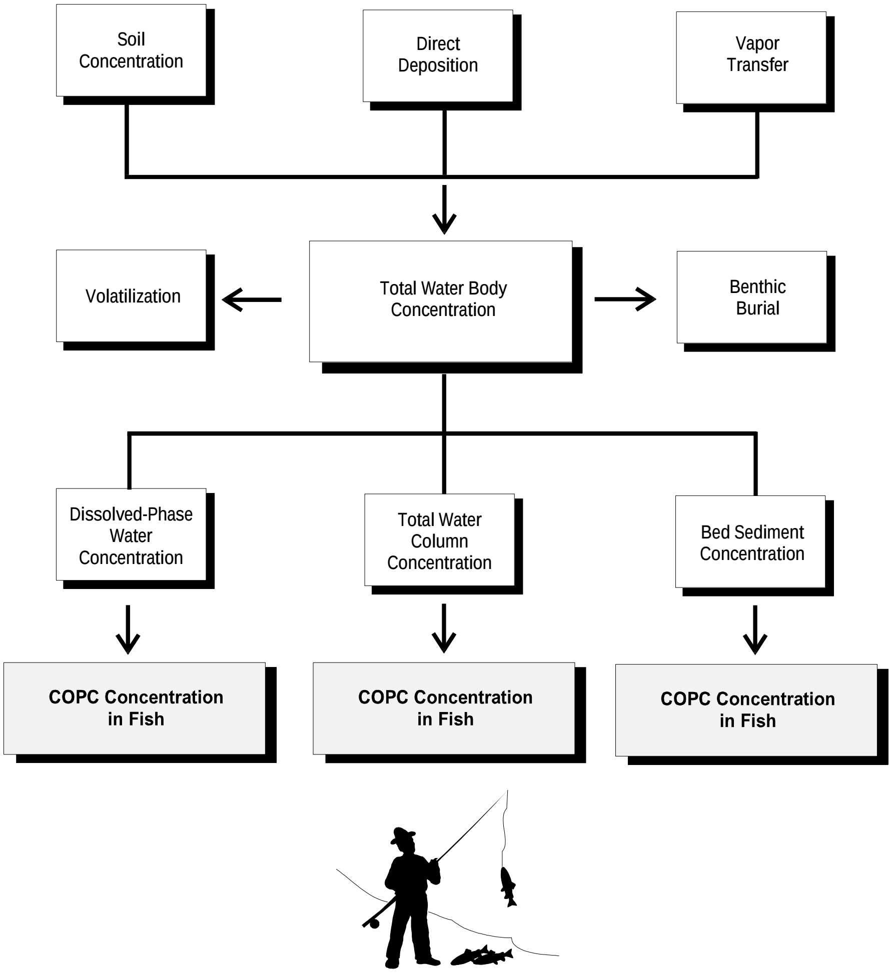
5.7.5 Concentration of COPC in Fish (C_{fish})



We generally recommend calculating the COPC concentration in fish using either a COPC-specific bioconcentration factor (*BCF*), a COPC-specific bioaccumulation factor (*BAF*), or a COPC-specific biota-sediment accumulation factor (*BSAF*). Under this approach you would use *BCFs* for COPCs with a $log\ K_{ow}$ less than 4.0. We assume that COPCs with a $log\ K_{ow}$ greater than 4.0 (except for extremely hydrophobic compounds such as dioxins, furans, and PCBs), have a high tendency to bioaccumulate. As a result, *BAF*s are used. While we assume that extremely hydrophobic COPCs like dioxins, furans, and PCBs also have a high tendency to bioaccumulate, they are expected to be sorbed to the bed sediments more than associated with the water phase. Therefore, we recommend using *BSAF*s to calculate concentrations of dioxins, furans, and PCBs in fish. Appendix A-2 provides a detailed discussion on the sources of the COPC-specific *BCF*, *BAF*, and *BSAF* values, and the method we used to derive them.

FIGURE 5-8

COPC CONCENTRATION IN FISH



BCF and *BAF* values are generally based on dissolved water concentrations. Therefore, when you use *BCF* or *BAF* values, it's appropriate to calculate the COPC concentration in fish using dissolved water concentrations. *BSAF* values are based on benthic sediment concentrations. Therefore, when using *BSAF* values, we recommend calculating COPC concentrations in fish using benthic sediment concentrations. We describe our recommended equations for calculating fish concentrations in the subsequent subsections.

5.7.5.1 Fish Concentration (C_{fish}) from Bioconcentration Factors Using Dissolved Phase Water Concentration

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-48 to calculate fish concentration from *BCF*s using dissolved phase water concentration. Using this equation is further described in Appendix B, Table B-4-26.

Recommended Equation for Calculating:
Fish Concentration (C_{fish}) from Bioconcentration Factors (BCF_{fish})
Using Dissolved Phase Water Concentration

$$C_{fish} = C_{dw} \cdot BCF_{fish} \qquad \text{Equation 5-48}$$

where

C_{fish} = Concentration of COPC in fish (mg COPC/kg FW tissue)
C_{dw} = Dissolved phase water concentration (mg COPC/L)
BCF_{fish} = Bioconcentration factor for COPC in fish (L/kg)

We discussed C_{dw} in Section 5.7.4.9. COPC-specific BCF_{fish} values are presented in the HHRAP companion database.

5.7.5.2 Fish Concentration (C_{fish}) from Bioaccumulation Factors Using Dissolved Phase Water Concentration

We recommend using Equation 5-49 to calculate fish concentration from *BAF*s using dissolved phase water concentration. The equation is also presented in Appendix B, Table B-4-27.

Recommended Equation for Calculating:
Fish Concentration (C_{fish}) from Bioaccumulation Factors (BAF_{fish})
Using Dissolved Phase Water Concentration

$$C_{fish} = C_{dw} \cdot BAF_{fish} \qquad \text{Equation 5-49}$$

where

C_{fish} = Concentration of COPC in fish (mg COPC/kg FW tissue)
C_{dw} = Dissolved phase water concentration (mg COPC/L)
BAF_{fish} = Bioaccumulation factor for COPC in fish (L/kg FW tissue)

We discussed C_{dw} in Section 5.7.4.9. COPC-specific bioaccumulation factor (BAF_{fish}) values are presented in the HHRAP companion database.

5.7.5.3 Fish Concentration (C_{fish}) from Biota-To-Sediment Accumulation Factors Using COPC Sorbed to Bed Sediment

As in U.S. EPA (1994r) and NC DEHNR (1997), we recommend using Equation 5-50 to calculate fish concentration from *BSAF*s using COPC concentrations sorbed to bed sediment. We recommend using *BSAFs* for very hydrophobic compounds such as dioxins, furans, and PCBs. The equation is also presented in Appendix B, Table B-4-28.

Recommended Equation for Calculating:
Fish Concentration (C_{fish}) from Biota-To-Sediment Accumulation Factors *(BSAF)*
Using COPC Sorbed to Bed Sediment

$$C_{fish} = \frac{C_{sb} \cdot f_{lipid} \cdot BSAF}{OC_{sed}} \qquad \text{Equation 5-50}$$

where

C_{fish} = Concentration of COPC in fish (mg COPC/kg FW tissue)
C_{sb} = Concentration of COPC sorbed to bed sediment (mg COPC/kg bed sediment)
f_{lipid} = Fish lipid content (unitless)
$BSAF$ = Biota-to-sediment accumulation factor (unitless)
OC_{sed} = Fraction of organic carbon in bottom sediment (unitless)

We discussed C_{sb} in Section 5.7.4.10. We discuss f_{lipid} and OC_{sed} below. Our recommended default values for f_{lipid} and OC_{sed} are given in Appendix B, Table B-4-28. We offer biota-to-sediment accumulation factors (*BSAF*), which are applied only to dioxins, furans, and PCBs, in the HHRAP companion database.

Values recommended by U.S. EPA (1998c) range from 0.03 to 0.05 for $_{OCSED}$. These values are based on an assumption of a surface soil organic carbon (OC) content of 0.01. U.S. EPA (1998c) states that the organic carbon content in bottom sediments is higher than the organic carbon content in soils because (1) erosion favors lighter-textured soils with higher organic carbon contents, and (2) bottom sediments are partially comprised of detritus materials.

The fish lipid content (f_{lipid}) value is site-specific and dependent on the type of fish consumed. As stated in Appendix B, Table B-4-28, we recommend a default range of 0.03 to 0.07 specific to warm or cold water fish species. U.S. EPA (2000c) provides information supporting a value of 0.03 (3 percent lipid content of the edible portion). U.S. EPA (1993e) recommended a default value of 0.04 for $_{OCSED}$, which is the midpoint of the specified range. U.S. EPA (1993f; 1993e) recommended using 0.07, which was originally cited in Cook et al. (1991).

5.8 USING SITE-SPECIFIC vs. DEFAULT PARAMETER VALUES

As initially discussed in Chapter 1, many of the parameter values we recommend in the HHRAP are not site-specific. After completing a risk assessment using HHRAP default values, you might choose to investigate using site-specific parameter values. More site-specific values might provide a more representative estimate of site-specific risk. If you use parameter values other than those specified in the HHRAP, we recommend that you clearly described them in the risk assessment report. We also recommend that you discuss them with the permitting authority prior to using them. We recommend that requests to change default parameter values include the following information, as appropriate:

1. An explanation of why using a more site-specific parameter value is warranted (e.g., the default parameter is based on data or studies at sites in the northwestern U.S., but the facility is located in the southeast);
2. The technical basis of the site-specific parameter value, including readable copies of any relevant technical literature or studies;

3. A comparison of the weight-of-evidence between the competing studies (e.g., the site-specific parameter value is based on a study that is more representative of site conditions, a specific exposure setting being evaluated, or a more scientifically valid study than the default parameter, the site-specific parameter is based on the analysis of 15 samples as opposed to 5 for the default parameter, or the site-specific study used more stringent quality control/quality assurance procedures than the study upon which the default parameter is based);

4. A description of other risk assessments or projects that used the site-specific parameter value, and how such risk assessments or projects are similar to the current risk assessment.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

C Identification of site-specific or alternate default media equations and/or inputs; including justification and full referencing

C Media concentration calculations

Chapter 6
Quantifying Exposure

What's Covered in Chapter 6:

6.1 Inhalation Exposure Pathways
6.2 Ingestion Exposure Pathways
6.3 Dermal Exposure Pathways
6.4 Exposure Frequency
6.5 Exposure Duration
6.6 Averaging Time

This chapter describes the factors to evaluate in quantifying the exposure received under each of the recommended exposure scenarios described in Chapter 4. Calculating COPC-specific exposure rates for each exposure pathway involves some or all of the following, depending upon the medium being assessed:

- the estimated COPC media concentrations calculated in Chapter 5,
- consumption rates of the medium,
- receptor body weight, and
- the frequency and duration of exposure.

We recommend repeating the appropriate calculation for each COPC and for each exposure pathway included in an exposure scenario, to generate multiple exposure concentration estimates, as recommended in the EPA information quality guidelines (see Chapter 1, page 1-11). We present recommended exposure pathway-specific equations in Appendix C. The following sections describe a general exposure rate calculation and the exposure pathway-specific variables that may affect this calculation. Acute exposure resulting from direct inhalation is discussed as a separate issue in Chapter 7, Section 7.4.

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

6.1 INHALATION EXPOSURE PATHWAYS

We recommend using COPC air concentrations calculated using the equation in Table B-5-1 to represent air concentrations for estimating exposure via inhalation by all exposure scenarios in the risk assessment (see Table 4-1).

Direct inhalation of vapors and particulate emissions from combustion sources is a potential pathway of exposure. Chapter 2 presented various variables and conditions that affect the rate, type, and quantity of combustor emissions. Chapter 3 presented the air dispersion and deposition modeling techniques we recommend using to estimate airborne concentrations of vapors and particulates in the assessment area.

As a result of normal respiration, receptors in the assessment area could be exposed to COPCs in vapor, particle, and particle-bound phases. Examples of factors that affect exposure from vapor and particulate inhalation include vapor and particulate COPC concentrations, particle size, and length of exposure.

Exposure can occur over a period of time. To calculate an average exposure per unit of time (Exposure Concentration, or *EC*), we recommend dividing the total exposure by the time period. We generally recommend using total COPC air concentrations (C_a, estimated using the equation in Table B-5-1) when estimating *EC* values. Estimating *EC*s doesn't involve or require adjustment for respiration rates, as those are inherent to inhalation toxicity factors. Sections 6.4 through 6.6 discuss exposure time-related parameters, and Appendix C Tables C-2-1 and C-2-2 further discuss estimating *EC*.

We consider it appropriate to estimate noncarcinogenic hazards and carcinogenic risks associated with direct inhalation exposure by combining *EC*s with inhalation toxicity factors (reference concentrations [RfCs] or unit risk factors [URFs]). These toxicity factors are developed for all human populations, including sensitive subpopulations (including children) who might be exposed to continuous concentrations over a lifetime. Inhalation risk parameters and inhalation pharmacokinetics are largely chemical-specific and a "one size fits all" approach to convert the standard RfC or URF parameters into scenario-specific toxicity values may not be appropriate. We therefore generally recommend that a single (lifetime) inhalation risk be predicted for each receptor identified in Chapter 4, Table 4-1.

Inhalation exposure concentrations for vapors and particles (arising from outdoor sources) can be influenced by the relative amount of time that a receptor spends indoors. Although vapors entering buildings and residences as a result of air exchange are likely to remain airborne and, therefore may be inhaled, particulates entering these same buildings are more likely to settle out and not be inhaled. However, for the purpose of the risk assessment, we recommend assuming that both vapor and particulates are inhaled throughout the day, both indoors and outdoors.

6.1.1 Soil Inhalation Resulting from Dust Resuspension

We don't typically recommend evaluating the soil inhalation of resuspended dust exposure pathway. However, site-specific exposure setting characteristics might support evaluating it (e.g. arid, windy climates). This section therefore discusses exposure to soil resulting from dust resuspension.

Inhalation of soil resulting from dust resuspension could be an issue for site-specific exposure scenario locations at which there is little vegetative cover. Wind erosion could resuspend pollutants in contaminated soil as particles in the air. As dust is resuspended, receptors could inhale the pollutant particles (direct inhalation of particulate matter is addressed in Section 6.1). The amount resuspended depends on the:

- moisture content of the soil,
- fraction of vegetation cover,
- wind velocity,
- soil particle size,
- pollutant concentration in the soil, and
- size of the contaminated area.

Study of estimated exposures to deposited combustor emissions via dust resuspension indicates that dust resuspension by wind erosion is usually not a significant pathway (U.S. EPA 1998c). Methods have also been developed to assess the exposure to pollutants resuspended by wind erosion for landfills and Superfund sites (U.S. EPA 1985a; 1988b; 1994q). We recommend consulting these reference documents if you'll be evaluating this exposure pathway. Also, it may be useful to review the methods described in U.S. EPA (1998c).

6.2 INGESTION EXPOSURE PATHWAYS

Exposure can occur over a period of time. To calculate an average exposure per unit of time, we recommend dividing the total exposure by the time period. Express an average exposure in terms of body weight. Ingestion exposures quantified per the HHRAP are

- unitized for time and body weight,
- presented in units of milligrams per kilogram of body weight per day, and
- termed "intakes."

Equation 6-1 is a generic equation for calculating ingestion chemical intake (U.S. EPA 1989e):

$$I = \frac{C_{gen} \cdot CR \cdot EF \cdot ED}{BW \cdot AT} \qquad \text{Equation 6-1}$$

where

I = Intake—the amount of COPC at the exchange boundary (mg/kg/day); for evaluating exposure to noncarcinogenic COPCs, the intake is referred to as average daily dose (*ADD*); for evaluating exposure to carcinogenic compounds, the intake is referred to as lifetime average daily dose (*LADD*)
C_{gen} = Generic COPC concentration in medium of concern (e.g., mg/kg for soil or mg/L for surface water; see Chapter 5)
CR = Consumption rate—the amount of contaminated medium consumed per unit of time or event (e.g., kg/day for soil or L/day for water)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Average body weight of the receptor over the exposure period (kg)
AT = Averaging time—the period over which exposure is averaged (days); for carcinogens, the averaging time is 25,550 days, based on a lifetime exposure of 70 years; for noncarcinogens, averaging time equals *ED* (years) multiplied by 365 days per year.

We recommend variations of Equation 6-1 to calculate pathway- and receptor-specific exposures to COPCs. We present the equations recommended for each exposure pathway in Appendix C. The variation of input variables is also described in Appendix C.

The exposures calculated using the HHRAP are intended to represent reasonable maximum exposure (RME) conditions, as further described in U.S. EPA (1989e). Studies of the compounding of conservatism in probabilistic risk assessments show that setting as few as two factors at RME levels or high end (e.g., near the 90th percentile), while setting the remaining variables at less protective typical, or "central tendency" values (e.g., near the 50th percentile) results in output insignificantly different from output generated using all input variables at an RME level (e.g., 99th percentile value) (Cullen 1994).

As described in Chapter 2 (Section 2.2.1), the estimated air concentrations and deposition rates are based on RME emissions from trial or risk burns. We recommend setting the following variables set at RME levels:

- the highest ISCST3 modeled air concentrations and deposition rates at chosen exposure scenario locations,
- the exposure frequency, and
- the exposure duration.

We generally recommend setting other exposure parameters (e.g. body weight) at average levels.

6.2.1 Body Weight

The choice of body weight to use in the risk characterization equations presented in Appendix C depends on the definition of the receptor at risk—which, in turn, depends on exposure and susceptibility to adverse effects. U.S. EPA (1990e) defines the body weight of the receptor as either adult weight (70 kilograms) or child weight (1 to 7 years; 17 kilograms) on the basis of data presented in Nelson et al. (1969). However, as in other Agency guidance (U.S. EPA 1991b; 1994r; 1994g), we recommend using a weight of 15 kilograms for the child (exposure duration of 6 years) in the risk assessment.

The daily intake for an exposure pathway is expressed as the dose per body weight. Because children have lower body weights than adults. Typical ingestion exposures per body weight, such as for soil, milk, and fruits & vegetables, can be substantially higher for children. This is the primary reason to evaluate the child resident scenario (U.S. EPA 1996g). However, using these two body weights may not account for significant differences between weights of infants and toddlers or weights of teenagers and adults. Please remember that, for the purposes of the risk assessment, the child scenario is defined by the average body weight, rather than the chronological age. Obviously, the weight of a child changes significantly over the first several years. We assume 15 kilograms is a realistic average estimate for an exposure duration of 6 years. 15 kilograms overestimates the weight of the child for the early years, and then underestimates it for the later years (U.S. EPA 1996g).

6.2.2 Food (Ingestion) Exposure Pathways

Plants and animals impacted by emission sources may take up emitted COPCs in the air or deposited COPCs in soil. Humans could then be exposed to COPCs via the food chain when they consume these plants and animals as food. We generally recommend determining human intake of COPCs based on the:

- types of foods consumed,
- amount of food consumed per day,
- concentration of COPCs in the food, and
- percentage of the diet contaminated by COPCs.

Chapter 6 describes procedures for determining the concentration of COPCs in food. It also considers the variations in exposure resulting from food preparation methods and type of food item (e.g., aboveground versus belowground - i.e. root - vegetables). Other variables, described below, may also significantly affect exposure estimates.

6.2.2.1 Types of Foods Consumed

The types of foods consumed will affect exposure, because different plant and animal tissues take up different COPCs, and take them up at different rates. The COPC concentrations a receptor is exposed to will then vary with the types of food in the diet. Therefore, it is important to determine COPC concentrations in food according to the type of food. The types of food consumed might also vary with the age of the receptor, geographical region, and socio-cultural factors.

6.2.2.2 Food Consumption Rate

Consumption rate is the amount of contaminated medium (soil, food) consumed per unit of time or event. The soil at an exposure location are inherent to the location. Food consumed at an exposure location, however, may or may not have originated there. The HHRAP assumes that only food produced at the exposure location is contaminated by emissions from the facility being assessed. Food not produced at the point of exposure is not assumed to be contaminated, and is irrelevant to the assessment. Therefore, the consumption rates we recommend in the HHRAP (see Table 6-1 and Appendix C) are for food that is both produced and consumed at the exposure location (i.e. at home).

> ***Please Note***: these rates do not represent the entire dietary intake of the individual, but only that portion of the diet produced at home. For example, the beef consumption rate represents only the amount of beef consumed each day which was raised on the farm property.

TABLE 6-1

MEAN CONSUMPTION RATES[a] FOR RECOMMENDED EXPOSURE SCENARIOS
(number of servings per week)

Contaminated food	Exposure Scenario					
	Farmer [b]	Farmer Child[b]	Resident	Resident Child	Fisher	Fisher Child
Produce (8 oz servings)	2.8	1.4	2.3	1.2	2.3	1.2
Beef (1/4 lb servings)	5.3	0.7	N/A	N/A	N/A	N/A
Milk (8 oz servings)	29.5	10.5	N/A	N/A	N/A	N/A
Chicken (1/4 lb servings)	2.8	0.4	N/A	N/A	N/A	N/A
Eggs (number[c])	4.3	0.7	N/A	N/A	N/A	N/A
Pork (1/4 lb servings)	2.4	0.4	N/A	N/A	N/A	N/A
Fish (1/4 lb servings)	N/A	N/A	N/A	N/A	5.4	0.8

Notes:
[a] Values derived from the U.S. EPA Exposure Factors Handbook (1997).
[b] Values based on consumption rates of a 154 lb adult and a 33 lb child.
[c] Values based on an assumed egg weight of 3.0 ounces.

As described in Section 6.2, exposures calculated using HHRAP methods are intended to represent RME conditions. Accordingly, the HHRAP recommends default values for exposure parameters that will result in RME estimates. However, there are likely to be differences between recommended default, and regional and site-specific exposure parameter values. This may be especially true for the consumption rates (a general term for intake rates and inhalation rates). In addition to estimates using the recommended default parameter values, you can refine the your risk assessment by including supplemental calculations using regional- or site-specific parameter values. We recommend doing this only if you document the regional- or site-specific parameter values in the risk assessment report. We recommend providing these supplemental calculations in addition to, and not instead of estimates based on recommended default exposure parameter values. This will help standardize risk assessment methods, thereby aiding the ability to compare outputs from different risk assessments. The following subsections describe exposure pathway-specific considerations regarding consumption rate.

This section gives some of the pertinent history of consumption rates. It also describes the series of steps we followed to derive the recommended consumption rates found in Table 6-1. Site-specific conditions might exist such that you need to derive an alternative, or additional consumption rate. For example, you may need a consumption rate for an additional exposure scenario (e.g. consumption of deer meat for a hunter scenario). If you need to calculate your own consumption rate(s), we recommend using the process described below. For transparency and clarity, we recommend identifying all consumption rates used in the risk assessment. We also recommend clearly identifying and discussing alterations to recommended default rates, and additional consumption rates, with the permitting authority prior to use.

The amount of daily food consumption varies with age, sex, body weight, and geographic region, and it also varies within these categories. U.S. EPA (1998c) recommended using values from the 1997 *Exposure Factors Handbook* (EFH) (U.S. EPA 1997b) to complete the risk assessment process. The EFH used the 1987-1988 USDA Food Consumption Survey to represent consumption rates for urban and suburban areas. However, if site-specific information indicates that the population is in a more rural or agricultural area, U.S. EPA (1990e) recommended using the 1966-67 USDA Food Consumption Survey to represent the consumption rates of a more agrarian population.

The 1997 *Exposure Factors Handbook* (EFH) (U.S. EPA 1997b) performed an analysis of the 1987-1988 USDA National Food Consumption Survey (NFCS). The NFCS collects information over a 7-day period on the socioeconomic and demographic characteristics of households, and the types, values, and sources of foods consumed. The following information was taken from the survey:

- whether or not the food product was used in the house that week;
- whether or not the food product used that week was home produced;
- the quantity (mass, such as pounds or kilograms) of food consumed (home produced or not) in the house that week;
- the number, age, and body weight of individuals in the household; and
- the number of weekly meals consumed by each family member.

All households were surveyed about the same food types, and consumption rates were averaged over the entire survey population - to calculate what are known in the EFH as the “Per Capita” rates. In addition, EPA calculated consumption rates for “consumers only:” a rate for only those households which consumed a particular food stuff during the week the survey was taken. In addition to total consumption (i.e. for the entire population of consumers), rates were broken out according to various criteria, such as age of consumer, geographic region, and level of urbanization. Survey participants were also asked if they operated a farm or ranch, raised animals, or had a home garden, and consumption rates were also broken out for these sub-groups. We recommend using food consumption rate information (ingestion rates) from the EFH; specifically, the section regarding home produced food items.

Chapter 13 of the EFH (Intake Rates for Various Home Produced Food Items) lists consumer only consumption rates of home produced food. For example, Table 13-65 (Consumer Only Intake of Home grown Root Vegetables (g/kg-day)) lists a mean consumption rate of 1.16 g/kg-day for all households that consumed root vegetables they produced themselves. From these households, the EFH also breaks out consumption rates specific to households who farm (e.g. a mean of 2.63 g/kg-day). These farm-specific rates represent the amount of food that farm families produced themselves that was consumed during the week the survey was taken. We recommend using these consumption rates to estimate exposures to the Farmer and Farmer Child scenarios.

The recommended farm-specific consumption rates, as listed in the EFH, are averaged across all family members, regardless of age or body weight. They also do not consider losses during preparation and cooking of the food. Additional work is needed, therefore, to acquire scenario-appropriate actual consumption rates:

As mentioned above, the EFH provides consumption rates for the entire population of consumers - the consumer only rate. Depending on data availability, the consumer only rate is also broken out into various age-specific demographics, covering 0-1 year, 1-2 years, 3-5 years, 6-11 years, 12-19 years, 20-39 years, and 40-69 years. We suggest combining the consumer only rate, age-appropriate demographic rates, and the sub-population rate (e.g. for a farm family) to derive a scenario-specific consumption rate (e.g. for an adult Farmer) using equation 6-2:

$$CR_{scen} = CR_{subpop} \times \frac{TWA_{gen}}{CR_{gen}} \qquad \text{Equation 6-2}$$

where:

CR_{scen} = Consumption rate for scenario
CR_{subpop} = Consumption rate of subpopulation
TWA_{gen} = Time-weighted average of age-appropriate subset of consumer only population
CR_{gen} = Consumption rate for consumer only population

Sufficient age-specific data were not always available. For example, there was insufficient data for the EFH to provide a consumer only poultry consumption rate for the 12-19 age group. In this event we suggest again using Equation 6-2, to combine the total consumer only population rate, the age-specific Per Capita rate, and the total Per Capita rate (for poultry, Per Capita values are found in EFH Table 11-11), to generate an age-specific consumer only rate.

See the example derivation below for a demonstration of how we recommend generating scenario-specific consumption rates. To derive consumption rates for the Farmer and Farmer Child exposure scenarios, we recommend using the home produced, consumers only consumption rates of households who farm, for home produced beef, pork, chicken, milk, eggs, vegetables, and fruits.

Consumption rates of households who farm are not necessarily the most appropriate rates for households who don't farm. For certain food types, the EFH also breaks out consumption rates of home-produced foods specific to those households who garden. This combination of subpopulation (i.e. gardeners) and food source (i.e. home produced) is the closest option available in the EFH to the residential and fishing scenarios we recommend in the HHRAP. Consumption rates for this subpopulation are available for the food types included in the produce-related exposure pathways we recommend (i.e. protected produce, exposed produce, and belowground vegetables). We recommend using these rates to generate produce-related consumption rates for the Resident, Resident Child, Fisher, and Fisher Child exposure scenarios.

The EFH provides information to account for cooking and post-cooking losses for food products which are home produced. These data are summarized in Table 6-2. See the example derivation below for a demonstration of how to use cooking loss data in deriving consumption rates.

TABLE 6-2

COOKING-RELATED WEIGHT LOSSES FOR VARIOUS HOME-PRODUCED FOODS

Percent Weight Losses from Preparation of Various Meats
(SOURCE: EFH Table 13-5)

Meat Type	Mean Net Cooking Loss (%)	Mean Net Post Cooking Loss (%)
Beef	27	24
Pork	28	36
Chicken	32	31
Milk	N/A	N/A
Eggs	N/A	N/A
Fish	30	11

Percent Weight Losses from Preparation of Various Produce
(SOURCE: EFH Tables 13-6 and 13-7)

Produce Type	Mean Paring or Preparation Loss (%)	Moisture Content (%)
Protected Fruits	29	87
Protected Vegetables	23	82
Exposed Fruit	21	85
Exposed Vegetables	16	90

Example Consumption Rate Derivation: Homegrown Poultry consumption for the Farmer

The Farmer scenario includes an average consumption rate during a 70-year lifetime, from age 7 to 70. Table 13-55 of the EFH lists the following Farmer-related mean consumption rates of home produced poultry (g/kg-day):

Total population	1.57	
broken out by age group:		
ages 6-11	ND	(5 years)
ages 12-19	ND	(8 years)
ages 20-39	1.17	(20 years)
ages 40-69	1.51	(30 years)
households who farm	1.54	

Please note that though the EFH demographic subset includes years 6-11, the Farmer scenario begins at age 7, so only 5 years of that consumption rate are used.

In addition, Table 11-11 (Mean Meat Intakes Per Individual in a Day, by Sex and Age (g/day as consumed) for 1987-1988) lists the following Per Capita consumption rates of poultry (among others):

All individuals	26
broken out by age group:	
ages 6-11	27
ages 12-19	27

Using the 12-19 age group as an example, equation 6-2 is used to calculate age-specific consumer only rates as follows:

$$CR_{ages12\text{-}19} = 1.57 * 27/26 = 1.63 \text{ g/kg-day}$$

A time weighted average for an adult Farmer =

$$TWA_a = \frac{(1.63 \times 5) + (1.63 \times 8) + (1.17 \times 20) + (1.51 \times 30)}{5 + 8 + 20 + 30} = 1.43$$

And, using Equation 6-2:

$$CR_{Farmer} = 1.54 * 1.43/1.57 = 1.40 \text{ g/kg-day}$$

Table 13-5 of the EFH lists cooking and post-cooking losses for poultry of 31% and 32%, respectively.

Therefore:

$$1.40 * (1-0.31)*(1-0.32) = 0.66$$

Poultry consumption rate for the Farmer = 0.66 g/kg-day

Additional consumption rate information is presented in Appendix C as follows: Table C-1-2 (produce); Table C-1-3 (beef, milk, pork, chicken, and eggs); and Table C-1-4 (fish).

6.2.2.3 Percentage of Contaminated Food

The percentage of food consumed by an individual which is home-grown will affect exposure, because the HHRAP assumes that only the portion of an individual's dietary intake which is home-grown is impacted by facility emissions.

We recommend assuming that all food produced at the exposure location - i.e. the farm for the farming scenarios, and the home garden for the residential and fishing scenarios - is impacted by facility emissions. Only that portion of the diet produced at home (and therefore exposed to facility emissions) is of consequence in the risk assessment. As detailed in Section 6.2.2.2, the consumption rates we recommend represent only the home-produced portion of the diet. Therefore, by using consumption rates specific to home produced foods, we consider it reasonable to assume that 100% of those home produced foods are contaminated.

6.2.3 Soil (Ingestion) Exposure Pathway

Soil ingestion, dermal exposure to soil, and inhalation of resuspended dust are potential soil exposure pathways. For the purpose of RCRA combustion permitting decisions, we recommend considering soil ingestion. However, we currently only recommend evaluating dermal exposure to soil (see Section 6.3) and inhalation of resuspended dust (see Section 6.1.1) if site-specific exposure setting characteristics support evaluating these exposure pathways. Based on air dispersion modeling and deposition of COPCs, emission concentrations in soil will vary with distance from the source. It's possible to determine potential routes of exposure by evaluating the way in which the soils in the area are used. Soil used for farming or recreation will be involved in pathways of human exposure that differ from those of soil on roadways or in urban areas.

Children and adults are exposed to COPCs in soil when they consume contaminated soil that has adhered to their hands. Factors that influence exposure by soil ingestion include soil concentration, the rate of soil ingestion during the time of exposure, and the length of time spent in the vicinity of contaminated soil. Soil ingestion rates in children are based on studies that measured the quantities of non-absorbable tracer minerals in the feces of young children. Ingestion rates for adults are based on assumptions about exposed surface area and frequency of hand-to-mouth activity. Indoor dust and outdoor soil may both contribute to the total daily ingestion. Exposure levels are also influenced by the amount of time that the individual spends in the vicinity of soil exposed to deposition of emitted pollutants.

In addition, some young children—referred to as “pica” children—may intentionally eat soil. As discussed in U.S. EPA (1989f), the typical medical and scientific use of the term “pica” refers to the ingestion of non-food items, such as soil, chalk, and crayons. Such behavior is considered a temporary behavior and a normal part of child development. For risk assessment purposes, pica is typically defined as “an abnormally high soil ingestion rate” and is believed to be uncommon in the general population (U.S. EPA 1989f). Agency risk assessment documents don’t identify a default “pica” soil ingestion rate (U.S. EPA 1989e; 1989f ; 1991b). Pica behavior is not generally included as part of risk assessments.

If available information indicates that there are children exhibiting pica behavior in the assessment area, and you determine that these children represent a special subpopulation potentially receiving significant exposure (see Chapter 4), it may be prudent to include these children in the risk assessment. We recommend making this evaluation on a case-by-case basis based on site-specific exposure setting characterization.

6.2.4 Water (Ingestion) Exposure Pathways

Evaluating HHRAP water exposure pathways involves estimating COPC concentrations in drinking water from surface water bodies or collected precipitation (e.g., cisterns). Contaminants moving through the water pathways also influence COPC concentrations in fish. Various models are available to estimate daily exposures of individuals using these water sources for various purposes, such as fishing and drinking water.

We recommend using site-specific information to determine which water exposure pathways to evaluate in the risk assessment. Whether it's collected precipitation, or from a surface water body such as a lake, farm pond, or city reservoir, the way in which water is used will suggest possible exposure pathways. For example, using a surface water body as a drinking water source will introduce water ingestion as a possible exposure pathway. Commercial and/or recreational fishing, with subsequent use of fish and shellfish as a food source, make the food chain an important route of exposure for communities having a surface water body in the vicinity of a combustor.

U.S. EPA (1998c) recommended varying the water input variables to determine a range of exposures. An individual that fishes and obtains drinking water from the same water source could represent an average exposure scenario. A worst-case possibility might involve a person who (1) uses drinking water from a cistern that collects precipitation, and (2) fishes in a small farm pond.

Because ground-level concentrations of COPCs generally decrease with distance from the source, important factors in determining the water concentration include:

- the location of the precipitation-collection apparatus,
- surface water body onto which emitted COPCs are deposited, and
- the COPC soil concentration (which affects runoff and leachate concentrations).

In addition, the location and size of the watershed will affect the concentration of COPCs suspended in runoff.

6.2.4.1 Ingestion of Drinking Water from Surface Water Sources

For evaluating a surface water body as a drinking water source, exposure is affected by the COPC concentration in the water, the daily amount of water ingested, and the length of time that the receptor spends in the area serviced by that water supply system. The equations we recommend for estimating the COPC concentration in a surface water body are discussed in Chapter 5 and Appendix B. These equations also consider contributions of COPC loading from the surrounding watershed. We recommend using the water consumption rates specified in U.S. EPA (1997b) and described in Appendix C.

As in previous Agency guidance (U.S. EPA 1998c), we recommend typically assuming that treatment processes for drinking water do not alter dissolved COPC concentrations.

6.2.4.2 Ingestion of Drinking Water from Ground Water Sources

For the purpose of RCRA combustion permitting decisions, we don't typically recommend evaluating exposure from ground water sources used as drinking water. Study of this pathway for combustor emissions indicates that this isn't a significant exposure pathway (U.S. EPA 1998c). However, COPCs may - because of special site-specific characteristics - infiltrate into ground water, resulting in COPC exposure via ingestion when ground water is used as drinking water. This could be because of extremely shallow aquifers being tapped for drinking water, or a karst environment in which the local surface water significantly affects the quality of ground water used as a drinking water source. The method developed to calculate risks from the ground water pathway was originally intended for use in evaluating impacts of the land disposal of various wastes (U.S. EPA 1998c; 1994q; 2003). We recommend consulting these reference documents if you intend to evaluate this exposure pathway.

6.2.4.3 Ingestion of Fish

You may find the fish ingestion rates specified in U.S. EPA (1997b) and further described in HHRAP Appendix C useful for evaluating the fish ingestion pathway. Factors that affect human exposure by ingestion of fish from a surface water body affected by combustion unit emissions include:

- COPC concentrations in the sediment and water column,
- the types of fish and shellfish consumed,
- the ingestion rates for the various fish and shellfish groups, and
- the percent of dietary fish caught in the surface water body affected by the combustor.

The types of fish consumed will affect exposure, because different types of fish and shellfish take up COPCs at different rates. For example, fatty fish tend to accumulate organic COPCs more readily than lean fish. 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. For example, populations such as Indian tribes, American & Pacific Islanders, and some immigrant groups are known to have high local fish consumption rates. Because 100 percent of a receptor's dietary fish may not originate from the surface water body near the combustion facility, the percentage of locally caught fish is also a variable for exposure.

6.3 DERMAL EXPOSURE PATHWAYS

6.3.1 Dermal Exposure to Soil

For the purpose of RCRA combustion permitting decisions, We don't typically recommend evaluating dermal exposure to COPCs through contact with soil. However, site-specific exposure setting characteristics may support evaluating this exposure pathway. Therefore, this section discusses dermal soil exposure.

Available data indicate that the contribution to overall risk from dermal exposure to soils impacted from hazardous waste combustion facilities is typically small relative to contributions resulting from exposures via the food chain (U.S. EPA 1995h; 1996g). For example, the risk assessment conducted for the Waste Technologies Industries, Inc., hazardous waste incinerator in East Liverpool, Ohio, indicated that—for an adult subsistence farmer in a subarea with high exposures—the risk resulting from soil ingestion and dermal contact was 50-fold less than the risk from any other exposure pathway and 300-fold less than the total estimated risk (U.S. EPA 1995h; 1996g).

Humans can be exposed to COPCs by absorption through the skin when it comes into contact with contaminated soil. Factors that affect dermal exposure include:

- exposed skin surface area;
- contact time;

- contact amount;
- amount of time spent near the combustion source; and
- fraction of COPCs absorbed through the skin.

In general, an increased dose of COPCs can potentially be absorbed through the skin as the surface area of the skin is increased. Surface area is affected by age and body weight; for example, children have less total surface area than adults. The amount of surface area available for exposure to soil is also affected by the amount of clothing worn. An adult working in the garden in long sleeves and pants will have a smaller exposed surface than an adult working in shorts and a short-sleeved shirt. For dermal exposure from soil, the exposed surface area affects the amount of soil that can adhere to exposed skin.

Contact time refers to the duration of time each day that skin is in contact with contaminated soil. As duration increases, so does the amount of COPCs that can be absorbed. Dermal exposure is also affected by the amount of time each day spent in the vicinity of the combustion source, where the soil may contain pollutants emitted from the combustion facility. Indoor dust and outdoor soil may both increase the daily contact. Consider seasonal exposure might also be appropriate, because regional climate will influence contact time.

The amount of COPCs that can be absorbed through the skin depends on the chemical properties of the COPC, properties of the soil matrix, and dermal pharmacokinetics. If a COPC can't be readily absorbed through the skin, the daily intake of the COPC may be small even if other exposure characteristics (e.g. contact time) encourage absorption. However, if site-specific conditions suggest that dermal exposure to soil may contribute significantly to total soil-related exposures, we recommend considering the assessment methods described in U.S. EPA (2004d).

6.3.2 Dermal Exposure to Water

We don't typically recommend evaluating the dermal water exposure pathway when assessing risk from hazardous waste combustor emissions. However, if the surface water body affected by combustor emissions is used frequently for recreational purposes such as swimming and boating, dermal absorption of contaminated water becomes another possible route for human exposure. Dermal exposure is affected by:

- the surface area of exposed skin,
- the COPC concentration in the water,
- the permeability of the skin to the COPC, and
- the length of time that the individual is in contact with the water.

6.4 EXPOSURE FREQUENCY

The HHRAP assumes that the receptors in each recommended exposure scenario are exposed to all of the scenario-specific exposure pathways 350 days per year (U.S. EPA 1989e; 1991b; 1991d). This assumption is based on the protective estimate that all receptors spend a maximum of 2 weeks away from the exposure scenario location selected in Section 4.3.

6.5 EXPOSURE DURATION

Exposure duration is the length of time that a receptor is exposed via a specific exposure pathway. A receptor is no longer exposed to COPCs via the direct inhalation exposure pathway after an emission source ceases operation. However, a receptor could be exposed via the indirect exposure pathways for as long as they remain in the assessment area. We recommend using default RME values to estimate exposure duration for specified receptors.

As in U.S. EPA (1998c), we recommend assuming that receptors are exposed to the long-term average COPC soil or water concentrations (and the subsequent COPC plant or animal concentrations) present in the environment or media following a period of time during which there were continuous hazardous waste combustor emissions. For existing facilities, U.S. EPA (1990e) assumed that this period of time can be represented by default time periods of 30, 60, or 100 years. U.S. EPA (1998c) simplified this to assume that the period could be ≥ 30 years. These values are based on the assumptions that the hazardous waste combustion unit or the emission source:

1. is already in place,

2. will continue to be used for the rest of its useful life (estimated to be 30 years), and
3. may be replaced when it reaches the end of its useful life (possibly as long as 60 or 100 years), because it is an integral part of the facility operations.

We consider these assumptions reasonable for a hazardous waste emission source, such as an industrial boiler burning a continuous stream of facility hazardous waste.

Although a combustor may remain in the same location for 100 years—and a person may have a lifetime of exposure to emissions from that combustor— data on population mobility (U.S. Bureau of the Census, 1986) indicate that many Americans do not remain in the same area for their 70-year lifetime. An estimate of the number of years that a person is likely to spend in one area, such as the vicinity of a combustion facility, can be derived from information about mobility rate and median time in a residence. In addition to the number of years at a particular location or residence, the amount of time spent at that location each day directly affects exposure. For example, children that attend day care or adults that work in a different location for part of the day may be exposed to higher or lower COPC levels.

The exposure duration values we recommend are presented in Table 6-3.

TABLE 6-3

EXPOSURE DURATION VALUES

Recommended Exposure Scenario Receptor	Value	Source
Child Resident	6 years	U.S. EPA 1990f; 1994r
Adult Resident	30 years	U.S. EPA 1990f; 1994r
Fisher	30 years	U.S. EPA 1990f; 1994r
Fisher Child	6 years	Assumed to be the same as the Child Resident
Farmer	40 years	U.S. EPA 1994l; 1994r
Farmer Child	6 years	Assumed to be the same as the Child Resident

6.6 AVERAGING TIME

For noncarcinogenic COPCs, we generally recommend using a value of exposure duration (years-as specified for each receptor in Section 6.4) x 365 days/year as the averaging time (U.S. EPA 1989e; 1991d). However, for carcinogenic COPCs—the effects of which may have long latency periods—the age of the receptor (i.e., child, adult, or elderly) influences the COPC exposure pathway, because the exposure duration and, therefore, the quantity of exposure, will vary. For carcinogenic COPCs, we recommend using an averaging time of 70 years.

We recommend evaluating carcinogenic exposures for different receptor ages separately, because the daily activities of these receptors (and body weights, as described in Section 6.6) vary, including:

- the amounts of food and water consumed;
- the types of food consumed; and
- the amount of exposed skin surface.

Health-based criteria, such as health advisories for drinking water, are also different for children and adults. As a result, for some exposure pathways, such as soil ingestion, children may have a greater quantifiable exposure and be at greater risk than adults. Some behaviors, such as mouthing of dirty objects or direct ingestion of soil, which could also contribute to exposure, are also much more prevalent in children than adults.

Because quantifying carcinogenic COPC exposure depends on the duration of exposure, the age of the receptor is important. For risk assessment purposes, the average human lifespan is generally considered to be 70 years. Childhood represents only about 10 percent of the lifespan (6 years) (U.S. EPA 1998c). In actual exposure scenarios, individuals may be exposed only during childhood or adulthood. In other cases, exposure may overlap these periods, such as a child who grows into adulthood and remains in the same geographical area. Based on the age of the receptor and information on the duration of exposure, U.S. EPA (1990e) recommended considering risk to three different receptors:

1. a child who grows to an adult and is exposed for his or her entire 70-year lifetime,

2. a child who grows to an adult and is exposed for only a part of his or her adulthood—a total of 30 years, and

3. an adult exposed for 16 years.

Because the effects of certain carcinogenic COPCs may have long latency periods—in some instances approaching the human lifespan—it may be appropriate to estimate daily intake by using the adult value for body weight and a longer averaging time. In cases where effects have a shorter latency period, U.S. EPA (1990e) recommended an averaging time period of less than 10 years. However, where children are known to be at special risk, it may be more appropriate to use this averaging time with a body weight value for toddlers, infants, or young children. For COPCs classified as carcinogens, we recommend using a longer averaging time and the adult body weight to calculate the risk resulting from air or water exposure.

It is significant that childhood is defined differently in the different references. U.S. EPA (1990e) defines childhood as being from 1 to 7 years old. As in this and other Agency guidance (U.S. EPA 1991b; 1994r; 1998c), we define childhood as having an exposure duration of 6 years. Please note that some of the data used for input into the various exposure scenario equations in Appendix C was not available for children, or was available for more restrictive age groups, such as 2-year-olds or 4- to 6-year-olds. In such cases, and as noted in Appendix C where such values are presented,

1. the available data were evaluated to ensure that the presented default values are sufficient for conducting a risk assessment, and

2. in cases in which the available data were not sufficient, reasonable interpolations of the available data were possible.

RECOMMENDED INFORMATION FOR THE RISK ASSESSMENT REPORT

- Identification of site-specific or alternate default media equations and/or inputs; including justification and full referencing
- Exposure calculations

Chapter 7
Characterizing Risk and Hazard

What's Covered in Chapter 7:

7.1 Quantitatively Estimating Cancer Risk

7.2 Quantitatively Estimating Noncancer Hazard

7.3 Target Levels

7.4 Estimating Acute Exposure from Direct Inhalation

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak in this chapter is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

The final step of a risk assessment is risk characterization. This involves combining the exposure quantities generated in Chapter 6, and the toxicity benchmarks available in the HHRAP companion database, to calculate the excess lifetime cancer risks (risk) and noncancer hazards (hazard) for each of the pathways and receptors identified in Chapter 4. Risks (and hazards) are then summed for each receptor, across all applicable exposure pathways, to obtain an estimate of total individual risk and hazard. Risk characterization also involves documenting the uncertainties and limitations associated with the rick assessment, as described in Chapter 8.

It is important that risk characterization exhibit the core values of transparency, clarity, consistency, and reasonableness (please see the related EPA Information Quality Guidelines recommendations as discussed in Chapter 1, page 1-11).

Risk from exposure to combustor emissions is the probability that a human receptor will develop ***cancer***, based on a unique set of exposure, model, and toxicity assumptions. We recommend using the slope or unit risk factor in risk assessments to estimate the probability of an individual developing cancer as a result of exposure to a particular level of a COPC. For example, a risk of 1×10^{-5} is interpreted to mean that an individual has up to a one in 100,000 chance of developing cancer during their lifetime from the exposure being evaluated. In contrast, ***hazard*** is the potential for developing ***noncancer*** health effects as a result of exposure to COPCs. A hazard is not a probability but, rather, a comparison (calculated as a ratio) of a receptor's potential exposure relative to a standard exposure level (*RfD* or *RfC*). The standard exposure level is calculated over a similar exposure period and is estimated to pose no appreciable likelihood of adverse health effects to potential receptors, including special populations (U.S. EPA 1989e).

Risks and hazards here are typically characterized for single scenarios, and are referred to as individual risks and hazards (U.S. EPA 1989e; 1994g; NC DEHNR 1997). Individual risk and hazard descriptors are intended to convey information about the potential risks to individuals potentially impacted by emissions from a facility burning hazardous waste. A risk assessment developed following the procedures described in Chapters 2 through 8 and Appendixes B and C will provide

- quantitative and qualitative estimates of risk and hazard associated with exposure to COPCs;
- estimates of blood levels associated with exposure to lead;
- evaluation of infant exposure via breast milk to COPCs with appropriate biotransfer factors[1], and
- evaluation of acute risk and hazard resulting from direct inhalation.

If a permitting authority feels that you need to consider calculating population risks, we recommend following the applicable methods described in the U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions* (U.S. EPA 1998c).

Standard rules for rounding apply which will commonly lead to an answer of one significant figure in both risk and hazard estimates. For presentation purposes, hazard quotients (and hazard indices) and

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[1] Currently 2,3,7,8-TCDD TEQ and dioxin-like PCBs are the only COPCs with biotransfer factors for the breastmilk pathway. However, appropriate biotransfer factors for other chemicals may become available and thus provide the information needed to include them in this pathway evaluation. We suggest consulting Chapter 9 (Breastmilk Pathway) of U.S. EPA (1998c).

cancer risk estimates are usually reported as one significant figure. We recommend rounding only the final reported results, not the intermediate calculations.

INFORMATION RECOMMENDED FOR RISK ASSESSMENT REPORT

- Indicate the scope of the risk assessment (match the level of effort to the scope)
- Summarize the major risk conclusions.
- Identify key issues (a key issue is critical to properly evaluate the conclusions). For example, was surrogate or measured emissions data used.
- Describe clearly the methods used to determine risk (provide qualitative narration of the quantitative results).
- Summarize the overall strengths and major uncertainties.

7.1 QUANTITATIVELY ESTIMATING CANCER RISK

As described above, risk estimates represent the incremental probability that an individual will develop cancer over a lifetime as a result of a specific exposure to a carcinogenic chemical (U.S. EPA 1989e). We recommend calculating these risks as follows:

Inhalation Cancer Risk

$$Cancer\ Risk = EC \times URF \qquad \text{Equation 7-1}$$

where
EC = Exposure concentration ($\mu g/m^3$) [see Chapter 6]
URF = Unit risk factor ($(\mu g/m^3)^{-1}$)

Ingestion Cancer Risk

$$Cancer\ Risk = LADD \times CSF \qquad \text{Equation 7-2}$$

where
$LADD$ = Lifetime average daily dose (mg/kg-day)
CSF = Cancer slope factor $(\text{mg/kg-day})^{-1}$

> ***PLEASE NOTE***: In the *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (U.S. EPA 2005g), the Agency recommends estimating inhalation and ingestion cancer risk slightly differently for carcinogens the Agency determines to cause cancer by a mutagenic Mode of Action (MOA, as defined in *Guidelines for Carcinogen Risk Assessment* [U.S. EPA 2005f]). Unfortunately, we haven't completed our recommendations for how to implement the guidelines set out in U.S. EPA (2005f; g). We recommend periodically checking the EPA hazardous waste combustion web site (http://www.epa.gov/epawaste/hazard/tsd/td/combust/index.htm) for updates on our recommendations.

It's possible for receptors to be exposed to multiple COPCs within a individual exposure pathway. We recommend estimating the total risk associated with exposure to all COPCs through a single exposure pathway as follows (U.S. EPA 1989e):

$$Cancer\ Risk_T = \sum_{i} Cancer\ Risk_i \qquad \text{Equation 7-3}$$

where

$Cancer\ Risk_T$ = Total cancer risk for a specific exposure pathway
$Cancer\ Risk_i$ = Cancer risk for COPC i for a specific exposure pathway

Receptors might be exposed through a number of exposure pathways (see Table 4-1). We consider it appropriate to sum risks from multiple exposure pathways for a given receptor. The cumulative risk posed to a receptor is the sum of total risks from each individual exposure pathway. Express the cumulative risk as follows:

$$Cumulative\ Cancer\ Risk = \sum Cancer\ Risk_T \qquad \text{Equation 7-4}$$

where

Cumulative Cancer Risk= Cumulative cancer risk from multiple exposure pathways
$Cancer\ Risk_T$ = Cumulative cancer risk for exposure pathway T

In addition to multiple pathways, a receptor might be exposed to emissions from multiple sources (See Chapter 2 for additional discussion on emission sources). In addition to emission source-specific risk/hazard estimates (see Chapter 3 regarding source-specific modeling), we recommend summing the risks from all modeled sources for each receptor at each exposure scenario location. For example, if a facility operates an incinerator and a boiler that both burn hazardous waste, sum the risks from both units for each receptor. For fugitive emissions from storage and handling of hazardous waste, add the risk associated with fugitive emissions to the risks from the combustion unit for each receptor at each exposure scenario location.

We present the equations we recommend to estimate dose and risk levels in Appendix C. The HHRAP companion database presents inhalation *URF*s and oral *CSF*s for many potential COPCs. However, for each risk assessment, we recommend checking the hierarchy of toxicity benchmark and slope factor resources listed in Appendix A-2, Section A2.6 (Human Health Benchmarks) for updated values. We suggest using the same hierarchy to acquire toxicity values for COPCs not identified in Appendix A-2.

In the assessment of carcinogenic risk from COPCs, we recommend U.S. EPA-derived or reviewed health benchmarks (*URFs* and *CSFs*). However, for numerous compounds, a complete set of inhalation and oral EPA-derived health benchmarks are not available. In such cases, we calculated the health benchmarks presented in the companion database based on available U.S. EPA-derived benchmark values.

If relevant information is not available from these sources, we recommend contacting the appropriate permitting authority, which may be able to assist in developing the necessary toxicity values. For example, Minimum Risk Levels published by the Agency for Toxic Substances and Disease Registry (ATSDR) might be applicable.

7.2 QUANTITATIVELY ESTIMATING NONCANCER HAZARD

Standard risk assessment models assume that, for most chemicals with noncancer effects, the noncancer effects exhibit a threshold response[2],. That is, there is a level of exposure below which no adverse effects will be observed (U.S. EPA 1989e). The default approaches used by USEPA to assess the potential for health effects associated with a nonlinear or threshold relationship with exposure as set out in U.S. EPA (2002; 2005f) involve:

1. Comparing an estimate of ingested exposure (see Chapter 6) to an *RfD* for oral exposures; and

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[2] Some chemicals don't demonstrate a threshold response. Lead and ozone are two examples of chemicals with noncancer effects that don't have a threshold below which no adverse effects are observed.

2. Comparing an estimated chemical-specific air concentration to the *RfC* for direct inhalation exposures.

An *RfD* is a daily oral intake rate that is estimated to pose no appreciable risk of adverse health effects, even to sensitive populations, over a 70-year lifetime. Similarly, an *RfC* is an estimated daily concentration of a chemical in air, the exposure to which over a specific exposure duration poses no appreciable risk of adverse health effects, even to sensitive populations (U.S. EPA 2002).

The exposure durations assumed for the exposure pathways identified in Table 4-1 range from subchronic to chronic in relative length. However, we consider it appropriate to use chronic *RfDs* and *RfCs* to evaluate all recommended exposure pathways. The comparisons of oral and inhalation exposure estimates to *RfD* and *RfC* values, described above, are known as hazard quotients (*HQ*), which are calculated as follows:

$$HQ = \frac{ADD}{RfD} \text{ or } HQ = \frac{EC}{RfC} \qquad \text{Equation 7-5}$$

where
HQ = Hazard quotient (unitless)
ADD = Average daily dose (mg/kg-day)
RfD = Reference dose (mg/kg-day)
EC = Exposure air concentration (mg/m^3)
RfC = Reference concentration (mg/m^3)

Please note that each program office within U.S. EPA determines what *HQ* level poses a concern to exposed individuals. For example, Superfund has determined that an *HQ* of less than or equal to 1 is considered health-protective (U.S. EPA 1989e). However, because *RfDs* and *RfCs* do not have equal accuracy or precision, and are not based on the same severity of effect, the level of concern does not increase linearly as an *HQ* approaches and exceeds 1 (U.S. EPA 1989e). In addition, noncancer estimates only identify the exposure level below which adverse effects are unlikely; an RfD or RfC does not say anything about incremental risk for higher exposures (U.S. EPA 1998c).

Also note that background exposures may be an important consideration in setting *HQ* levels of concern. This is because you generally model noncancer effects as thresholds, and biologic systems (including human receptors) do not distinguish between exposures from regulated versus non-regulated sources. In certain cases, a permitting authority may elect to adjust the assessed facility-specific *HQ* downward, to account for any exposure that individuals may have from non-assessed sources.

As with carcinogenic chemicals, a receptor might be exposed to multiple chemicals associated with noncancer health effects. We recommend calculating the total chronic hazard for each exposure pathway by following the procedures outlined in U.S. EPA (1986e; 1989e; and 2000e). Specifically, the total chronic hazard attributable to exposure to all COPCs through a single exposure pathway is known as a hazard index (*HI*). The *HI* is calculated as follows:

$$HI = \Sigma_{i} \, HQ_{i} \qquad \text{Equation 7-6}$$

where

HI = Hazard index for a specific exposure pathway
HQ_i = Hazard quotient for COPC i

This method assumes that the health effects of the various COPCs are additive. This method is a simplification of the *HI* concept because it doesn't, at this stage, directly consider the portal of entry associated with each exposure pathway (i.e. inhalation, or ingestion). This method also doesn't consider the often unique toxic endpoints and toxicity mechanisms of the various COPCs.

As discussed in Section 7.1 for carcinogenic risks, a receptor might be exposed to COPCs associated with noncancer health effects through more than one exposure pathway, and from multiple emissions sources. We recommend estimating the noncancer hazards from each modeled source (including fugitive emissions) separately, as well as all sources summed for each receptor. We consider it reasonable to estimate a receptor's total hazard as the sum of the *HI*s for each of the exposure pathways chosen for the receptor. Specifically, a receptor's cumulative hazard is the sum of hazards from each individual exposure pathway, expressed as follows:

$$Cumulative\ HI = \Sigma \ HI \qquad \text{Equation 7-4}$$

where

$Cumulative\ HI$ = Cumulative hazard index from all scenario-specific exposure pathways
HI = Hazard index for a specific exposure pathway

As in U.S. EPA (1989e), we recommend further evaluating a cumulative *HI* which exceeds the target hazard level. A cumulative *HI* can exceed the target hazard level due to either

- One or more COPCs with an *HQ* exceeding the target hazard level, or
- The summation of several COPC-specific *HQ*s that are each less than the target hazard level.

In the former case, you can interpret the presence of at least one COPC-specific hazard greater than the target hazard level as indicating the potential for noncancer health effects. In the latter case, you need to perform a detailed analysis to determine whether the potential for noncancer health effects is accurately estimated by the total *HI*. This is because the toxicological effects associated with exposure to multiple chemicals, often through different exposure pathways, may not be additive. The total *HI* might therefore overestimate the potential for noncancer health effects.

To address this issue, we recommend summing the COPC-specific hazards according to toxicological similarity (e.g. the same target organs or systems) (U.S. EPA 2000e). This process is referred to as segregating the *HI*. It is especially important to consider any differences related to exposure route. If any segregated *HI* exceeds the target hazard level, noncancer health effects cannot be ruled out. However, if all segregated *HI*s are less than the target hazard level, noncancer health effects are not likely to result from exposure to the COPCs included in the *HI*.

Summing all *HI*s
As stated above, estimating a single *HI* encompassing all *HI*s across all exposure pathways is a simplification of the *HI* concept. However, it may save valuable resources: if the single *HI* is not above the target hazard level, then no further segregation would be necessary. We recommend this as a first step, and going to the expense of segregating *HI*s only if the single *HI* falls above the target hazard level.

Technically, segregating the *HI* based only on target organs or systems is a simplification of *HI*. Ideally, the *HI* would also be segregated according to the often unique mechanisms of toxicity of the COPCs. However, segregating the *HI* based on mechanisms of toxicity is beyond a screening level or initial risk evaluation approach (U.S. EPA 2000e).

The HHRAP companion database includes information on target organs and systems that are affected by each COPC. The database also presents *RfDs* and *RfCs* for these same COPCs. If you include COPCs not identified in Appendix A-2 (and therefore not in the companion database) in the risk assessment, we recommend obtaining *RfDs* and *RfCs* for these compounds using the hierarchy of toxicity benchmark and slope factor resources listed in Appendix A-2, Section A2.6 (Human Health Benchmarks). If relevant information is not available from these sources, we recommend working with the permitting authority to 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.

In the assessment of noncancer risk from COPCs, we recommend U.S. EPA-derived or reviewed *RfD*s and *RfC*s. However, for numerous compounds, a complete set of inhalation and oral health benchmarks is not available. If such was the case for COPCs listed in Appendix A-2, we calculated the health benchmarks presented based on available U.S. EPA-derived benchmarks values. For instance, if the *oral RfD* (mg/kg/day) was available and the *RfC* (mg/m^3) was not, we calculated the *RfC* by multiplying the *RfD* by an average human inhalation rate of 20 m^3/day and dividing by the average human body weight of 70 kg. This conversion is called a route-to-route extrapolation, which assumes that the toxicity of the given compound is equivalent over all routes of exposure.

Route-to-route extrapolation introduces additional uncertainty into the risk assessment, and there isn't Agency consensus regarding the appropriateness of its use. This method assumes that the qualitative data supporting the benchmark value for a certain route also applies to the route in question. For example, if an *RfD* is available and you calculate the *RfC* from that value, you are assuming that the toxicity seen following oral exposure will be equivalent to toxicity following inhalation exposure. This assumption could overestimate or underestimate the toxicity of the given compound following inhalation exposure.

Because of the degree of uncertainty involved in using toxicity benchmark values calculated based on route-to-route extrapolation, we recommend using route-to-route extrapolations for organic compounds (but not inorganic), and revisiting the appropriateness of applying this extrapolation for individual chemicals if they are found to be risk drivers. An example might include using route-to-route extrapolations as the first step in a screening risk assessment, then expending resources evaluating the appropriateness of only those extrapolations associated with risk drivers. Including this further evaluation (a qualitative assessment of the toxicity information available for the compound and exposure route) in the Uncertainty section of the risk assessment report will enable the risk manager to make an informed decision concerning the validity of values calculated based on route-to-route extrapolation.

7.3 TARGET LEVELS

Target levels are risk management-based and set by the permitting authority. Target values are not a discrete indicator of observed adverse effect. If a risk estimate falls below target levels, a regulatory authority may, without further investigation, conclude that a proposed action does not present an unacceptable risk. A risk estimate that exceeds these targets, however, would not, in and of itself, necessarily indicate that the proposed action is not safe or that it presents an unacceptable risk. Rather, a risk estimate that exceeds a target value triggers further careful consideration of the underlying scientific basis for the calculation.

7.4 ESTIMATING ACUTE EXPOSURE FROM DIRECT INHALATION

In addition to long-term chronic effects, we recommend considering short-term or acute effects from direct inhalation of vapor phase and particle phase COPCs. Short-term emissions don't typically have a significant impact through the indirect exposure pathways (as compared to impacts from long-term emissions). Therefore, we recommend evaluating acute effects only through the short-term (maximum 1-hour) inhalation of vapors and particulates exposure pathway of the acute risk scenario. We give our recommendations for where and when to evaluate the acute risk scenario in Sections 4.2 and 4.3.
In order to establish acute inhalation exposure criteria (AIEC), we needed to identify and evaluate

1. Existing guidelines for acute inhalation exposure; and
2. Existing approaches for developing acute inhalation exposure levels.

Existing approaches are composed of hierarchical guidelines for acute inhalation exposure, ranked in order of applicability and technical basis, and all being protective of the general public.

> ***Please Note***: hierarchical approaches are needed because criteria values are COPC-specific, and no single organization or method has developed acute criteria values or benchmarks for all of the potential COPCs.

7.4.1 Existing Hierarchical Approaches for Acute Inhalation Exposure

Existing guidelines or criteria for evaluating acute inhalation exposure have been or are being developed by several organizations in the United States, including:

- Agency for Toxic Substances and Disease Registry (ATSDR 1997);
- American Conference of Governmental Industrial Hygienists (ACGIH 1996);
- American Industrial Hygiene Association (AIHA 1997);
- California Environmental Protection Agency (Cal/EPA) (Cal/EPA 1999);
- National Advisory committee (NAC 1997); and
- National Institute of Occupational Safety and Health (NIOSH 1994);
- National Research Council Committee on Toxicology (NRC COT 1986; U.S. EPA 1987b);
- Occupational Safety and Health Administration (NIOSH 1994);
- U.S. Department of Energy, Subcommittee on Consequence Assessment and Protective Actions (SCAPA) (SCAPA 2001a; 2001b).
- U.S. EPA (U.S. EPA 1987b);

Acute inhalation exposure guidelines and criteria are

- Designed to protect a variety of exposure groups, including occupational workers, military personnel, and the general public,
- Based on varying exposure durations up to 24 hours in length, and
- Intended to protect against a variety of toxicity endpoints ranging from discomfort or mild adverse health effects to serious, debilitating, and potentially life-threatening effects, up to and including death.

Hierarchical approaches for establishing acute inhalation exposure levels protective of the general public have been developed by a variety of organizations and teams of organizations. These organizations include:

C U.S. Department of Defense (DoD 1996);

C U.S. Department of Energy (DoE) (SCAPA 1997a; WSRC 1998).

C U.S. EPA Region 3 (EPA 1996b);

C U.S. EPA Region 10 (U.S. EPA 1996a); and

- Federal Emergency Management Agency, Department of Transportation (DoT), and U.S. EPA (U.S. EPA 1993i).

The acute inhalation exposure guidelines developed by these organizations are generally quite heterogenous, developed to protect different subpopulations against different effects and apply to various exposure durations. All the hierarchical approaches listed above except the SCAPA approach needed to adjust the existing guidelines using safety factors (usually multiples of 10) to account for differences in exposure group, exposure duration, and toxicity endpoint, to arrive at acute inhalation exposure values applicable to the general public.

In contrast to the hierarchical approaches developed using safety factors, the DoE's Emergency Management Advisory Committee's SCAPA developed temporary emergency exposure limits (TEELs) based on tiered, formula-like statistical analyses between existing guidelines for acute inhalation exposure and AIHA emergency response planning guidelines (ERPG) (Craig et al. 1995; WSRC 1998). The methodology is described at http://www.orau.gov/emi/scapa/files/Method_for_deriving_TEELs.pdf and available on-line at http://www.atlintl.com/DOE/teels/teel/teel_pdf.html. Like ERPGs, TEELs are multiple-tiered, representing concentrations associated with no effects (TEEL-0), mild, transient effects (TEEL-1), irreversible or serious effects (TEEL-2), and potentially life-threatening (TEEL-3). DOE developed TEELs for situations where no other value is available. TEELs do not undergo peer review. For compounds for which TEEL values could not be developed using this approach, SCAPA developed a supplementary approach using available toxicity information, primarily (1) lethal dose and concentration median, and (2) lethal dose and concentration low values (DoE 1997a).

7.4.2 Our Recommended Hierarchical Approach

After reviewing the existing hierarchical approaches, we recommend the following approach. Because of the daily operations of most combustion units and the potential for upset conditions to sometimes occur during operations, we consider acute values that address intermittent exposures more appropriate and more protective than values that are based on the assumption that acute exposures will be one-time only. When available, we recommend using values from all of the sources that are based on one-hour exposures.

1. **Cal/EPA Acute RELs** – the concentration in air at or below which no adverse health effects are anticipated in the general population, including sensitive individuals, for a specified exposure period (Cal/EPA 1999)
(On-Line Address – http://www.oehha.ca.gov/air/pdf/acuterel.pdf)

2. **Acute inhalation exposure guidelines (AEGL-1)** – "the airborne concentration of a substance above which it is predicted that the general population, including susceptible individuals, could experience notable discomfort, irritation, or certain asymptomatic nonsensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure." (NOAA 2001; U.S. EPA 2001a) (On-Line Address – http://www.epa.gov/oppt/aegl/)

3. **Level 1 emergency planning guidelines (ERPG-1)** – "the maximum concentration in air below which it is believed nearly all individuals could be exposed for up to one hour without experiencing other than mild transient adverse health effects or perceiving a clearly defined objectionable odor." (DoE 2001; SCAPA 2001b)
(On-Line Address – http://www.bnl.gov/emergencyservices/)

4. **Temporary emergency exposure limits (TEEL-1)** – "the maximum concentration in air below which it is believed nearly all individuals could be exposed without experiencing other than mild transient adverse health effects or perceiving a clearly defined odor." (DoE 2001; SCAPA 2001a) (On-Line Address –

   http://orise.orau.gov/emi/scapa/files/Method_for_deriving_TEELs.pdf)

5. **AEGL-2 values** – "the airborne concentration of a substance above which it is predicted that the general population, including susceptible individuals, could experience irreversible or other serious, long-lasting adverse health effects or an impaired ability to escape." AEGL-2 values are to be used only if lower ERPG-1 or TEEL-1 values are not available. (NOAA 2001; U.S. EPA 2001a) (On-Line Address – http://www.epa.gov/oppt/aegl/)

The hierarchy is presented in order of preference, from 1 (most preferred) to 5 (least preferred). We generally recommend the Acute Reference Exposure Levels (Acute RELs) developed by Cal/EPA (Cal/EPA 1999) as the first choice for acute inhalation values. If no acute REL value is available for a given COPC, you can work down the list in order. If no AEGL-1 value is available, but an AEGL-2 value is available, select the AEGL-2 as the AIEC only if it's a more protective value (lower in concentration) than an ERPG-1, or a TEEL-1 value if either of these values is available. If no acute values are available for a COPC, an acute value can be developed following the toxicity-based approach used by SCAPA (Tier 5) (DoE 1997a; WSRC 1998). The companion database provides a listing of AIECs compiled from values currently available following the hierarchical approach presented above.

Please note that the TEEL-1 values (SCAPA 2001a) are calculated assuming a 15-minute exposure period. As discussed in Section 3.10, for the purposes of this protocol, we recommend evaluating risks due to acute exposure based on the highest 1-hour average air concentrations. Therefore, the TEEL-1 values were extrapolated from a 15-minute to a 1-hour exposure basis using a modification to Haber's Rule developed by ten Berge et al (1986) and used by Cal/EPA to develop acute RELs (Cal/EPA 1999), as shown below.

$$C^n \cdot T = K \qquad \text{Equation 7-8}$$

where

C = Concentration (mg/m^3)
n = Constant greater than zero (unitless)
T = Time of exposure (hour)
K = Constant level or severity of response (unitless)

Where available, chemical-specific values for the exponent *n* were used to make the extrapolations (Cal/EPA 1999). For chemicals for which a chemical-specific value of *n* was not available, extrapolations were made using a value of n = 1, as recommended by OEHHA, because the extrapolations were all based on an initial exposure period (15-minutes) of less than 1 hour duration (Cal/EPA 1999).

Using the modified form of Haber's Rule allows you to consider contributions by both concentration and time to the overall severity of effect. However, we highly recommend taking special care interpreting the extrapolated air concentrations, as they aren't absolutes. For example, chemical-specific values of the exponent *n* are sometimes based on a relatively limited set of dose-response data. Also, the majority of extrapolated TEEL-1 values were calculated using default exponent values and, therefore, are likely to be even less certain than exponent values based on limited data sets.

The EPA IRIS program is currently developing additional acute reference values that do not exclude intermittent exposures. When available, we recommend using those values (referred to as Acute Reference Concentrations [Acute RfCs]) as the first choice, with the Cal/EPA acute RELs second in the hierarchy.

7.4.3 Characterizing Potential Health Effects from Acute Exposure

We recommend characterizing the potential for adverse health effects from acute exposure to COPC-specific emissions by comparing the acute air concentration (C_{acute}) resulting from maximum emissions over a 1-hour period to the COPC-specific AIEC (see Appendix C, Table C-4-1). This comparison is known as the acute hazard quotient (AHQ_{inh}). Chapter 3 discusses air dispersion modeling related to obtain 1-hour maximum values. Appendix B, Table B-6-1 describes how to calculate C_{acute}. We recommend using Equation 7-9 to calculate the AHQ_{inh}:

$$AHQ_{inh} = \frac{C_{acute} \cdot 0.001}{AIEC} \qquad \text{Equation 7-9}$$

where

AHQ_{inh} = Acute hazard quotient (unitless)
C_{acute} = Acute air concentration (:g/m^3)
0.001 = Conversion factor (mg/:g)
$AIEC$ = Acute inhalation exposure criteria (mg/m^3)

We recommend calculating acute hazard quotients at the selected acute exposure scenario locations (see Sections 4.2 and 4.3) for COPCs specific to emissions from each source, and from all facility sources combined. We recommend summing acute hazard quotients from individual chemicals (e.g. acid gases), if they have similar effects. Setting target levels for evaluating acute hazard quotients is a risk management decision made by the permitting authority.

Chapter 8
Interpreting Uncertainty for Human Health Risk Assessment

What's Covered in Chapter 8:

8.1 Uncertainty and Limitations of the Risk Assessment Process

8.2 Types of Uncertainty

8.3 Qualitative Estimates of Uncertainty

8.4 Quantitative Estimates of Uncertainty

8.5 Risk Assessment Uncertainty Discussion

This section discusses interpreting uncertainties associated with the risk assessment. The discussion of uncertainties in Section 8.1 and 8.2 was adopted from the *1996 Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Waste*.

> ***PLEASE NOTE***: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.
>
> The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak in this chapter is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

8.1 UNCERTAINTY AND LIMITATIONS OF THE RISK ASSESSMENT PROCESS

Uncertainty can be introduced into a health risk assessment at every step of the process outlined in this document. Uncertainty occurs because risk assessment is a complex process, requiring the integration of the following:

- Release of pollutants into the environment
- Fate and transport of pollutants, in a variety of different and variable environments, by processes that are often poorly understood or too complex to quantify accurately

- Potential for adverse health effects in humans, as extrapolated from animal studies
- Probability of adverse effects in a human population that is highly variable genetically, and in age, activity level, and lifestyle

Uncertainty is inherent in the process even when using the most accurate data and the most sophisticated models. The method we recommend in the HHRAP relies on a combination of point values—some protective and some typical—yielding a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk. For this reason, the degree of conservatism in risk estimates cannot be known. Therefore, you need a formal uncertainty analysis to determine the degree of conservatism. Section 8.2 discusses the types of uncertainty and the areas in which uncertainty can be introduced into an assessment. The remaining Sections discuss methods for qualitatively and quantitatively addressing uncertainty in risk assessments.

It should also be noted, variability is often used interchangeably with the term “uncertainty,” but this is not strictly correct. “Variability” may be tied to variations in physical and biological processes. Variability can’t be reduced with additional research or information, although 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 a group.

In general, uncertainty is reducible by additional information-gathering or analysis activities (that is, better data or better models), whereas real variability won’t change (although it may be more accurately known) as a result of better or more extensive measurements (Hattis and Burmaster 1994).

8.2 TYPES OF UNCERTAINTY

Finkel (1990) and U.S. EPA (1999f) classified all uncertainty into four types:

1. variable uncertainty,
2. model uncertainty,
3. decision-rule uncertainty, and
4. variability.

Variable uncertainty and model uncertainty are generally recognized by risk assessors as major sources of uncertainty. Variable uncertainty occurs when variables appearing 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 variable uncertainty that are especially critical for small sample sizes. It is more difficult to recognize nonrandom, or systematic, errors that result from the basis for sampling, experimental design, or choice of assumptions.

Model uncertainty is associated with all models used in all phases of a risk assessment, including:

- animal models used as surrogates for testing human carcinogenicity,
- the dose-response models used in extrapolations, and
- the computer models used to predict the fate and transport of chemicals in the environment.

Using rodents as surrogates for humans introduces uncertainty into the risk factor because of the considerable interspecies variability in sensitivity. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models because of (1) increased complexity, or (2) a lack of data for these variables. The importance of excluded variables is generally considered on a case-by-case basis. A specific variable may be important, in terms of its impacts on uncertainty, in some instances and not in others. A similar problem can occur when a model that is applicable under average conditions is used for a case in which conditions differ from the average. Finally, choosing the correct model form is often difficult, because conflicting theories appear to explain a phenomenon equally well.

The models we recommend in the HHRAP were selected based on scientific policy. We selected the air dispersion and deposition model, and the indirect exposure models, because they provide the information you need to conduct indirect assessments, and we consider them state-of-the-science. ISCST3—the air dispersion model we recommend—has not been widely applied in its present form. Few data are available on atmospheric deposition rates for chemicals other than criteria pollutants, thereby making it difficult to select input variables related to deposition, or validate modeled deposition rates. Long-range transport of pollutants into and out of the study areas wasn't modeled, resulting in an underestimation of risk attributable to each facility.

In addition to air dispersion modeling, using other fate and transport models recommended by this guidance can also result in some uncertainty. For example, the models which estimate COPC concentrations in waterbodies may be particularly protective for waterbodies located in estuarine environments with tidal influence. Because tidal influence is not considered in the models presented in Chapter 5, the resulting dilution of COPC concentrations in water and sediments likely caused by tidal influence won't be considered in the risk assessment. Thus, the risk assessment results will likely be more protective for tidally influenced waterbodies than for those waterbodies that aren't tidally influenced. We recommend that permitting decisions based on risk estimates for estuarine environments consider this uncertainty. The delineation of this uncertainty may be one area that could be addressed in a more refined site-specific risk assessment, if warranted.

Decision-rule uncertainty 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. The uncertainty associated with risk analysis influences many policy and risk management decisions. Possibly the most important aspect for the risk estimates is the selection of constituents to be assessed. Constituents that are identified using the process provided in this document will include compounds that have the potential to pose the greatest risk to human health through indirect exposure routes. For example, many PICs are highly lipophilic and tend to bioaccumulate in the food chain, thereby presenting a potentially high risk through the consumption of contaminated food.

A second area of decision-rule uncertainty includes the use of standard Agency default values in the analysis. These include inhalation rates, body weight, and lifespan, which are standard default values used in most Agency risk assessments. Inhalation rate is highly correlated to body weight for adults. Using a single point estimate for these variables instead of a joint probability distribution ignores a variability that may influence the results by a factor of up to two or three.

A third area of decision-rule uncertainty is the use of Agency-verified cancer *SFs*, *RfDs*, and *RfCs*. These health benchmarks are used as single-point estimates throughout the analysis, and uncertainty and variability are both associated with them. U.S. EPA has developed, however, a process for setting verified health benchmark values to be used in all Agency risk assessments. This process is used to account for much of the uncertainty and variability associated with the health benchmarks. With the exception of the dioxin toxicity equivalency methodology, health benchmarks which can be found on

IRIS, have been verified through Agency work groups. Estimating the uncertainty in using Agnecy-verified health benchmarks or the dioxin toxicity equivalency methodology is beyond the scope of the HHRAP.

8.3 QUALITATIVE ESTIMATES OF UNCERTAINTY

Often, sources of uncertainty in a risk assessment can be identified but not quantified. For example, this can occur when you know (or suspect) a factor to vary, but have no data (e.g., presence of COPCs without toxicity data, amount of time that people at a specific site spend outdoors). In such cases, default data may be available that can be useful in estimating a possible range of values. Uncertainty also often arises out of a complete lack of data. A process may be so poorly understood that you can't quantify the uncertainty with any confidence. In addition, some sources of uncertainty (such as uncertainty in theories used to deduce models) are inherent qualifications reflecting subjective modes of confidence rather than probabilistic arguments. When you can only present the uncertainty qualitatively, you might consider the possible direction and orders of magnitude of the potential error.

8.4 QUANTITATIVE ESTIMATES OF UNCERTAINTY

It's also possible to use knowledge of experimental or measurement errors to introduce a degree of quantitative information into a qualitative presentation of uncertainty. For example, standard laboratory procedures or field sampling methods may have a known error level that you can use to quantify uncertainty. In many cases, it's possible to express the uncertainty associated with particular variable values or estimated risks quantitatively, and further evaluate them with variations of sensitivity analyses. Finkel (1990) identified a six-step process for producing a quantitative uncertainty estimate:

1. Define the measure of risk (such as deaths, life-years lost, maximum individual risk (MIR), or population above an "unacceptable" level of risk). More than one measure of risk may result from a particular risk assessment: however, the uncertainty may be quantified or reached individually.

2. Specify "risk equations" that present mathematical relationships that express the risk measure in terms of its components. This step is used to identify the important variables in the risk estimation process.

3. Generate an uncertainty distribution for each variable or equation component. These uncertainty distributions may be generated by using analogy, statistical inference techniques, expert opinion, or a combination of these.

4. Combine the individual distributions into a composite uncertainty distribution.

5. Recalibrate the uncertainty distributions. Inferential analysis could be used to "tighten" or "broaden" particular distributions to account for dependencies among the variables and to truncate the distributions to exclude extreme values.

6. Summarize the output clearly, highlighting the important risk management implications. Address specific critical factors.
   - Implication of supporting a point estimate produced without considering uncertainty

   - Balance of the costs of under- or over-estimating risks

   - Unresolved scientific controversies, and their implications for research

When you need a detailed quantitative treatment of uncertainty, statistical methods are generally considered the most appropriate. We describe two possible approaches to a statistical treatment of uncertainty with regard to variable values here, though other methods are certainly available. The methods described here were used in this analysis where appropriate.

The first approach is to use an appropriate statistic to express all variables for which uncertainty is a major concern. For example, if a value comes from a sample (such as yearly emissions from a stack), the mean and standard deviation may both be presented. If the sample size is very small, it may be appropriate to

- give the range of sample values and use a midpoint as a best estimate in the model, or

- use the smallest and largest measured value to obtain two estimates that bound the expected true value.

Selecting the appropriate statistic depends on the amount of data available and the degree of detail you need. It's possble to propagate uncertainties by using analytical or numerical methods.

A second approach is to use the probability distributions of major variables to propagate variable value uncertainties through the equations you use in the risk assessment. You then develop a probability distribution of expected values for each variable value. These probability distributions are typically expressed as either probability density functions (*PDF*) or cumulative probability density functions (*CPF*). The *PDF* presents the relative probability for discrete variable values, whereas the *CPF* presents the cumulative probability that a value is less than or equal to a specific value.

A composite uncertainty distribution is created by combining the individual distributions with the equations used to calculate the probability of particular adverse health effects and points. In Monte Carlo simulations, a computer program is used to repeatedly solve the model equations, under different selections of variable values, to calculate a distribution of exposure (or risk) values. Each time the equations are calculated, values are randomly sampled from the specified distributions for each variable. The end result is a distribution of exposure (or risk). These can again be expressed as *PDF*s or, more appropriately, as *CPF*s. The distribution enables you to choose the value corresponding to the appropriate percentile in the overall distribution. For example, you could select an exposure level or risk level that corresponds to the 95th percentile of the overall risk distribution rather than a point estimate of risk that is based on the 95th percentile values for each variable. For more information on how to conduct a quantitative uncertainty analysis, we refer you to Risk Assessment Guidance for Superfund, Volume III: Part A (Process for Conducting Probabilistic Risk Assessment), which is located at the web address www.epa.gov/oswer/riskassessment/rags3a/index.htm.

8.5 RISK ASSESSMENT UNCERTAINTY DISCUSSION

The science of risk assessment is evolving. Where the science-base is incomplete and uncertainties exist, science policy assumptions must me made. It is important for risk assessments to fully explain the areas of uncertainty in the assessments and to identify the key assumptions used in conducting the assessments. Toward that end, one option is to add a table at the end of each section (e.g., stack emissions, air modeling, exposure assessment, toxicity evaluation, risk characterization) that lists the key assumptions in that section, the rationale for those assumptions, their effect on estimates of risk (overestimation, underestimation, neutral), and the magnitude of the effect (high, medium, low). For example, it could explain that using a particular input variable, such as exit gas temperature, will under- or overestimate chronic emissions, and the resulting risks and hazards, by a factor of (x). These tables could be used to evaluate the extent to which you used public health-protective assumptions in the risk assessment. They could also help determine the nature of the uncertainty analysis to be performed. The assumptions listed in the risk characterization section, which synthesizes the data outputs from the exposure and toxicity analyses, might include the most significant assumptions from each of the previous sections.

Within the HHRAP, we've identified uncertainties and limitations within the discussion of specific technical issues (e.g., TOE, estimates of emission rates, COPC selection process, quantifying non-detects) as they are presented in their respective sections. We present the limitations associated with parameter values and inputs to equations in Appendices A, B, and C, respectively.

As an example discussion, the following summarizes some of the uncertainty involved in the air dispersion modeling component of the risk assessment process.

Although dispersion modeling is a valuable tool for estimating concentration and deposition impacts, it has many limitations. The accuracy of the models is limited by

- the ability of the model algorithms to depict atmospheric transport and dispersion of contaminants, and
- the accuracy and validity of the input data.

For example, most refined models require input of representative meteorological data from a single measuring station. In reality, a release will encounter highly variable meteorological conditions that are constantly changing as it moves downwind. U.S. EPA's *Guideline on Air Quality Models—Revised* (U.S. EPA 1986b, 1993b; Title 51 CFR Appendix W) describes two types of model uncertainty. Inherent uncertainty involves deviations in concentrations that occur even if all of the model input is accurate. Reducible uncertainty is associated with the model and the uncertain input values that will affect the results. Although it is important to accurately represent actual conditions by selecting the right model, and using accurate and representative input data, all model results are subject to uncertainty. Nevertheless, models are generally considered reasonably reliable in estimating the magnitude of highest concentrations resulting from a release, although they may not necessarily be time-and space-specific (Title 51 CFR Appendix W). When applied properly, air dispersion models are typically accurate to ± 10 to 40 percent and can be used to yield a "best estimate" of air concentrations (Title 51 CFR Appendix W).

As mentioned earlier, uncertainties specific to other technical components of the risk assessment process (e.g., TOE, quantification of non-detects) are further described in their respective chapters or sections of this guidance. For more information on Agency policy and guidelines regarding uncertainty in risk assessments, please see the U.S. EPA Science Policy Council's *Risk Characterization Handbook* (U.S. EPA 2000f, available at http://www.epa.gov/osa/spc/pdfs/rchandbk.pdf).

Chapter 9
Completing the Risk Assessment Report and Follow-On Activities

What's Covered in Chapter 9:

9.1 Conclusions

9.2 Activities Following Risk Assessment Completion

Our main purpose in developing the HHRAP was to provide you with the tools you need to efficiently complete quality, consistent, and defensible risk assessments. You can produce the risk assessment in a relatively short amount of time, rather than spending years determining which COPCs, exposure pathways, and receptors to include and evaluate.

It's important to note that final risk assessments might include both human health and ecological evaluations. In addition to available Agency guidance for conducting ecological risk assessments (U.S. EPA 1997e) and Volume 63, Number 93, of the Federal Register, we are currently finalizing an ecological risk assessment guidance document titled *U.S. EPA OSW Screening Level Ecological Risk Assessment Protocol* (U.S. EPA 1999a), prepared as a companion to this guidance.

PLEASE NOTE: for the purposes of this guidance, "we" refers to the U.S. EPA OSW.

The HHRAP is written for the benefit of a varied audience, including risk assessors, regulators, risk managers, and community relations personnel. However, the "you" to which we speak in this chapter is the performer of a risk assessment: the person (or persons) who will actually put the recommended methods into practice.

9.1 CONCLUSIONS

We recommend that each risk assessment include a Conclusions section. This section is included primarily to interpret the results of the risk and hazard characterization in light of the uncertainty analysis. We recommend that, at a minimum, it present and interpret all risk and hazard results exceeding target levels. It might also identify receptors having the greatest risks and hazards, in addition to COPCs and exposure pathways contributing significantly to these risks and hazards. Finally, the Conclusions section is a place for you to present and defend your position on whether actual or potential releases from the facility you studied pose significant risks and hazards to human populations.

9.2 ACTIVITIES FOLLOWING RISK ASSESSMENT COMPLETION

As stated previously, we developed the HHRAP to promote a consistent approach for completing risk assessments that:

- Encourages the use of appropriate site-specific information early in and throughout the process;
- Provides a tool for completing quality and defensible risk assessments;
- Minimizes inefficient expenditure of time and resources by suggesting up front a process to determine which COPCs to consider, which exposure pathways, and receptors the risk assessment report to include and evaluate, and
- Provides a comprehensive explanation of the procedures and uncertainties involved in the process.

However, the risk assessment process does not end following the completion, submittal, and approval of the risk assessment report.

Facilities burning hazardous waste are also responsible for communicating the results of the risk assessment process to affected members of the community. One reason for our comprehensive explanation of the procedures and uncertainties involved in the process was to provide the facilities, risk assessors, and regulators with the tools they need to clearly communicate the procedures, results, and limitations of the risk assessment process. Communication is an ongoing process.

Finally, completing the risk assessment process involves using

- site-specific environmental data,
- various assumptions, and
- an evolving procedure for estimating risk.

We expect that facilities will periodically review each of these factors, and update the process with the latest facility-specific operating and emission information. Updating will help determine whether the best data and procedures are used to estimate the risk resulting from operating the facility hazardous waste combustor. The permitting authority might establish the period for this review in the operating permit. However, we recommend reviewing any significant changes involving newly available data or risk assessment procedures as they become available, to gauge if they significantly affect the outcome of the risk assessment process.

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APPENDIX A-1

CHEMICALS OF POTENTIAL INTEREST

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APPENDIX A-1

CHEMICALS OF POTENTIAL INTEREST

Table A-1 presents a comprehensive list of compounds typically found in

- hazardous waste; and
- hazardous waste combustion stack gas emissions.

Table A-1 identifies the Chemical Abstracts Service (CAS) number for each compound, and indicates whether a compound has been identified as a potential COPC by:

- U.S. EPA and state risk assessment reference documents;
- Emission test results that have identified the compound in the emissions from hazardous waste combustion facilities; or
- Other literature that suggests that the risks from the compound may be significant.

We provide Table A-1 to help you make sure that the trial/risk burn considers the full range of compounds potentially emitted from a combustor, and the appropriate analytical method. A risk assessment won't necessarily evaluate every metal and potential PIC listed in Table A-1. Once the trial/risk burn stack tests are completed, the risk assessment COPCs are selected from the stack test data, rather than Table A-1.

This discussion lists reference documents for each of the columns in Table A-1 and briefly describes the quality of data associated with these references. The evaluation of chemical toxicity is also discussed at the end of this section. This information is presented for informational purposes only.

A1.1 COLUMN 1: CHEMICAL ABSTRACTS SERVICE (CAS) NUMBER

The CAS number is a unique number assigned to each compound in the table.

A1.2 COLUMN 2: COMPOUND NAME

The most common compound name is listed. Where appropriate, common synonyms are also listed to help you identify particular compounds.

A1.3 COLUMN 3: COMPOUNDS LISTED IN 40 CFR PART 261 APPENDIX VII OR VIII

Appendix VII of Title 40 Code of Federal Regulations (40 CFR) Part 261 identifies compounds for which specific hazardous wastes, from specific and nonspecific sources, are listed (U.S. EPA 1995). Appendix VIII of 40 CFR Part 261 identifies acute hazardous wastes and toxic hazardous wastes associated with commercial chemical products, manufacturing chemical intermediates, and off-specification commercial chemical products (U.S. EPA 1995). This column lists hazardous waste codes for the associated compounds. We provide this list for reference purposes only, because it's commonly cited by other U.S. EPA combustion risk assessment documents as an original source of the product of incomplete combustion (PIC) lists. An explanation of the reasons for including a COPC on this list is beyond the scope of the HHRAP.

A1.4 COLUMN 4: CHEMICAL-SPECIFIC DATA AVAILABLE

This column lists those compounds for which the following are available (as presented in Appendix A-2): (1) chemical-specific physical and chemical information, and (2) chemical-specific fate-and-transport information.

A1.5 COLUMN 5: PICS RECOMMENDED BY U.S. EPA (1994a) FOR ALL HUMAN HEALTH RISK ASSESSMENTS (HHRA)

Compounds marked with an “X” in this column are identified by U.S. EPA (1994a) as PICs to be included in all HHRAs. U.S. EPA (1994a) does not describe the basis or references for including these PICs in all HHRAs. More information regarding these compounds is presented in Section 2.2 of the HHRAP.

A1.6 COLUMN 6: PICS IDENTIFIED IN COMBUSTION UNIT EMISSIONS (U.S. EPA 1993)

Compounds marked with an “X” in this column are identified in U.S. EPA (1993) as PICs. The source documents cited by U.S. EPA (1993) are described in the following subsections. These references have been cited by this and other Agency reference documents as “sources” of information regarding PIC emissions from hazardous waste combustors. U.S. EPA (1993) has, in turn, been cited by later guidance documents as a “source” of information regarding PIC emissions from hazardous waste combustors. However, as is indicated by the listing of the references from Dempsey and Oppelt (1993) (which is a summary of existing information), many of the reference documents appear to simply cite additional “sources” of information. We’ve not yet identified the original research and sampling data regarding PIC emissions but, based on a preliminary review of the information below, the sources of the “original” information cited by all of the most common reference documents may be limited and may have been published over 15 years ago.

A1.6.1 Demsey and Oppelt (1993)

The sections of Demsey and Oppelt (1993) regarding PICs from hazardous waste combustion facilities (“Combustion Byproduct Emissions” and “Table XVII: Organics that Could Potentially be Emitted from Devices Burning Hazardous Waste”) cite the following references:

- U.S. EPA (1989b) didn’t include a list of PICs from combustion sources. U.S. EPA (1989b) discussed ways of ensuring that PIC emissions don’t pose an unacceptable risk to human health and the environment. Stack gas carbon monoxide (CO) concentration is a good indicator of combustion efficiency; therefore, controlling CO is a prudent and reasonable approach for minimizing the potential risk from PICs. The destruction and removal efficiency (DRE) standard of 40 CFR Part 264.242(a) limits stack emissions of principal organic hazardous constituents (POHCs) to 0.01 percent (or 0.0001 percent for dioxin-containing waste) of the quantity of POHC in the waste. This standard, however, does not impose a limit on PICs. Therefore, a limit of 100 parts per million by volume (ppmv) (Tier I) was imposed, below which PIC emissions do not pose unacceptable risks to human health. The proposed rule allows a waiver to the 100-ppmv CO limit, by (1) restricting total hydrocarbon (THC) emissions to 20 ppmv (Tier II), or (2) showing that THC emissions do not pose an unacceptable health risk by using prescribed risk assessment procedures.

The above limitations were also provided in the Federal Register, dated January 23, 1981 (U.S. EPA 1981) and April 27, 1990 (U.S. EPA 1990b)

- U.S. EPA (1981) doesn't contain any information regarding PICs not contained in U.S. EPA (1989b). There is no discussion of "risk" in this document. Although the notice deals with permitting standards, there is no risk-based approach, and it appears to be an entirely technical discussion. Specifically, it deals with updated material for specific parts of 40 CFR.
    - 40 CFR Part 122 (Incinerator Facility Permits)
    - 40 CFR Part 264 (General Standards for Hazardous Waste Incineration)
    - 40 CFR Part 265 (Interim Status Standards for Hazardous Waste Incineration)

  Standards are technology-based, not risk-based.

- U.S. EPA (1990a) describes amendments to the hazardous waste incinerator regulations for the following purpose:

  > Improve control of toxic metal emissions, HCl emissions, and residual organic emissions; amend the definitions of incinerators and industrial furnaces; propose definitions for plasma arc incinerators and infrared incinerators; propose to regulate carbon regeneration units as thermal treatment devices; and make a number of minor revisions to permitting procedures.

  U.S. EPA (1990a) also states the following:

  > The database on PIC emissions is limited therefore, the risk assessments may under-estimate risk. The assessments consider only the organic compounds that have been actually identified and quantified. Zero to 60 percent of total unburned hydrocarbon emissions have been chemically identified at any particular facility. Thus, the bulk of the hydrocarbon emissions have not been considered in those risk assessments. Although many of the unidentified, unquantified organic compounds may be non-toxic, some fraction of the organic emissions is undoubtedly toxic. . . .data on typical PIC emissions from hazardous waste combustion sources were compiled and assessed in recent EPA studies. These studies identified 37 individual compounds in the stack gas of the eight full-scale hazardous waste incinerators tested, out of which 17 were volatile compounds and 20 semivolatile compounds. Eight volatile compounds (benzene, toluene, chloroform, trichloroethylene, carbon tetrachloride, tetrachloroethylene, chlorobenzene, and methylene chloride), and one semivolatile compound (naphthalene) were identified most frequently in more than 50 percent of the tests. Some of these compounds are carcinogenic.

  The sources for these statements appear to be Wallace et al. (1986) and Trenholm and Lee (1986).

  Trenolm and Lee (1986), prepared by Andrew R. Trenholm of Midwest Research Institute and C.C. Lee at the U.S. EPA Hazardous Waste Engineering Research Laboratory, discussed that emissions from incinerators are only characterized for constituents listed in Appendix VIII. However, constituents not listed in Appendix VIII are also emitted from the stacks.

  Data was obtained from HWERL-sponsored tests at eight hazardous waste incinerators, nine boilers that co-fired hazardous wastes, and five mineral processing kilns that fired hazardous wastes as fuel. In addition, SVOC emissions data for two municipal solid waste incinerators and seven coal-fired power plants were also reviewed. The common PICs are presented in the following table:

Volatile PICs Most Frequently Present in Stack Gases	
VOCs	**SVOCs**
Benzene	Naphthalene
Toluene	Phenol
Carbon Tetrachloride	Bis(2-ethylhexyl)phthalate
Chloroform	Diethylphthalate
Methylene Chloride	Butylbenzylphthalate
Trichloroethylene	Dibutylphathlate
Tetrachloroethylene	
1,1,1-Trichloroethane	
Chlorobenzene	

Tests were conducted for three incinerator runs to search for constituents not listed in Appendix VIII . These constituents include:

Non-Appendix VIII Constituents Present in Highest Concentrations in Stack Gases	
Acetone	Ethylbenzaldehyde
Ethylbenzene	Ethylbenzoic acid
Acetophenone	Ethylphenol
Benzaldehyde	Ethylphenyl-ethanone
Benzenedicarboxaldehyde	Ethynylbenzene
Benzoic acid	Phenylacetylene
Chlorocyclohexanol	1,1'-(1,4-phenylene)bisethanone
Cyclohexane	Phenylpropenol
Cyclohexanol	Propenylmethylbenzene
Cyclohexene	Tetramethyloxirane
Dioctyl adipate	Trimethylhexane
Ethenyl ethylbenzene	

Emission rates of compounds not in the waste feed were also provided.

C U.S. EPA (1985) didn't include a list of PICs from combustion sources. U.S. EPA (1985) discussed views and reviews by the Environmental Effects, Transport, and Fate Committee of the Science Advisory Board of issues related to the environmental impacts of the incineration of liquid hazardous wastes at sea and on land. Several issues were addressed, including issues concerning the combustion and incineration of hazardous waste. Major findings of the committee were as follows:

- Fugitive emissions and spills may release as much or more material to the environment than the direct emissions from waste incineration processes.

- Numerous PICs are formed during the combustion processes. However, only a fraction of them are identified or detected. It is possible that the aggregate of all compound emissions that are not categorized as other POHCs or PICs can be more toxic and pose greater risks than those listed. Although 99.99 percent DRE has been claimed, if the unburned or undetected hydrocarbon output is included, the DRE may actually be less than 99.99 percent. Therefore, the concept of destruction efficiency used by EPA was found to be incomplete and not useful for subsequent exposure assessments. All emissions and effluents must be identified and quantified, including their physical form and characteristics.

- Local site-specific conditions must be used in characterizing exposure to receptors from waste incinerator emissions.

- The evaluation of exposure durations and concentrations should be based on a detailed assessment of transport processes and the habits of the exposed organisms. The role of food chains needs particular attention.

- At a minimum, the toxicities of representative emissions and effluents from incinerators should be tested on sensitive life stages of representative aquatic and terrestrial vertebrates, invertebrates, and plants of ecological importance.

C U.S. EPA (1990b) does not include a list of PICs from combustion sources. It was prepared by the PIC subcommittee of the Science Advisory Board to review the OSW proposal to control emissions of PICs from hazardous waste incinerators by instituting process controls that are based on CO and THC emission concentrations. U.S. EPA risk assessments indicate that emissions of PICs at currently measured levels are not likely to produce human effects. However, because the current DRE standard applies only to designated POHCs, 99.99 percent DRE does not preclude the possibility that emission of PICs could present significant human health risk. The following summarizes the major findings of the subcommittee review.

- The concept of using CO and THC as guidance for incinerator operational control is reasonable.

- At low CO levels, CO correlates well with THC; therefore, limiting CO in order to ensure high combustion efficiency and low THC levels is reasonable. At high CO concentrations, CO and THC do not correlate well; therefore, relying solely on the controlling of CO may not provide a reasonable control for THC. Continuous emissions monitoring of THC is preferred. Quantification of PICs alone is not practical with the sampling techniques that are available, primarily because PICs are normally emitted in the range of parts per billion (ppb) to parts per trillion (ppt).

- A 100-ppmv limit for CO is reasonable. However, supporting documentation does not demonstrate that a CO concentration of 100 ppmv is better than 50 ppmv or 150 ppmv.

- Continuous emissions monitoring of THC with a cold system appears to be practical for routine operations. However, a hot transfer line produces better analysis of THC concentrations and detection of a larger fraction of the THCs emitted.

- The database characterizing PICs in emissions would not allow a correlation to be established with CO or THC levels for various combustion devices and conditions. Limited data introduces large uncertainties into U.S. EPA's risk assessment. Therefore, U.S. EPA's site-specific risk assessment process is limited in its usefulness in establishing acceptable THC levels. However, the risk assessment procedures are risk-based.

C U.S. EPA (1987) is a report prepared by Andrew R. Trenholm, Acurex Corporation, California, and staff members from the U.S. EPA Hazardous Waste Engineering Research Laboratory in Cincinnati, Ohio. The paper discussed the lack of information on total emissions from combustion of hazardous wastes, particularly under conditions of less than optimal performance. The focus issue was whether additional constituents that are listed in Appendix VIII or not listed in Appendix VIII which were not identified in early tests might be emitted from hazardous waste combustion units. To address this issue and related issues, U.S. EPA initiated this project to qualitatively and quantitatively study the characteristics of all possible effluents, under steady-state and transient conditions. The following summarizes the major findings:

- THC emissions detected as specific compounds ranged from 50 to 67 percent for five runs and were 91 percent for one run. The fraction of THC not detected is most likely explained by uncertainty in the measurements or other analytical problems.

- Methane accounted for the largest fraction of THC.

- Oxygenated aliphatic compounds made up the largest class of compounds among the SVOCs, both in total mass and number of compounds.

- Transient upsets did not cause significant increases in the concentration of SVOCs or most VOCs. Three VOCs that were increased were methane, methylene chloride, and benzene.

- Particulate and HCl emissions did not change between the steady-state and transient test runs.

C Duval and Rubey (1976) was prepared by D.S. Duval and W.A. Rubey of the University of Dayton Research Institute, Ohio. The objective of the study was to provide data from which requirements can be assigned for the thermal disposal of kepone. This report was primarily concerned with the high-temperature destruction of kepone, with DDT and Mirex used as comparative Analog. Laboratory tests were conducted to establish destruction temperature characteristics of the vaporized pesticides at preselected residence times. The following summarizes the major findings.

  - Kepone was essentially destroyed at a 1-second residence time and a temperature range of 500°C to 700°C, depending on the pesticide.

  - Major decomposition products detected were hexachlorocyclopentadiene and hexachlorobenzene for both kepone and Mirex. These products were formed in different thermal regions.

  - The study demonstrated that the chemical nature of the effluent products depends on the temperature and residence time that the basic molecule experiences.

C Duval and Rubey (1977) discusses the experimental destruction temperature and residence time relationships for various PCB compounds and mixtures of PCBs. The document states that “upon thermal stressing in air, PCBs decompose to low-molecular-weight products.” However, the document doesn’t identify any of these low-molecular-weight products. In fact, the document states directly that the products weren’t identified in the study. It further recommends that additional research be conducted on the “degradation products and effluents.”

C Dellinger et al. (1984) was prepared by Barry Dellinger and others of the University of Dayton, Ohio. This paper presented the gas-phase thermal stability method under controlled laboratory conditions to rank the incinerability of compounds. The objective of this study was to determine the gas-phase thermal decomposition properties of 20 hazardous organic compounds.

The compounds were selected on the basis of (1) frequency of occurrence in hazardous waste samples, (2) apparent prevalence in stack effluents, and (3) representativeness of the spectrum of hazardous waste organic waste materials. The following summarizes the major findings.

  - Gas-phase thermal stability method is a more effective means of ranking the incinerability of hazardous compounds in a waste.

  - Numerous PICs were formed during the thermal decomposition of most of the compounds tested. However, PICs were not identified.

  - Destruction efficiency of 99.99 percent is achieved at 2 seconds mean residence time in flowing air at 600°C to 950 °C.

  - No single physical or chemical property describes the ranking scheme for incinerability.

C Dellinger et al. (1986) was prepared by Barry Dellinger, B. Douglas, L. Hall, John L. Graham, Sueann L. Mazer, and Wayne A. Rubey of the University of Dayton Research Institute, Dayton, Ohio, and Myron Malanchuk of U.S. EPA, Cincinnati, Ohio. The paper discussed the development of an incineration model based on laboratory studies conducted by using the nonflame mode of hazardous waste thermal decomposition. The results of these studies were compared to the flame-mode studies and field tests to evaluate the incineration model proposed. The model was based on the premise that incinerators do not operate continuously at optimum conditions. As a result, 1 percent or more of the feed and its flame treatment products must undergo further decomposition in the nonflame region to meet the DRE criterion of greater than 99.99 percent.

In the past, several methods were used to rank the incinerability of compounds. Nonflame studies, however, indicated that tests on compounds conducted at low oxygen concentrations provided a better correlation with field tests to determine the relative incinerability of compounds. Four experimental studies were conducted to develop and expand the database on POHCs and PICs.

Studies were conducted on individual compounds to evaluate degradation compounds and PICs from the original parent compound. The thermal degradation of 2,3',4,4',5-PCB was studied under four reaction atmospheres (at varying levels of oxygen) at a constant gas phase residence time of 2.0 seconds. Tests were conducted at temperatures ranging from 500ºC to 1,000ºC. Tests indicated that the yield of combustion products decreased with increased oxygen levels. Numerous major degradation products were identified from the thermal degradation of 2,3',4,4',5-PCB, including:

- Penta-, tetra-, and trichlorodibenzofurans
- Tetra- and trichlorobiphenyls
- Tri- and dichlorobenzene
- Tetra- and trichloronaphthalene
- Tri- and dichlorochlorophenylethlyene
- Tetrachlorobiphenylenes
- C_9H_8OCl
- $C_{10}H_3Cl_3$

Thermal decomposition of chloroform was studied. Numerous decomposition products were identified, including:

- CCl_4
- $C_2H_4Cl_2$
- C_2HCl_3
- C_2HCl_5
- C_2Cl_2
- C_2Cl_4
- C_3Cl_4
- C_4Cl_6

Thermal decomposition of polychlorinated phenols was studied in nitrogen (N_2) and oxygen atmospheres because of the potential formation of polychlorinated dibenzodioxins. Pentachlorophenol (PCP) thermal decomposition was studied. Numerous decomposition products of PCP were identified in N_2 and/or air atmospheres, including:

- Dichlorobutadiyne (in N_2)
- Tetrachloroethylene (in air)
- Tetrachloropropyne (in air)
- Trichlorofuran (in air)
- Tetrachlorofuran (in air)
- Trichlorobenzene (in N_2 and air)
- Tetrachlorobenzene (in N_2 and air)
- Pentachlorobenzene (in N_2 and air)
- Hexachlorobenzene (in N_2)

  - Octachlorostyrene (in N_2)
  - Hexachlorodihydronaphthalene (in N_2 and air)

The paper concluded that PICs in the air atmosphere may have formed directly from the parent material, whereas, in the nitrogen atmosphere, the principal PICs may have evolved from the thermal decomposition of other PICs.

C Kramlich et al. (1984) doesn’t include a list of PICs from combustion sources. It was prepared by J.C. Kramlich, W.R. Seeker, and M.P. Heap of Energy and Environmental Research Corporation, California; and C.C. Lee of the Industrial Waste Combustion Group, U.S. EPA. This paper presented a research program to study the flame-mode incineration of hazardous waste liquids in laboratory scale reactors. The objective of this study was to supply the flame-mode data used in evaluating the applicability of various approaches to ranking the ease of incinerability.

Five compounds were tested—chloroform, 1,1-dichloroethane, benzene, acrylonitrile, and chlorobenzene—because (1) their range of incinerabilities is broad, and (2) they are representative of liquid hazardous wastes. The following summarizes the findings.

- The flame section of the incinerator destroys greater than 99.995 percent of the wastes.

- The post-flame region destroys the remainder of the wastes.

- The destruction efficiency is reduced because of flame-related failures.

- Incinerability ranking depends on actual failure condition.

- No incinerability ranking system completely predicts the destruction efficiency of the compounds tested for all failure conditions.

C Trenholm and Hathaway (1984) was prepared by Andrew Trenholm and Roger Hathaway of Midwest Research Institute (MRI) in Missouri, and Don Oberacker, U.S. EPA, Cincinnati, Ohio. PICs were defined as any Appendix VIII hazardous organic constituent detected in the stack gas but not present in the waste feed at a concentration of 100 micrograms per gram or higher. Benzene and chloroform were the most commonly found PICs. PIC emissions were comparable to POHC emissions in concentration and total mass output. This document discussed PIC formation mechanisms and criteria for PIC formations.

MRI conducted a series of tests at eight operating hazardous waste incineration facilities and analyzed the collected samples for PICs. The tests were part of the technical support of U.S. EPA’s preparation of a regulatory impact analysis for hazardous waste incinerators. Each incinerator had a liquid injection burner, and some facilities also included a rotary kiln or hearth. Three incinerators had no air pollution control devices. The remaining five had wet scrubbers for HCl control, and four of these had other particulate control devices. Twenty-nine compounds were classified as PICs from the eight incinerator tests and are presented in Table A1.6-1. In general, PIC concentrations were slightly higher than POHC concentrations, although this ratio varied from site to site. PIC output rate very rarely exceeded 0.01 percent of the POHC input rate. The document stated that the measurement of Appendix VIII compounds at low concentrations in the waste feed, auxiliary fuel, and inflow streams to control systems is often necessary to explain the presence of PICs.

TABLE A1.6-1
PICS IDENTIFIED BY TRENHOLM AND HATHAWAY (1984)

PICs Found In Stack Effluents			
PIC	Number of Facilities	Low Concentration (ng/L)	High Concentration (ng/L)
Benzene	6	12	670
Bromochloromethane	1	14	14
Bromodichloromethane	4	3	32
Bromoform	3	0.2	24
Bromomethane	1	1	1
Carbon disulfide	1	32	32
Chlorobenzene	3	1	10
Chloroform	5	1	1,330
Chloromethane	1	3	3
Chlorophenol, o-	1	22	22
Dibromochloromethane	4	1	12
Dichlorobenzene	1	4	4
Diethyl phthalate	1	7	7
Dimethylphenol, 2,4-	1	21	21
Fluoranthene	1	1	1
Hexachlorobenzene	1	7	7
Methyl ethyl ketone	1	3	3
Methylene chloride	2	2	27
Methylene bromide	1	18	18
Naphthalene	3	5	100
Nitrophenol, o-	2	2	50
Pentachlorophenol	1	6	6
Phenol	2	4	22
Pyrene	1	1	1
Tetrachloroethylene	3	0.1	2.5
Toluene	2	2	75
Trichlorobenzene	1	7	7
Trichloroethane, 1,1,1,-	3	0.1	1.5
Trichlorophenol, 2,4,6-	1	110	110

Notes:

ng/L = Nanograms per liter
PIC = Product of incomplete combustion

C Olexsey et al. (1985) was prepared by Robert A. Olexsey and others of the U.S. EPA Hazardous Waste Engineering Research Laboratory in Cincinnati, Ohio. This document discussed PIC generation mechanisms and criteria for PIC formations. The paper provided data on emissions of PICs during full-scale tests conducted on incinerators and boilers burning hazardous waste (Trenholm et al. 1984; Castaldini et al. 1984). The documents referenced by this paper summarized a series of full-scale tests conducted on seven incinerators and five boilers conducted by U.S. EPA to support its regulatory development for incinerators and boilers. Commonly found PICs identified in these tests are presented in Tables A1.6-2 and A1.6-3.

TABLE A1.6-2
VOLATILE PICS MOST FREQUENTLY IDENTIFIED IN BOILER EMISSIONS (OLEXSY, HUFFMAN, AND EVANS 1985)

PIC	Number of Facilities	Low Concentration (ng/L)	High Concentration (ng/L)
Benzene	3	9.4	270
Chloroform	5	4.2	1,900
Chloromethane	4	4.6	410
Dichloroethane, 1,2-	3	1.3	1,200
Methylene chloride	4	83	2,000
Tetrachloroethylene	5	0.3	760
Trichloroethane, 1,1,1-	3	5.9	270

Notes:
ng/L = Nanograms per liter
PIC = Product of incomplete combustion

TABLE A1.6-3
VOLATILE PICS MOST FREQUENTLY IDENTIFIED IN INCINERATOR EMISSIONS (OLEXSY, HUFFMAN, AND EVANS 1985)

PIC	Number of Facilities	Low Concentration (ng/L)	High Concentration (ng/L)
Benzene	6	12	670
Chloroform	5	1	1,330
Methylene chloride	2	2	27
Tetrachloroethylene	3	0.1	2.5
Toluene	2	2	75
Trichloroethane, 1,1,1-	3	0.1	1.5

Notes:
ng/L = Nanograms per liter
PIC = Product of incomplete combustion

- For incinerators, ratios of PIC emissions to POHC input ranged from 0.00007 to 0.0028 percent; and ratios of PIC emissions to POHC emissions ranged from 0.01 to 3.89. For boilers, ratios of PIC emissions to POHC input ranged from 0.0032 to 0.3987 percent, and ratios of PIC emissions to POHC emissions ranged from 5.44 to 22.5. These data indicated that PIC emissions were higher for boilers than for incinerators; that is, PIC emissions were reduced with increased POHC

DRE which is higher for incinerators. The document proposed seven methods to control PICs and recommended further research on PIC generation mechanisms and control technologies.

C Trenholm et al. (1992) was prepared by Andrew R. Trenholm and David W. Kapella of MRI in North Carolina and Gary D. Hinshaw of MRI in Missouri. The paper discusses the following issues regarding emissions from incinerators that burn hazardous waste:

- emissions of specific constituents presented in Appendix VIII,
- emissions of specific compounds or types of compounds, and
- data on the size and molecular weight of compounds emitted.

The following were among the major issues discussed.

- PICs were studied through U.S. EPA-sponsored tests at eight incinerators, nine industrial boilers, and five mineral processing kilns. The study was limited to compounds presented in Appendix VIII. In all, 52 organic compounds (32 VOCs and 20 SVOCs) were identified. The VOC concentrations were significantly higher than the SVOC concentrations. PICs listed in this paper included:
  - benzene,
  - toluene,
  - carbon tetrachloride,
  - trichloromethane,
  - dichloromethane,
  - trichloroethene,
  - tetrachloroethene,
  - 1,1,1-trichloroethane,
  - cholorobenzene,
  - naphthalene, and
  - phenol.

- From the U.S. EPA-sponsored tests, (1) volatile compounds listed in Appendix VIII identified were only a fraction—sometimes about one-half—of the total organic compounds identified, and (2) semivolatile compounds not listed in Appendix VIII identified were three to 30 times the quantity of organic compounds listed in Appendix VIII . Table A1.6-4 lists the compounds identified by the U.S. EPA-sponsored tests.

TABLE A1.6-4
MOST FREQUENTLY IDENTIFIED PICS
(TRENHOLM, KAPELLA, AND HINSHAW 1992)

Appendix VIII Volatile Organic Compounds	Appendix VIII Semivolatile Organic Compounds	Compounds Not Listed in Appendix VIII
Benzene	Bis(2-Ethylhexyl)phthalate	(1,4-Phenylene)bisethanone, 1,1'-
Carbon tetrachloride	Butylbenzylphthalate	Acetone
Chlorobenzene	Dibutylphtahlate	Acetophenone
Chloroform	Diethylphthalate	Benzaldehyde
Methylene chloride	Naphthalene	Benzenedicarboxaldehyde
Tetrachloroethylene	Phenol	Benzoic acid
Toluene		Chlorocyclohexanol
Trichloroethane, 1,1,1-		Cyclohexane
Trichloroethylene		Cyclohexanol
		Ethylbenzene
		Ethylbenzoic acid
		Ethylphenol
		Ethylphenyl-ethanone
		Ethynylbenzene
		Phenylpropenol
		Propenylmethylbenzene
		Tetramethyloxirane
		Trimethylhexane

- A study of hazardous waste incinerator stack effluent was conducted to characterize the types of compounds emitted. Twenty-nine compounds were identified at a concentration range of 0.1 to 980 nanograms per liter. Methane, chloromethane, and chloroform accounted for more than one-half of the total mass of VOCs detected. Other than methane, oxygenated aliphatic hydrocarbons formed the highest fraction of the total emissions.

- Based on the incinerator stack effluent study, it was found that as combustion conditions deteriorate, increases in mass emissions are first noted with VOCs. Emissions of these compounds, most notably C1 to C3 compounds, increase proportionately more than larger compounds. For larger compounds, available data indicate that emission increases are more likely to be aromatic compounds.

A1.6.3 CARB (1990b)

CARB prepared “Technical Support Document of Proposed Dioxins Control Measures for Medical Waste Incinerators” to meet the requirements of California Health and Safety Code Section 39666 that a needs report be prepared for proposed rules. The report presents a proposed airborne toxic control measure for dioxin emissions from medical waste-burning facilities. The report concentrates on dioxin, furan, and cadmium emissions, although other pollutants detected during the tests are listed. Table A1.6-5 lists these pollutants.

TABLE A1.6-5
COPCS IDENTIFIED BY CARB (1990b)

COPC		
Ammonia	Dibromoethane, 1,2-	Nickel
Arsenic	Dichloroethane	Nitrogen oxides
Benzene	Dichloromethane	PM
Bromodichloromethane	Dichloropropane, 1,2-	PAHs
Cadmium	Ethylbenzene	Sulfur dioxide
Carbon dioxide	Freon	Tetrachloroethene
Carbon monoxide	Hydrocarbon, total	Tetratrichloromethylene
Carbon tetrachloride	Hydrogen chloride	Toluene
Chlorobenzenes	Hydrogen fluoride	Tribromomethane
Chlorodibromomethane	Iron	Trichlorethane
Chloroform	Lead	Trichloroethane, 1,1,1-
Chlorophenols	Manganese	Trichloroethylene
Chromium, hexavalent	Mercury	Trichlorotrifluroethane
Chromium, total	Mesitylene	Vinyl chloride
Copper	Methyl isobutyl ketone	Xylenes
Cumene	Napthalene	Zinc

Notes:
PAH = Polynuclear aromatic hydrocarbons
PM = Particulate matter

A1.6.4 CARB (1991)

CARB prepared “Air Pollution Control at Resource Recovery Facilities 1991 Update” to update information presented in its 1984 report, entitled “Air Pollution Control at Resource Recovery Facilities.” Specifically, the document updates available guidelines concerning incinerator technology, emissions control technology, and emission limits for municipal waste, hospital waste, biomass, tire, manure, landfill and digester gas, and sewer sludge incinerators. The document states that its guidelines represent levels that have been achieved by existing facilities.

In addition, the document summarizes the ultimate analysis of waste types undergoing treatment in the facilities described above. An appendix summarizes stack gas analysis data for numerous operating facilities. Pollutants identified in the analyses are summarized in Table A1.6-6.

TABLE A1.6-6
STACK GAS ANALYSIS DATA (CARB 1991)

	Incinerator Type [a]						
Pollutant	**Municipal Waste (5)**	**Hospital Waste (7)**	**Biomass (4)**	**Manure (1)**	**Tire (1)**	**Landfill Gas (20)**	**Sewage Sludge and Digester Gas (5)**
Nitrogen oxides	U	U	U	U	U	U	U
Sulfur oxides	U	U	ND	U	U	U	U
Particulate matter	U	U	U	U	U	U	U
Carbon monoxide	U	U	U	U	U	U	U
Total hydrocarbons	U	U	U	U	U	U	U
Hydrogen chloride	U	U	NA	NA	U	NA	NA
Hydrogen fluoride	U	NA	NA	NA	NA	NA	NA
Amonnia	NA	NA	U	NA	U	NA	NA
Carbon dioxide	U	U	U	U	U	NA	U
Oxygen	U	U	U	U	U	NA	U
Arsenic	U	U	U	NA	U	U	U
Beryllium	U	NA	NA	NA	U	U [b]	U
Cadmium	U	U	U	NA	ND	U [b]	U
Chromium (total)	U	U	U	NA	U	U	U
Chromium (hexavalent)	ND	U	NA	NA	U	NA	NA
Copper	U	NA	NA	NA	NA	U	NA
Mercury	U	U	NA	NA	ND	U	U
Iron	NA	NA	U	NA	NA	NA	NA
Manganese	NA	NA	U	NA	NA	NA	NA
Nickel	U	U	U	NA	ND	U	U
Lead	U	U	U	NA	ND	U	U
Zinc	NA	NA	NA	NA	NA	U	NA
Polyaromatic hydrocarbons [b]	U	NA	U	NA	U	NA	NA
Polychlorinated biphenyls [b]	U	ND	U	NA	U	NA	NA
CP [b]	U	NA	U	NA	U	NA	NA
CB [b]	U	NA	U	NA	U	NA	NA
Benzene	U	U	U	NA	NA	NA	NA
Polychlorinated dibenzo(p) dioxins [b]	U	U	U	NA	U	NA	NA
Polychlorinated dibenzofurans [b]	U	U	U	NA	U	NA	NA
2,3,7,8-Tetrachloro dibenzo(p)dioxin equivalents [b]	U	U	U	NA	U	NA	U

Notes:

U = Detected in at least one emission test
ND = Not detected in any emission test
NA = No analysis

[a] Number in parentheses indicates the number of facilities for which data were tabulated.

[b] Isomers and/or homologues that were not detected were added to total values at one-half the detection limit; pollutant may not have actually been detected.

A1.6.5 U.S. EPA (1988)

This document, referenced by some documents as a 1989 document, was prepared in 1988.

U.S. EPA prepared "Hospital Waste Combustion Study: Data Gathering Phase" to assemble available information on hospital waste combustion so that U.S. EPA can evaluate whether airborne pollutant emissions from hospital waste combustion should be regulated. While preparing this document, U.S. EPA reviewed the pertinent literature to determine which studies would be helpful in completing the database on toxic emissions from medical waste incinerators. The report clearly addresses only those pollutants for which emissions data were found. The data reviewed were mostly for larger, controlled air incinerators; and the more commonly used retort incinerators were not evaluated.

The study identified several categories of pollutants that were measured in stack gases; these are discussed in the following paragraphs.

Where evaluated, acid gases were detected in stack gases. For example, HCl was detected in 24 of 28 tests; HCl concentration not recorded in the remaining four tests.

Particulate matter (PM) was detected in all stack tests for 30 facilities at concentrations ranging from 0.001 grains per dry standard cubic foot (gr/dscf), at a facility with PM add-on control devices, to 0.22 gr/dscf at facilities without such control devices.

Trace metals were detected in stack tests for three medical waste incineration facilities. Metals detected include arsenic, cadmium, chromium, iron, manganese, nickel, and lead. The document also states that fine-particle enrichment processes could lead to emissions of molybdenum, tin, selenium, vanadium, and zinc. However, test results for these trace metals are not presented.

With respect to organic emissions, dioxins and furans were detected in emissions from three facilities, both with and without pollution control devices. Other organic emissions detected in stack tests cited in this report include CO, THC, trichlorotrifluoroethane, tetrachloromethane, tetrachloroethene, and trichloroethylene.

In a stack testing conducted on three Canadian biomedical waste incinerators, PCBs and PAHs were either not detected (one facility) or not analyzed (two facilities).

A1.6.6 CARB (1996)

In May 1996, CARB prepared "Proposed Amendments to the Emission Inventory Criteria and Guidelines Report Published in Accordance with the Air Toxics 'Hot Spots' Information and Assessment Act of 1987." The purpose of the report is to present the basis of CARB's recommended amendments to the Air Toxics Hot Spots Program. The report states that California Health and Safety Code (HSC) 44321 requires CARB to compile the list of toxic substances that must be monitored from "designated reference lists of substances." Therefore, the document is not a primary source of toxics emission information. The primary sources of information are mandated by California HSC 44321, as follows:

- California HSC 44321(a): National Toxicology Program, International Agency for Research on Cancer

- California HSC 44321(b): Governor's List of Carcinogens and Reproductive Toxicants

- California HSC 44321(c): CARB
- California HSC 44321(d): Hazard Evaluation System and Information Service
- California HSC 44321(e): U.S. EPA
- California HSC 44321(f): California HSC

The lists of toxic substances presented in the document are not restricted to incinerator facilities, but apply to any facility discharging airborne pollutants to the atmosphere. The document also removes numerous substances, primarily medicinal compounds, from lists of toxic chemicals that must always be evaluated, and places them on lists of toxic compounds that require evaluation only if a facility manufactures that substance.

A1.7 COLUMN 7: U.S. EPA-RECOMMENDED AND POTENTIAL PICS (1994a; 1994b)

Compounds marked with an “X” in the appropriate cells are identified in U.S. EPA (1994a and 1994b). Based on information presented in U.S. EPA (1994b), these tables were developed from available U.S. EPA data and from lists of toxic compounds from various U.S. EPA programs. Because the source lists were not developed as lists of toxic PICs, U.S. EPA deleted compounds that were not appropriate (U.S. EPA 1994b). U.S. EPA acknowledged the importance of using focused studies to develop a PIC list that is (1) appropriately protective of the environment, and (2) not excessively burdensome on the regulated community. Nevertheless, Tables 1 and 2 in U.S. EPA (1994b) were compiled as draft lists for use during the interim period. Tables in U.S. EPA (1994b) were to be revised as additional PIC data were collected. U.S. EPA Permits and State Program Division is currently updating these tables; however, a target completion date is not available. Tables 1 and 2 are based on the following (U.S. EPA 1994b):

- Hazardous waste constituent list in 40 CFR Part 261, Appendix VIII
- hazardous air pollutants (HAP) list
- Office of Research and Development list of organic compounds found in combustion devices developed for U.S. EPA (1993)

The following compounds were deleted from this list:

- Pesticide compounds not likely to be a PIC
- Federal Drug Administration-regulated drugs
- Carcinogenic sugar substitutes
- Compounds without chemical-specific listings (for example, “coal tar”)
- Compounds without U.S. EPA-established sampling and analysis methods
- Metallic compounds (because of difficulty in analyzing the specific compounds; metals are still included in elemental totals)

- Compounds with low octanol-water partition coefficients and no inhalation toxicity data
- Compounds with low toxicity values
- Naturally-occurring plant toxins

Specific compounds were retained on Tables 1 and 2 on the following basis:

- Pesticides with a molecular structure simple enough to be of concern as a PIC
- Compounds with very high octanol-water partition coefficients

A1.8 COLUMN 8: PICS ACTUALLY DETECTED IN STACK EMISSIONS

Compounds marked by an “X” in the appropriate cells are PICs that have actually been detected in stack emissions. U.S. EPA compiled this list by evaluating the studies highlighted in Section A1.6.

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APPENDIX A-2

CHEMICAL-SPECIFIC PARAMETER VALUES

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LIST OF VARIABLES AND COMPOUND-SPECIFIC PARAMETERS

D_{air} = Density of air (g/cm^3)
D_{forage} = Density of forage (g/cm^3)

Ba_{beef} = Biotransfer factor in beef
(mg COPC/kg FW tissue)/(mg COPC/day) OR (day/kg FW tissue)
$Ba_{chicken}$ = Biotransfer factor in chicken
(mg COPC/kg FW tissue)/(mg COPC/day) OR (day/kg FW tissue)
Ba_{egg} = Biotransfer factor in eggs
(mg COPC/kg FW tissue)/(mg COPC/day) OR (day/kg FW tissue)
Ba_{milk} = Biotransfer factor in milk
(mg COPC/kg FW tissue)/(mg COPC/day) OR (day/kg FW tissue)
Ba_{pork} = Biotransfer factor in pork
(mg COPC/kg FW tissue)/(mg COPC/day) OR (day/kg FW tissue)
BAF_{fish} = Bioaccumulation factor in fish
(mg COPC/kg FW tissue)/(mg COPC/L total water column)
OR (L water/kg FW tissue)
BCF_{fish} = Bioconcentration factor in fish (L/kg FW OR unitless)
Br_{ag} = Plant-soil bioconcentration factor in aboveground produce
(:g COPC/g DW plant)/(:g COPC/g DW soil)—unitless
$Br_{forage/silage}$ = Plant-soil bioconcentration factor in forage and silage
(:g COPC/g DW plant)/(:g COPC/g DW soil)—unitless
Br_{grain} = Plant-soil bioconcentration factor in grain
(:g COPC/g DW grain)/(:g COPC/g DW soil)—unitless
$\mathrm{Br}_{\mathrm{rootveg}}$ = Plant-soil bioconcentration factor for belowground produce
(:g COPC/g DW plant)/(:g COPC/g DW soil)—unitless
$BSAF_{fish}$ = Biota-sediment accumulation factor in fish
(mg COPC/kg lipid tissue)/(mg COPC/kg sediment)—unitless
B_{vol} = Volumetric air-to-leaf biotransfer factor in leaf
(:g COPC/L FW plant)/(:g COPC/L air)—unitless
Bv_{ag} = COPC air-to-plant biotransfer factor for aboveground produce
(:g COPC/g DW plant)/(:g COPC/g air)—unitless
$Bv_{forage/silage}$ = Air-to-plant biotransfer factor in forage and silage
(:g COPC/g DW plant)/(:g COPC/g air)—unitless

c = Junge constant = 1.7 x 10^{-04} (atm-cm)

D_a = Diffusivity of COPC in air (cm^2/s)
D_w = Diffusivity of COPC in water (cm^2/s)

$f_{oc,bs}$ = Fraction of organic carbon in bottom sediment (unitless)
$f_{oc,s}$ = Fraction of organic carbon in soil (unitless)
$f_{oc,sw}$ = Fraction of organic carbon in suspended sediment (unitless)
f_{water} = Fraction of COPC in water (unitless)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
Fw = Fraction of wet deposition that adheres to plant surfaces (unitless)

H = Henry’s law constant

Inhalation CSF = Inhalation cancer slope factor $(mg/kg\text{-}day)^{-1}$
Inhalation URF = Inhalation unit risk factor $(\mu g/m^3)^{-1}$

Kd_s = Soil-water partition coefficient (mL water/g soil OR cm^3 water/g soil)
Kd_{sw} = Suspended sediment-surface water partition coefficient
(L water/kg suspended sediment OR cm^3 water/g suspended sediment)
Kd_{bs} = Bed sediment-sediment pore water partition coefficient
(L water/kg bottom sediment OR cm^3 water/g bottom sediment)
K_{ow} = Octanol/water partitioning coefficient
(mg COPC/L octanol)/(mg COPC/L octanol)—unitless
K_{oc} = Soil organic carbon-water partition coefficient (mL water/g soil)
ksg = COPC soil loss constant due to biotic and abiotic degradation (yr^{-1})

MW = Molecular weight of COPC (g/mole)

p°_{L} = Liquid-phase vapor pressure of COPC (atm)
p°_{S} = Solid-phase vapor pressure of COPC (atm)

Oral CSF = Oral cancer slope factor $(mg/kg\text{-}day)^{-1}$

R = Universal gas constant ($atm\text{-}m^3/mol\text{-}K$)
RCF = Root concentration factor
(μg COPC/g DW plant)/(μg COPC/mL soil water)
RfC = Reference concentration (mg/m^3)
RfD = Reference dose (mg/kg/day)
Rp = Interception factor of edible portion of plant (unitless)

S = Solubility of COPC in water (mg COPC/L water)
ΔS_f = Entropy of fusion [$\Delta S_f/R$ = 6.79 (unitless)]
S_T = Whitby's average surface area of particulates (aerosols)
= 3.5×10^{-06} cm^2/cm^3 air for background plus local sources
= 1.1×10^{-05} cm^2/cm^3 air for urban sources

$t_{1/2}$ = Half-time of COPC in soil (days)
T_a = Ambient air temperature (K)
T_m = Melting point temperature (K)
TEF = Toxicity equivalency factor (unitless)

Vp = Vapor pressure of COPC (atm)

APPENDIX A-2

The following sections provide the general methodology and references we used to determine our recommended parameter values. You can find the recommended parameter values themselves in the HHRAP Companion Database (also known as the HHRAP database). In the HHRAP database we provide compound-specific values for

1. physical and chemical properties;
2. fate-and-transport parameters; and
3. Health benchmarks for chronic and acute exposure.

A2-1 GENERAL ANALYSIS AND METHODOLOGY

This section describes the general analysis and criteria we followed to determine our recommended contaminant-specific parameter values.

1. We compared parameter values among primary and available sources of applicable data, placing priority for selection on sources that are (1) U.S. EPA or State agency peer reviewed, (2) recent, (3) original sources of the values, and (4) generally accessible.

2. We further further researched and evaluated sources of parameter values, and to the extent possible, included observations affecting usability in parameter-specific discussions for each compound.

3. We selected only parameter values from sources that could be verified and cited.

4. We generally preferred source-recommended parameter values. As necessary, we determined parameter values using correlations or equations, using input parameter values provided in the HHRAP database.

5. When multiple parameter values were reported in a particular source (e.g., CHEMFATE), we selected the source-recommended value in most cases. If more than one parameter value is recommended by the source, then we selected the recommended value that falls closest to the average of the source-recommended values.

6. When reviewing and selecting parameter values from published literature or studies, we preferred using measured values over other types of data.

A2-2 PHYSICAL AND CHEMICAL PROPERTIES

A2-2.1 Molecular Weight (*MW*)

Molecular weight (*MW*) of a compound is defined as the sum of atomic weights of all atoms in the compound’s molecule.

For most compounds (except PCDDs and PCDFs, and methyl mercury), we obtained *MW* values from one of the following sources, as cited:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.

- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.

- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.

- Lide, D.R. 2003. *CRC Handbook of Chemistry and Physics, 83rd Edition*. CRC Press. Boca Raton, Florida.

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- Recommended value reported in published literature. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

PCDDs and PCDFs We obtained *MW* values for PCDDs and PCDFs from U.S. EPA (2000).

- U.S. EPA. 2000. *Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds*. Draft Final Report. Office of Research and Development. Washington, D.C. EPA/600/P-00/001 Bc. September.

Mercuric Compounds We obtained *MW* values for elemental mercury and mercuric chloride from the preferred sources listed above. We obtained the *MW* value for methyl mercury from U.S. EPA (1997b).

- U.S. EPA. 1997b. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* OAQPS and ORD. EPA-452/R-97-005. December.

A2-2.2 Melting Point Temperature (T_m)

Melting point temperature (T_m) is the temperature of the compound (in degrees Kelvin [K]) at which the solid state of the compound undergoes a phase change to a liquid phase. At ambient temperatures and at an atmospheric pressure of 1 atmosphere, compounds are generally in either a solid or liquid state.

For most compounds (except PCDDs and PCDFs), we obtained values for T_m from one of the following sources, as cited:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.

- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.

- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.

- Lide, D.R. 2003. *CRC Handbook of Chemistry and Physics, 83rd Edition*. CRC Press. Boca Raton, Florida.
- O'Neil, M., and A. Smith. 2001. *The Merck Index: An Encyclopedia of Chemicals, Drugs, and Biologicals.* 13th Edition. Merck and Company, Inc. Rahway, New Jersey.
- Recommended value reported in published literature. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

When a recommended range is provided instead of a singe numerical value, we selected the average of the range.

<u>PCDDs and PCDFs</u> We obtained T_m values for PCDDs and PCDFs from U.S. EPA (2000). U.S. EPA (2000) provides T_m values for PCDDs and PCDFs, and states that the values were obtained from various cited literature sources. We selected the midpoint of the range of values provided in U.S. EPA (2000).

<u>Metals</u> We obtained T_m values for metals, if available and except for mercury, from the preferred sources listed above.

<u>Mercuric Compounds</u> We obtained *Tm* values for elemental mercury and mercuric chloride from the list of preferred sources noted above. A *Tm* value for methyl mercury wasn't available in preferred sources or literature.

A2-2.3 Vapor Pressure (*Vp*)

The vapor pressure (*Vp*) of a substance is defined as the pressure in atmospheres exerted by the vapor (gas) of a compound when it is under equilibrium conditions. It provides a semi-quantitative rate at which it will volatilize from soil and/or water.

For most compounds (except PCDDs and PCDFs), we obtained values for *Vp* from the following preferred sources:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.
- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.
- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.
- Lide, D.R. 2003. *CRC Handbook of Chemistry and Physics, 83rd Edition*. CRC Press. Boca Raton, Florida.
- O'Neil, M., and A. Smith. 2001. *The Merck Index: An Encyclopedia of Chemicals, Drugs, and Biologicals.* 13th Edition. Merck and Company, Inc. Rahway, New Jersey.

C Calculated using procedures set forth by:

Lyman et al. 1990. *Handbook of Chemical Property Estimation Methods*. American Chemical Society. Washington, DC.

- Recommended value reported in published literature. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

If a *Vp* value wasn't available in SCDM, then we obtained a vapor pressure value from CHEMFATE, PHYSPROP, CRC Handbook of Chemistry and Physics, or The Merck Index. If more than one value was recommended by the source, then we selected the recommended value that fell closest to the average of the recommended values. If a recommended value wasn't available, we selected a value measured at 25/C. If more than one value measured at 25/C was available, then we selected the value falling closest to the average of the available values. If no value was available at 25/C, then we followed the same method using values determined within the range of 20 to 30/C.

If no vapor pressure values were available in any of the preferred references, then we used the procedures described in Lyman et al. (1990) to calculate vapor pressure.

For nonmetallic substances, if vapor pressure wasn't available, we obtained a normal boiling point from the sources and used it to assign a default vapor pressure. If the boiling point is <25/C at 1 atmosphere (atm) , we selected a default vapor pressure of 760 Torr, and we assume the substance is a gas at 25/C. As with SCDM, if no vapor pressure is available for a substance and the normal boiling point is equal or greater than 25/C, we assume the substance is in a particulate form, rather than a gaseous form, and we assigned a vapor pressure of zero in order to calculate *Fv*. We made this assumption because the absence of a vapor pressure value often reflects an extremely low and difficult to measure (under standard conditions) value for nongaseous substances (U.S. EPA 2004b).

SCDM prefers CHEMFATE-recommended values over estimated or calculated values. If more than one recommended value is in CHEMFATE, SCDM selected the highest of the values. If a recommended value is not available, SCDM uses a value measured at 25/C. If more than one value measured at 25/C is available, SCDM uses the highest one. If no value is available at 25/C, values determined within the range of 20 to 30/C are used. If more than one value measured at the same temperature is available and none is recommended, SCDM uses the highest value. If no temperature is specified in CHEMFATE for all vapor pressure measurements for a substance, SCDM uses the highest value. For values not available in CHEMFATE, SCDM followed a similar approach to that outlined above.

<u>PCDDs and PCDFs</u> We obtained *Vp* values for PCDDs and PCDFs from U.S. EPA (2000). Congener group average values were substituted for missing individual congener specific values.

<u>Metals</u> Except for mercury compounds, metals that do not have *Vp* values in the references above we assigned a *Vp* value of zero since they are assumed to be (1) nonvolatile at ambient temperatures, and (2) insoluble in water, except as certain weak acids.

<u>Mercuric Compounds</u> Mercury is a relatively volatile compound. We obtained the *Vp* value for elemental mercury from a preferred source listed above. We obtianed the *Vp* value for mercuric chloride from U.S. EPA (1997b). The *Vp* value for methyl mercury was not available in preferred sources or literature.

A2-2.4 Aqueous Solubility (*S*)

The aqueous solubility (*S*) of a compound is defined as the saturated concentration of the compound in water (mg COPC/L water) at a given temperature and pressure, usually at soil/water temperatures and atmospheric pressure (Montgomery and Welkom 1991).

For most compounds (except PCDDs and PCDFs and metals), we obtained values for *S* from the following preferred sources:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.

- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.

- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.

- Dean, J.A. 2002. *Lange's Handbook of Chemistry.* 15th Edition. McGraw-Hill. New York.

- Calculated using procedures set forth by:

  Lyman et al. 1990. *Handbook of Chemical Property Estimation Methods*. American Chemical Society. Washington, DC.

- Recommended value reported in published literature. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

If a *S* value wasn't available in SCDM, then we obtained solubility values from CHEMFATE, PHYSPROP, or Lange's Handbook of Chemistry. If more than one value was recommended by the source, then we selected the recommended value falling closest to the average of the recommended values. If a recommended value wasn't available, we selected a value measured at 25/C. If more than one value measured at 25/C is available, then we selected the value falling closest to the average of the available values. If no value was available at 25/C, we used the same method using values determined within the range of 20 to 30/C.

If no solubility values were available in any of the preferred references, we used the procedures described in Lyman et al. (1990) to calculate vapor pressure.

SCDM prefers CHEMFATE-recommended values over estimated or calculated values. If more than one recommended value is in CHEMFATE, SCDM selected the highest of the values. If a recommended value is not available, SCDM uses a value measured at 25/C. If more than one value measured at 25/C is available, SCDM uses the highest one. If no value is available at 25/C, values determined within the range of 20 to 30/C are used. If more than one value measured at the same temperature is available and none is recommended, SCDM uses the highest value. If no temperature is specified in CHEMFATE for all vapor pressure measurements for a substance, SCDM uses the highest value. For values not available in CHEMFATE, SCDM followed a similar approach to that outlined above for determining values reported in Table A-1.

PCDDs and PCDFs We obtained *S* values for PCDDs and PCDFs from U.S. EPA (2000). Congener group average values were substituted for missing individual congener specific values.

Metals We obtained *S* values for metallic compounds, if available and except for mercury, from the preferred sources listed for organic and inorganic compounds.

Mercuric Compounds Mercury is a relatively volatile compound. We obtained *S* values for elemental mercury and mercuric chloride from the preferred sources listed above. Methyl mercury is stated in the Merck Index as being insoluble in water. Therefore, we assigned an *S* value of zero for methyl mercury.

A2-2.5 Fraction of Contaminant Air Concentration in the Vapor Phase (F_v)

Organics For most compounds (except metals and some mercury compounds), we calculated the fraction of contaminant air concentration in the vapor phase (F_v) using the following equation:

$$Fv = 1 - \frac{c\ S_T}{p^\circ_L + c\ S_T}$$

Equation A-2-1

C Junge, C. E. 1977. *Fate of Pollutants in the Air and Water Environments*, Part I; Suffet, I. H., Ed.; Wiley; New York. Pages 7-26.

If the contaminant is a liquid at ambient temperatures (that is, when p°_L is known), we used Equation A-2-1 to calculate F_v using the *Vp* value recommended for that contaminant in the HHRAP database. If the contaminant is a solid at ambient temperatures (that is, when p°_S is known), we used the following equation (Bidleman 1988) calculate p°_L from p°_S, for use in Equation A-2-1 (using *Vp* and T_m values presented for each contaminant in the HHRAP database):

$$\ln\left(\frac{p^\circ_L}{p^\circ_S}\right) = \frac{\Delta\ S_f}{R}\ \frac{(T_m - T)}{T}$$

Equation A-2-2

where

c = Junge constant = 1.7 x 10^{-04} (atm-cm)
p°_L = Liquid phase vapor pressure of compound (atm)
p°_s = Solid phase vapor pressure of compound (atm)
R = Universal ideal gas constant (atm-m^3/mole$^-$K)
)S_f = Entropy of fusion [)S_f/R = 6.79 (unitless)]
S_T = Whitby's average surface area of particulates (aerosols)
T_a = Ambient air temperature (K)—assumed to be 25°C or 298 K

C Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

According to Bidleman (1988), Equation A-2-1 assumes that the Junge constant (*c*) is constant for all compounds. However, *c* can depend on (1) the compound (sorbate) molecular weight, (2) the surface concentration for monolayer coverage, and (3) the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate.

Metals As in U.S. EPA (1994b), we assumed all metals (except mercury) are present predominately in the particulate phase and not in the vapor phase ($Vp = 0$), and therefore, assigned F_v values of zero.

Mercuric Compounds Elemental mercury and mercuric chloride are relatively volatile and exist in the vapor phase (U.S. EPA 1997b). Therefore, we calculated the *Fv* value for elemental mercury using Equation A-2-1.

Based on discussions in U.S. EPA (1997b), we assigned mercuric chloride an *Fv* value of 0.85. Also, consistent with information provided in U.S. EPA (1997b), we assumed methyl mercury doesn't exist in the air phase and, therefore, assigned it an *Fv* of zero.

A2-2.6 Henry's Law Constant (*H*)

Henry's Law constant (*H*) is also referred to as the air-water partition coefficient, and is defined as the ratio of the partial pressure of a compound in air to the concentration of the compound in water at a given temperature under equilibrium conditions. Henry's Law constant values generally can be (1) calculated from the theoretical equation defining the constant, (2) measured, or (3) estimated from the compound structure.

For most compounds (excluding PCDDs and PCDFs), we obtained *H* values from the following preferred sources:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.
- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.
- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.
- Calculated using procedures set forth by:

  Lyman et al. 1990. *Handbook of Chemical Property Estimation Methods*. American Chemical Society. Washington, DC.

$$H = \frac{Vp \cdot MW}{S} \qquad \text{Equation A-2-3}$$

where

H = Henry's Law constant (atm-m^3/mole)
Vp = Vapor pressure of COPC (atm)
S = Solubility of COPC in water (mg COPC/L water)

H values calculated with Equation A-2-3 used *MW, S,* and *Vp* values found in the HHRAP database.

PCDDs and PCDFs We obtained *H* values for PCDDs and PCDFs from U.S. EPA (2000). We substituted congener group average values for missing individual congener specific values.

Metals We obtained *H* values for metallic compounds, if available and except for mercury, from the preferred sources listed above. Metals that do not have *H* values in the references above we assigned a value of zero since the subject metals are assumed to be (1) nonvolatile at ambient temperatures, and (2) insoluble in water, except as certain weak acids.

Mercuric Compounds We obtained *H* values for elemental mercury, mercuric chloride, and methyl mercury from U.S. EPA (1997b).

A2-2.7 Diffusivity in Air (D_a) and Water (D_w)

Diffusivity or diffusion coefficients in air (D_a) and water (D_w) are used to calculate the liquid or gas phase transfer of a contaminant into a water body.

For most compounds (except PCDDs and PCDFs), we obtained diffusion coefficients in air (D_a) and water (D_w) values from the following preferred source:

- U.S. EPA. 2004c. *WATER9—Air Emissions Models Wastewater Treatment.* Version 2.0.0. OAQPS. Research Triangle Park. North Carolina. July 1.

If values weren't available in U.S. EPA (2004c), we calculated values using procedures set forth by:

- U.S. EPA. 1997b. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* OAQPS and ORD. EPA-452/R-97-005. December.

$$D_{a,i} = \frac{1.9}{(MW_i)^{2/3}} \qquad \text{Equation A-2-4}$$

$$D_{w,i} = \frac{22\times10^{-5}}{(MW_i)^{2/3}} \qquad \text{Equation A-2-5}$$

D_w and D_a values calculated with Equations A-2-4 and A-2-5 used *MW* values recommended in the HHRAP database.

PCDDs and PCDFs We obtained Diffusivity values in air and water for 2,3,7,8-TCDD and 2,3,7,8-TCDF from U.S. EPA (2004c). For all other congeners of PCDDs and PCDFs, (1) we recommend a default *Dw* value of 8 x 10^{-06} cm^2/s, and (2) we estimated *Da* values from PCDD and PCDF values using the following equation recommended by U.S. EPA (2000) and obtained from Thibodeaux (1979):

$$\frac{D_x}{D_y} = \left(\frac{MW_y}{MW_x}\right)^{0.5} \qquad \text{Equation A-2-6}$$

where

$D_{x,y}$ = Diffusivities in air of compounds *x* and *y* (cm^2/s)
$MW_{x,y}$ = Molecular weights of compounds *x* and *y* (g/mol)

We calculated *Da* values for PCDD congeners using the *Da* value and *MW* for 2,3,7,8-TCDD. We calculated *Da* values for PCDF congeners using the *Da* value and *MW* for 2,3,7,8-TCDF. This approach is consistent with the methodology specified in U.S. EPA (2000). Values for diffusivity in water range from 1 x 10^{-06} to 1 x 10^{-05} cm^2/s; therefore, U.S. EPA (1995b) recommended a default value of 8 x 10^{-06} cm^2/s. Diffusivity values calculated using Equations A-2-4 and A-2-5 were within the range specified by U.S. EPA (1995b).

Metals We obtained dDiffusivity values for metallic compounds, if available and except for mercury, from the preferred sources listed for organic and inorganic compounds. If values for metals were not available in the preferred sources, we assigned a default value of zero since metals (except mercury and chromium) are generally considered (1) nonvolatile at ambient temperatures, and (2) insoluble in water, except as certain weak acids.

Mercuric Compounds We obtained the diffusivity value for elemental mercury from the WATER9 database (U.S. EPA 2004c). We calculated diffusivity values for mercuric chloride and methyl mercury using Equations A-4 and A-5.

A2-2.8 Octanol/Water Partitioning Coefficient (K_{ow})

The *n*-octanol/water partitioning coefficient (K_{ow}) is defined as the ratio of the solute concentration in the water-saturated *n*-octanol phase to the solute concentration in the *n*-octanol-saturated water phase (Montgomery and Welkom 1991).

For most compounds (except PCDDs and PCDFs, and mercuric chloride), we obtained K_{ow} values from the following preferred sources:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.

- Syracuse Research Corporation (SRC). 2003a. CHEMFATE Database. SRC. Syracuse, NY.

- Syracuse Research Corporation (SRC). 2003b. PHYSPROP Database. SRC. Syracuse, NY.

- Lide, D.R. 2003. *CRC Handbook of Chemistry and Physics, 83rd Edition*. CRC Press. Boca Raton, Florida.

- Recommended value reported in published literature. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

PCDDs and PCDFs We obtained K_{ow} values for the PCDDs and PCDFs from U.S. EPA (2000). Congener group average values were substituted for missing individual congener specific values.

Metals We obtained K_{ow} values for metals, if available and except for mercury, from the preferred sources listed above. K_{ow} values for metals not reported in the above sources we assumed to be zero. This assumption is based on the affinity of most metals to octanol approaches zero.

Mercuric Compounds We obtained the K_{ow} value for elemental mercury from the list of preferred sources above. The K_{ow} value for mercuric chloride comes from U.S. EPA (1997b). We couldn't find a K_{ow} value for methyl mercury.

A2-2.9 Soil Organic Carbon-Water Partition Coefficient (K_{oc})

The soil organic carbon-water partition coefficient (K_{oc}) or the organic carbon normalized soil sorption coefficient is defined as the ratio of adsorbed compound per unit weight of organic carbon to the aqueous solute concentration (Montgomery and Welkom 1991).

The partitioning of ionizing organic compounds can be significantly influenced by soil pH. Because of the soil mechanisms that are inherently involved, we discuss K_{oc} values for the ionizing organics and nonionizing organics separately.

A2-2.9.1 Ionizing Organic Compounds

Ionizing organic compounds include amines, carboxylic acids, and phenols. These compounds contain the functional groups that ionize under specific pH conditions, and include the following:

- Organic acids (2,4,6-trichlorophenol; pentachlorophenol; 2,3,4,5-tetrachlorophenol; 2,3,4,6-tetrachlorophenol; 2,4,5-trichlorophenol; 2,4-dichlorophenol; 2-chlorophenol; phenol; 2,4-dimethylphenol; 2-methylphenol; 2,4-dinitrophenol; and benzoic acid)

- Organic bases—n-nitroso-di-n-propylamine; n-nitrosodiphenylamine, and 4-chloroaniline)

We obtained K_{oc} values for ionizing organic compounds from the following:

C U.S. EPA. 1996. *Soil Screening Guidance: Technical Background Document and User's Guide*. Office of Solid Waste and Emergency Response. Washington, DC. EPA/540/R-95/128. May.

The K_{oc} values for ionizing organic compounds provided by U.S. EPA (1996) were estimated on the basis of the degree of ionization and the relative proportions of neutral and ionized species. K_{oc} values for ionizing compounds can vary vastly, depending on the pH conditions in the environment. Therefore, for the aforementioned ionizing organic compounds, K_{oc} values in the HHRAP database are based on a pH value of 6.8 to correlate with typical environmental soils.

K_{oc} values were estimated on the basis of the assumption that the sorption of ionizing organic compounds is similar to hydrophobic organic sorption, because the soil organic carbon is the dominant sorbent. According to U.S. EPA (1996), for low pH conditions, these estimated values may overpredict sorption coefficients, because they ignore sorption to components other than organic carbon.

A2-2.9.2 Nonionizing Organic Compounds

Nonionizing organic compounds include volatile organics, chlorinated pesticides, polynuclear aromatic hydrocarbons (PAHs), and phthalates. We obtained K_{oc} values for nonionizing organic compounds from the following:

C U.S. EPA. 1996. *Soil Screening Guidance: Technical Background Document and User's Guide*. Office of Solid Waste and Emergency Response. Washington, DC. EPA/540/R-95/128. May.

U.S. EPA (1996) calculated the geometric mean value from various measured values. For compounds for which K_{oc} values are not directly provided by U.S. EPA (1996), we calculated K_{oc} values using K_{ow} correlation equations provided in U.S. EPA (1996), as obtained from DiToro (1985). We used K_{ow} values recommended in the HHRAP database to calculate K_{oc} values.

For most semi-volatile nonionizing organic compounds -

$$\log K_{oc} = 0.00028 + (0.983 * \log K_{ow}) \qquad \text{Equation A-2-7}$$

C DiToro, D.M. 1985. "A Particle Interaction Model of Reversible Organic Chemical Sorption" *Chemosphere*. 14(10):1503-1538.

For the purposes of applying Equation A-2-7, we define semi-volatile compounds as having a Henry's Law Constant (H) value less than 10-3, consistent with general descriptions provided in Lyman et al. (1990).

For volatile nonionizing organics, chlorinated benzenes, and certain chlorinated pesticides -

$$\log K_{oc} = 0.0784 + (0.7919 * \log K_{ow}) \qquad r^2 = 0.97 \qquad \text{Equation A-2-8}$$

For the purposes of applying Equation A-2-8, we define volatile compounds as having a Henry's Law Constant (*H*) value greater than 10-3, consistent with general descriptions provided in Lyman et al. (1990).

PCDDs and PCDFs For PCDDs and PCDFs, we used the following correlation equation obtained from Karickhoff, et al. (1979), as cited by U.S. EPA (2000), , and K_{ow} values provided in the HHRAP database to calculate K_{oc} values.

$$\log K_{oc} = \log K_{ow} - 0.21 \qquad (n = 10,\ r^2 = 1.0)$$ Equation A-2-9

- Karickhoff, S.W., D.S. Brown, and T.A. Scott. 1979. "Sorption of Hydrophobic Pollutants on Natural Sediments." *Water Resources.* 13:241-248.

Metals We couldn't find K_{oc} values for metals within the preferred sources or available literature.

Mercuric Compounds We couldn't find K_{oc} values for mercury compounds within the preferred sources or available literature.

A2-2.10 Partitioning Coefficients for Soil-Water (Kd_s), Suspended Sediment-Surface Water (Kd_{sw}), and Bottom Sediment-Sediment Pore Water (Kd_{bs})

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* is 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}) describes the partitioning of a compound between surface water and suspended solids or sediments. The bed sediment-sediment pore-water partition coefficient (Kd_{bs}) describes the partitioning of a compound between the bed sediments and bed sediment pore-water.

For most compounds (including PCDDs and PCDFs), we obtained Kd_s values from the following preferred sources:

- U.S. EPA. 2004b. *Superfund Chemical Data Matrix (SCDM).* OERR. Washington, D.C. January.

- U.S. EPA. 1996. *Soil Screening Guidance: Technical Background Document and User's Guide*. Office of Solid Waste and Emergency Response. Washington, DC. EPA/540/R-95/128. May.

- Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. "Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture." Oak Ridge National Laboratory. Oak Ridge, Tennessee.

- RTI. 1996. *Chemical Properties for SCDM Development.* Prepared for U.S. EPA Office of Emergency and Remedial Response. Washington, DC.

- Calculated using procedures consistent with:

  U.S. EPA. 1993d. *Review Draft Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. Office of Health and Environmental Assessment. Office of Research and Development. EPA-600-AP-93-003. November 10.

We assume that soil organic carbon is the dominant sorbing component in soils and sediments. Therefore, *Kd* values not directly available in the preferred sources were calculated using the following fraction organic carbon (f_{OC}) correlation equations provided in U.S. EPA (1993d):

$$Kd_s = f_{oc,s} \cdot K_{oc}$$ Equation A-2-10

$$Kd_{sw} = f_{oc,sw} \cdot K_{oc}$$ Equation A-2-11

$$Kd_{bs} = f_{oc,bs} \cdot K_{oc}$$ Equation A-2-12

- U.S. EPA. 1993d. *Review Draft Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. Office of Health and Environmental Assessment. Office of Research and Development. EPA-600-AP-93-003. November 10.

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

- 0.002 to 0.024 in soils—for which a mid-range value of $f_{oc,s}$ = 0.01 generally can be used.

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

- 0.03 to 0.05 in bottom sediments—for which a mid-range value of $f_{oc,bs}$ = 0.04 generally can be used.

We calculated *Kd* values using the K_{oc} values recommended in the HHRAP database, and mid-range f_{oc} values recommended by U.S. EPA (1993d).

Metals For metals (except mercury), *Kd* is governed by factors other than organic carbon, such as pH, redox, iron content, cation exchange capacity, and ion-chemistry. Therefore, *Kd* values for metals can't be calculated using the same correlation equations specified for organic compounds. We obtained *Kd* values, except lead and mercury, from the preferred sources listed above. SCDM obtained its values from U.S. EPA (1996), which provides values for *Kd* that are based on pH and are estimated using the MINTEQ2 geochemical speciation model. The MINTEQ2 model analyses were conducted under a variety of geochemical conditions and metal concentrations. The MINTEQ2 pH-dependent *Kd* values were estimated by holding constant the iron oxide at a medium value and the f_{oc} at 0.002.

Because organic carbon does not play a major role in partitioning for the metals, U.S. EPA (1994b) assumed that the partitioning is the same, regardless of the soil, suspended sediment, or bottom sediment phase. Therefore, we assumed that the values for partitioning coefficients Kd_s, Kd_{sw}, and Kd_{bs} for the metals are the same.

We obtained the *Kd* value for lead from the following:

- Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. "Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides Through Agriculture." Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Mercuric Compounds We obtained Kd_s, Kd_{sw}, and Kd_{bs} values for mercury , mercuric chloride, and methyl mercury from U.S. EPA (1997b).

A2-2.11 Soil Loss Constant Due to Degradation (*ksg*)

Soil loss constant due to degradation (*ksg*) reflects loss of a compound from the soil by processes other than leaching. Degradation rates in the soil media include biotic and abiotic mechanisms of transformation. Abiotic degradation includes photolysis, hydrolysis, and redox reactions. Hydrolysis and redox reactions can be significant abiotic mechanisms in soil (U.S. EPA 1990).

Lyman et al. (1990) states that degradation rates can be assumed to follow first order kinetics in a homogenous media. Therefore, the half-life ($t_{½}$) of compounds can be related to the degradation rate constant (*ksg*) as follows:

$$ksg = \frac{0.693}{t_{½}}$$ Equation A-2-13

Ideally, *ksg* is the sum of all biotic and abiotic rate constants in the soil. Therefore, if the $t_{½}$ for all of the mechanisms of transformation are known, the degradation rate can be calculated using Equation A-2-13. However, literature sources don't generally provide sufficient data for all such mechanisms, especially for soil.

For most compounds (except PCDDs and PCDFs, PCBs, metals, and mercury), we calculated *ksg* values using half-life soil values obtained from the following preferred sources:

- Howard, P.H., Boethling, R.S., Jarvis, W.F., Meylan, W.M., and Michalenko, E.M. 1991. *Handbook of Environmental Degradation Rates.* Lewis Publishers. Chelsea, Michigan.

- Recommended value reported in published sources or literature other than the preferred sources listed above. When multiple parameter values were reported in a particular source (e.g., database), we selected the recommended value in most cases.

Half-life values provided in Howard et al. (1991) indicate the disappearance of a substance in ground water or soil; with the principal degradation mechanisms being biodegradation and hydrolysis. Values reported were highly variable because of the different methods used for measurements, in addition to the various controlling factors that could affect them. Therefore, Howard et al. (1991) provided a range of half-life values found in the literature, usually for the fastest reaction mechanism. We used high-end half-life values to calculate *ksg* values.

For half-life values obtained from literature sources, if more than one value was recommended by the source, then we selected the value falling closest to the average of the recommended values. This value was used to calculate the *ksg* value recommended in the HHRAP database. If no recommended values were available, we used a value measured at 25/C. If more than one value measured at 25/C was available, then we used the value falling closest to the average of the available values to calculate the *ksg* value recommended in the HHRAP database. If no value was available at 25/C, we used values determined within the range of 20 to 30/C following the same method.

For contaminants with no reported soil degradation rates, we recommend a default value of zero.

PCDDs, PCDFs, and PCBs For PCDDs, PCDFs, and PCBs, we obtained *ksg* values from U.S. EPA (2000); which discussed experimental studies that were conducted on PCDDs and PCDFs degradation mechanisms. U.S. EPA (2000) stated that based on available studies, it appears reasonable to assign a uniform rate of degradation for all PCDD/F congeners, and PCBs. The specific degradation rate provided by U.S. EPA (2000) was 0.0277 yr^{-1}, which translates to a half-life of 25 years.

Metals For the metals, literature states that the metals are transformed, but not degraded, by such mechanisms; therefore, we assume *ksg* values are zero for metals with no reported half-life values in soil.

Mercuric Compounds For mercury, mercuric chloride, and methylmercury, U.S. EPA (1997b) recommended *ksg* values of zero.

A2-2.12 BIOCONCENTRATION AND BIOTRANSFER FACTORS FOR PLANTS

A2-2.12.1 Root Concentration Factor (*RCF*)

The root concentration factor (*RCF*) is used to calculate the belowground transfer of contaminants from soil to a root vegetable. The *RCF* was developed based on experiments conducted by Briggs et al. (1982) which measured uptake of compounds into barley roots from growth solution.

For compounds with log K_{ow} values of 2.0 and higher, we used the following correlation equation to obtain *RCF* values:

$$\log (RCF) = 0.77 \log K_{ow} - 1.52 \qquad (n = 7, r = 0.981)$$ Equation A-2-14

For compounds with log K_{ow} values less than 2.0, we used the following correlation equation to obtain *RCF* values:

$$\log (RCF - 0.82) = 0.77 \log K_{ow} - 1.52$$ Equation A-2-15

We obtained these equations from the following document:

- Briggs, G.G., R.H. Bromilow, and A.A. Evans, 1982. “Relationships Between Lipophilicity and Root Uptake and Translocation of Non-ionized Chemicals by Barley.” *Pesticide Science*. Volume 13. Pages 495-504.

These equations estimate a *RCF* value in fresh weight (FW) units, which was then converted to dry weight (DW) units using a moisture content of 87 percent in root vegetables (U.S. EPA 1997c; Pennington 1994).

Briggs et al. (1982) derived the correlation equations above from studies using 18 compounds with log K_{ow} values ranging from -0.57 to 4.6. In addition, a validation exercise, in which predictions of dioxin-like compounds (log Kow values from 6.0 to 8.2) in carrots were compared with observations, shows this factor to adequately perform for this class of compounds (Muller et al. 1994). Therefore, *RCF* values for compounds with outlying log K_{ow} values have been capped in line with the test data used to formulate the correlation equation. We assigned compounds with log K_{ow} values less than -0.57 an *RCF* value corresponding to a log K_{ow} value of -0.57. At the high end of the range, we assigned compounds with log K_{ow} values greater than 8.2 (considering the validation studies using dioxin-like compounds) an *RCF* value corresponding to a log K_{ow} value of 8.2.

As in previous U.S. EPA guidance (U.S. EPA 1994a), we recommend using Equation A-2-14 in calculating exposure to dioxin-like compounds. We used K_{ow} values available in the HHRAP database to calculate each *RCF* value.

Metals For metals, no referenced *RCF* values were available in published literature. However, plant-soil biotransfer factors for root vegetables ($Br_{rootveg}$) were available in the literature and, therefore, *RCF* values, normally used to calculate $Br_{rootveg}$ values, aren’t needed for the metals.

Mercuric Compounds No *RCF* values were available in the literature for mercury, mercuric chloride, and methyl mercury. However, plant-soil biotransfer factors for root vegetables ($Br_{rootveg}$) were available in U.S. EPA (1997b) and, therefore, *RCF* values, normally used to calculate $Br_{rootveg}$ values, aren’t needed for the mercuric compounds.

A2-2.12.2 Plant-Soil BCFs in Root Vegetables ($Br_{rootveg}$)

The plant-soil bioconcentration factor for compounds in root vegetables ($Br_{rootveg}$) accounts for uptake from soil to the belowground root vegetables or produce.

For most compounds, we used the following equation from U.S. EPA (1995b) to calculate values for $Br_{rootveg}$ on a dry weight basis:

$$Br_{rootveg} = \frac{RCF}{Kd_s} \qquad \text{Equation A-2-16}$$

We calculated $Br_{rootveg}$ values using the RCF and Kd_s values listed in the HHRAP database.

Metals We obtained the $Br_{rootveg}$ values for metals (except nickel, cadmium, selenium, and arsenic) from Baes et al. (1984). *Br* values in Baes et al. (1984) are dry weight values provided for nonvegetative (reproductive) growth, such as tubers.

For nickel, cadmium, selenium, and arsenic, *we obtained* $Br_{rootveg}$ values from the following document:

- U.S. EPA. 1992b. *Technical Support Document for the Land Application of Sewage Sludge. Volumes I and II.* EPA 822/R-93-001a. Office of Water. Washington, D.C.

For nickel, cadmium, selenium, and arsenic—$Br_{root\ veg}$ values were calculated by multiplying the uptake slope factors [(µg COPC/g DW plant)/(kg COPC applied/hectare)] for root vegetables by a conversion factor of $2\text{x}10^9$ g/hectare soil. In deriving the conversion factor, U.S. EPA (1992b) assumed a soil average dry bulk density of 1.33 g/cm^3 and a soil incorporation depth of 15 cm.

Mercuric Componds We obtained $Br_{rootveg}$ values for mercuric chloride and methyl mercury from U.S. EPA (1997b) on a dry weight basis. The HHRAP methodolgy assumes elemental mercury doesn't deposit onto soils, and therefore there's no plant uptake through the soil. Therefore, a $Br_{rootveg}$ value for elemental mercury isn't applicable.

A2-2.12.3 Plant-Soil BCFs for Aboveground Produce (Br_{ag}) and Forage (Br_{forage})

The plant-soil bioconcentration factor (*Br*) for aboveground produce accounts for the uptake from soil and the subsequent transport of contaminants through the roots to the aboveground plant parts. As addressed in U.S. EPA (1995b), the *Br* value for most compounds is a function of water solubility, which is inversely proportional to K_{ow}. The *Br* value for metals is a function of the bioavailability of the metals in soil.

For all compounds, including PCDDs and PCDFs, (1) the subscript "*ag*" represents aboveground produce which applies to exposed fruits and vegetables, and protected fruits and vegetables, and (2) the subscript "*forage*" represents forage, but the values also apply to silage and grain. For metals, (1) aboveground fruits (both exposed and protected) are represented by $Br_{ag\ (fruit)}$; (2) aboveground vegetables (both exposed and protected) are represented by $Br_{ag(veg)}$, (3) forage is represented by Br_{forage}, but the values also apply to silage, and (4) grains are represented by Br_{grain}.

For most compounds, we used the following correlation equations to calculate values for Br_{ag} and Br_{forage} on a dry weight basis:

$$\log Br_{ag} = 1.588 - 0.578\ (\log K_{ow}) \qquad (n = 29,\ r = 0.73)$$ Equation A-2-17

$$\log Br_{forage} = 1.588 - 0.578\ (\log K_{ow}) \qquad (n = 29,\ r = 0.73)$$ Equation A-2-18

We obtained these correlation equations from;

C Travis, C.C. and A.D. Arms. 1988. Bioconcentration of Organics in Beef, Milk, and Vegetation. *Environmental Science and Technology*. 22:271-274.

Travis and Arms (1988) developed a correlation equation for vegetation that doesn't distinguish between aboveground produce and forage or silage or grain. Due to lack of literature data, we used the Travis and Arms (1988) correlation equation to calculate *Br* values for both aboveground produce and forage.

The above correlation equations for *Br* values were derived from experiments conducted on compound classes such as DDT, pesticides, PCDDs, PCDFs, and PCBs, representing compounds with log K_{ow} values ranging from 1.15 to 9.35. Therefore, *Br* values for compounds with outlying log K_{ow} values have been capped in line with the test data used to formulate the correlation equation. Compounds with log K_{ow} values less than 1.15 we assigned a *Br* value corresponding to a log K_{ow} value of 1.15. At the high end of the range, compounds with log K_{ow} values greater than 9.35 we assigned a *Br* value corresponding to a log K_{ow} value of 9.35. We used the log K_{ow} values listed in the HHRAP database to calculate each *Br* value.

Metals For metals, we derived *Br* values from uptake slope factors provided in the following document:

C U.S. EPA. 1992b. *Technical Support Document for the Land Application of Sewage Sludge*. Volumes I and II. EPA 822/R-93-001a. Office of Water. Washington, DC.

Uptake slopes provided in U.S. EPA (1992b) are the ratio of contaminant concentration in dry weight plant tissue to the mass of contaminant applied per hectare soil. These uptake slopes were multiplied by 2 x 10^9 g/hectare soil to convert to *Br* values. The conversion factor was derived using the U.S. EPA (1992b) assumed soil bulk density of 1.33 g/cm^3, and an incorporation depth of 15 cm.

For other metals, we obtained *Br* values from Baes et al. (1984). Baes et al. (1984) described biotransfer factors (on a dry weight basis) from plant-soil uptake for (1) vegetative growth (leaves and stems) "Bv"; and (2) nonvegetative or reproductive growth (fruits, seeds, and tubers) "Br". Note that *Bv* is defined as the air-to-plant biotransfer factor.

We used the following methodology to derive *Br* values on a dry weight basis:

a. For nickel, cadmium, selenium, zinc, and arsenic, $Br_{ag\ (fruit)}$ values were calculated by multiplying the uptake slope factors with a conversion factor of 2 x10^9 g/ha soil (uptake slope factor and the conversion factor were obtained from U.S. EPA (1992b) for garden fruits). All other metals were obtained by selecting "Br" values for nonvegetative growth (reproductive) provided in Baes et al. (1984).

b. For nickel, cadmium, selenium, zinc, and arsenic, $Br_{ag\ (veg)}$ values were calculated by weighting the uptake slope factors for garden fruits (75%) and leafy vegetables (25%) and multiplying the result with a conversion factor of 2 x10^9 g/ha soil. The uptake slope factors and the conversion factor were obtained from U.S. EPA (1992b). For other metals, "Br" values for nonvegetative (reproductive) growth and "Bv" values for vegetative growth—obtained from Baes et al. (1984)—were weighted as 75% (reproductive) and 25% vegetative. The resulting values were adopted as $Br_{ag\ (\mathrm{veg})}$ values.

c. For nickel, cadmium, selenium, zinc, and arsenic, Br_{forage} values were calculated by multiplying the uptake slope factors with a conversion factor of 2x10^9 g/ha soil. The uptake slope factors and the conversion factor were obtained from U.S. EPA (1992b) for leafy vegetables. For other metals, Br_{forage} values were obtained from Baes et al. (1984). "Bv" values for vegetative growth (such as leaves and stems) in Baes et al. (1984) were used for Br_{forage}.

d. For nickel, cadmium, selenium, zinc, and arsenic, Br_{grain} values were calculated by multiplying the uptake slope factors with a conversion factor of 2 x10^9 g/ha soil. The uptake slope factors and the conversion factor were obtained from U.S. EPA (1992b) for grains/cereals. For other metals, Br_{grain} value was obtained from Baes et al. (1984). "Br" values for nonvegetative growth as recommended by Baes et al. (1984) were used for Br_{grain}.

e. Consumption rates used to obtain the weighted average Br_{ag} value for aboveground produce are as follows: (1) Br_{ag} values for fruits combined with a human consumption rate of fruits of 1.44E-03 kg/kg/day, and (2) Br_{ag} values for vegetables combined with a human consumption rate of vegetables of 1.49E-03 kg/kg/day.

Mercuric Compounds We obtained Br_{ag} values for mercuric chloride and methyl mercury as described above. We obtained Br_{forage} values on a dry weight basis for mercuric chloride and methyl mercury from U.S. EPA (1997b). The HHRAP methodology assumes that elemental mercury doesn't deposit onto soils. Therefore, it's assumed that there's no plant uptake through the soil.

A2-2.12.4 Air-to-Plant BTFs for Aboveground Produce (Bv_{ag}) and Forage (Bv_{forage})

The air-to-plant biotransfer factor (Bv) is defined as the ratio of contaminant concentration in aboveground plant parts to the contaminant concentration in air. We calculated Bv values only for aboveground exposed produce (both fruits and vegetables). We assume that aboveground protected produce (both fruits and vegetables) and belowground produce are protected from air-to-plant transfer. According to U.S. EPA (1995b), root vegetables are assumed to be also protected from air-to-plant transfer.

For most compounds (excluding PCDDs and PCDFs), we calculated the air-to-plant biotransfer factor for aboveground produce (Bv_{ag}) and forage (Bv_{forage}) using correlation equations derived for azalea leaves in the following documents:

- Bacci E., D. Calamari, C. Gaggi, and M. Vighi. 1990. “Bioconcentration of Organic Chemical Vapors in Plant Leaves: Experimental Measurements and Correlation.” *Environmental Science and Technology*. Volume 24. Number 6. Pages 885-889.

- Bacci E., M. Cerejeira, C. Gaggi, G. Chemello, D. Calamari, and M. Vighi. 1992. “Chlorinated Dioxins: Volatilization from Soils and Bioconcentration in Plant Leaves.” *Bulletin of Environmental Contamination and Toxicology*. Volume 48. Pages 401-408.

Bacci et al. (1992) developed a correlation equation using data collected for the uptake of 1,2,3,4-tetrachlorodibenzo-p-dioxin (TCDD) in azalea leaves, and data obtained from Bacci et al. (1990). The *Bv* obtained was then evaluated for 14 organic compounds to develop a correlation equation with K_{ow} and *H*. Bacci et al. (1992) derived the following equations without distinguishing between forage and aboveground produce:

$$\log B_{vol} = 1.065 \log K_{ow} - \log \left(\frac{H}{RT}\right) - 1.654 \qquad (r = 0.957)$$

Equation A-2-19

$$Bv = \frac{\rho_{air} \cdot B_{vol}}{(1 - f_{water}) \cdot \rho_{forage}}$$

Equation A-2-20

where

B_{vol} = Volumetric air-to-plant biotransfer factor (fresh-weight basis)
Bv = Mass-based air-to-plant biotransfer factor (dry-weight basis)
ρ_{air} = 1.19 g/L (Weast 1981)
ρ_{forage} = 770 g/L (Macrady and Maggard 1993)
f_{water} = 0.85 (fraction of forage that is water—Macrady and Maggard [1993])

We calculated *Bv* vlaues using Equations A-2-19 and A-2-20, and the values for *H* and K_{ow} listed in the HHRAP database; based on a *T* of 25°C or 298.1 K. We adopted the resulting *Bv* values for both forage (Bv_{forage}) and aboveground produce (Bv_{ag}). Please note the following uncertainty associated with these variables:

- For organics (except PCDDs and PCDFs), U.S. EPA (1993d) recommended reducing *Bv* values by a factor of 10 before use. This was based on the work conducted by U.S. EPA (1993d) for U.S. EPA (1994a) as an interim correction factor. Welsch-Pausch et al (1995) conducted experiments to determine concentrations of PCDDs and PCDFs in air and resulting biotransfer to welsh ray grass. This was documented in the following:

    - Welsch-Pausch, K.M. McLachlan, and G. Umlauf. 1995. “Determination of the Principal Pathways of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans to Lolium Multiflorum (Welsh Ray Grass)”. *Environmental Science and Technology*. 29: 1090-1098.

A follow-up study based on Welsch-Pausch et al. (1995) was conducted by Lorber (1995) (see discussion below for PCDDs and PCDFs). In a following publication, Lorber (1997) concluded that the Bacci factor reduced by a factor of 100 was close in line with observations made by him through various studies, including the Welsch-Pausch et al. (1995) experiments. Therefore, we calculated *Bv* values using the Bacci et al. (1992) correlation equations and then reduced by a factor of 100 for all organics, excluding PCDDs and PCDFs.

PCDDs and PCDFs For PCDDs and PCDFs, we obtained *Bv* values, on a dry weight basis, from the following:

- Lorber, M. and P. Pinsky. 1999. "An Evaluation of Three Empirical Air-to-Leaf Models for Polychlorinated Dibenzo-p-Dioxins and Dibenzofurans". National Center for Environmental Assessment (NCEA). U.S. EPA. Washington D.C. Accepted for publication in *Chemosphere*.

U.S. EPA (1993d) stated that, for dioxin-like compounds, using the Bacci et al. (1992) equations may overpredict *Bv* values by a factor of 40. This was because the Bacci et al. (1990) and Bacci et al. (1992) experiments did not take photodegradation effects into account. Therefore, U.S. EPA (1993d) recommended that *Bv* values calculated using Equations A-2-19 and A-2-20 be reduced by a factor of 40 for dioxin-like compounds.

However, according to Lorber (1995), the Bacci algorithm divided by 40 may not be appropriate because

- the physical and chemical properties of dioxin congeners are generally outside the range of the 14 organic compounds used by Bacci (1990), and
- the factor of 40 derived from one experiment on 2,3,7,8-TCDD may not apply to all dioxin congeners.

Welsch-Pausch et al. (1995) conducted experiments to obtain data on uptake of PCDDs and PCDFs from air to *Lolium Multiflorum* (Welsh Ray grass). The data includes grass concentrations and air concentrations for dioxin congener groups, but not the invidual congeners. Lorber (1995) used data from Welsch-Pausch et al.(1995) to develop an air-to-leaf transfer factor for each dioxin-congener group. *Bv* values developed by Lorber (1995) were about an order of magnitude less than values calculated using the Bacci et al. (1990) and (1992) correlation equations. Lorber (1995) speculated that this difference could be attributed to several factors including experimental design, climate, and lipid content of plant species used.

We obtained the *Br* values for PCDDs and PCDFs listed in the HHRAP database from the experimentally derived values of Lorber (1995). However, Lorber (1995) stated that these values should be considered carefully by users of this methodology because of the inherent uncertainties associated with the data.

Metals For metals, we found no literature sources for the *Bv* values. U.S. EPA (1995b) quoted from the following document, that metals were assumed not to experience air to leaf transfer:

- Belcher, G.D., and C.C. Travis. 1989. "Modeling Support for the RURA and Municipal Waste Combustion Projects: Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model." Interagency Agreement No. 1824-A020-A1. Office of Risk Analysis, Health and Safety Research Division. Oak Ridge National Laboratory. Oak Ridge, Tennessee. October.

As with the above cited references, we assumed *Bv* values for metals (excluding elemental mercury) are zero.

Mercury The HHRAP assumes that mercury emissions consist of both the elemental and divalent forms. However, only small amounts of elemental mercury is assumed to be deposited based on its behavior in the atmosphere. Elemental mercury either dissipates into the global cycle or is converted to the divalent form. The HHRAP assumes methyl mercury doesn't exist in air emissions or in the air phase. Additional basis for the assumption regarding mercury transfer from air to plants includes

- elemental mercury reaching or depositing onto the plant surfaces is negligible, and
- biotransfer of methyl mercury from air is zero.

This is based on assumptions made regarding speciation and fate and transport of mercury from emissions. Therefore, we assume the *Bv* values for elemental and methyl mercury are zero. The *Bv* value for mercuric chloride (dry weight basis) comes from U.S. EPA (1997b).

If field data suggests otherwise, *Bv* values (1) provided in U.S. EPA (1997b) for methyl mercury can be used, and (2) need to be determined for elemental mercury. Please note that uptake of mercury from air into the aboveground plant tissue is primarily in the divalent form. We assume that a part of the divalent form of mercury is converted to methyl mercury once in the plant tissue.

A2-2.13 BIOTRANSFER FACTORS FOR ANIMALS

The biotransfer factor for animals (*Ba*) is the ratio of contaminant concentration in fresh weight animal tissue to the daily intake of contaminant by the animal.

A2-2.13.1 BTFs for Beef (Ba_{beef}) and Milk (Ba_{milk})

The main route of human exposure to many highly lipophilic compounds is through ingestion of contaminated agricultural products such as beef and milk (McLachlan 1993). The transfer of contaminants from environmental media (e.g., air, soil, water) and food (e.g. grain, silage) into livestock products (e.g., beef, milk) has historically been either determined by direct measurement of contaminants in livestock products, or predicted using regression models.

Organic Compounds
When empirical data are lacking for biotransfer of organc chemicals, one of the most widely used approaches to predict contaminant transfer from environmental media and food to beef tissue and milk are the regression models developed by Travis and Arms (1988), which relate chemical octanol-water partition coefficient (K_{ow}) to biotransfer into beef and milk. These regressions, however, are hampered by the limited log K_{ow} range and questions surrounding the validity of the underlying biotransfer data set. In response, EPA developed a new methodology for predicting beef and milk biotransfer factors (See report entitled: "Methodology for Predicting Cattle Biotransfer Factors" (RTI 2005) for complete documentation of the approach).

Using EPA's updated methodology, we predicted biotransfer factors for organic chemicals with the following single equation:

$$\log Ba_{fat} = -0.099\ (\log K_{ow})^2 + 1.07\ \log K_{ow} - 3.56 \qquad (n = 305,\ r^2 = 0.8259)$$

Equation A-2-21

where

Ba_{fat} = Biotransfer factor ([mg /kg Fat]/[mg/day])
K_{ow} = Octanol-water partition coefficient (unitless).

Values for Ba_{fat} were adjusted to account for the assumed fat content of milk and beef as shown below:

$$Ba_{milk} = 10^{\log Ba_{fat}} \times 0.04 \qquad \text{Equation A-2-22}$$

$$Ba_{beef} = 10^{\log Ba_{fat}} \times 0.19 \qquad \text{Equation A-2-23}$$

The log K_{ow}'s of the chemicals used to derive the equation ranged from -0.67 to 8.2. Therefore we used equation A-2-21 for organic chemicals having a log Kow between -0.67 and 8.2. Compounds with log K_{ow} values less than -0.67 were assigned Ba_{beef} and Ba_{milk} values corresponding to a log K_{ow} value of -0.67. At the high end of the range, compounds with log K_{ow} values greater than 8.2 were assigned Ba_{beef} and Ba_{milk} values corresponding to a log K_{ow} value of 8.2. We used the K_{ow} values in the HHRAP database to calculate Ba_{beef} and Ba_{milk} values.

Highly Metabolized Organic Compounds

As discussed in RTI (2005), Equation A-2-21 might overestimate biotransfer of highly metabolized chemicals, producing an upper bound estimate for these chemicals. Of those chemicals relevant to the HHRAP (and therefore included in the HHRAP companion database), Phthalates and PAHs fall within this group (see HHRAP Chapter 2 for more information on the phthalate and PAH chemical categories).

One way to account for this potential overestimation is to rely upon a metabolism factor to improve model predictions. For example, EPA developed a metabolism factor of 0.01 (i.e., 99% of the chemical ingested is metabolized) for bis-ethylhexyl phthalate (BEHP, See Chapter 2). When this factor is applied to the biotransfer factors predicted using the regression equation recommended above for BEHP, the biotransfer factors are reduced by two orders of magnitude. These metabolism-adjusted predicted biotransfer factors are close in magnitude to the empirically derived biotransfer factors found in the literature, which supports using this metabolism factor. Unfortunately, EPA has not developed metabolism factors for other organic chemicals, due to limited availability of empirically derived data (see Chapter 2).

For those highly metabolized chemicals that don't have metabolism factors, we still consider it reasonable to use estimated Ba values for the following reasons:

- Few chemicals have had all their degradation products identified;
- If identified, the degradation products may in fact be as toxic as, or even more toxic than, the parent compound (the degradation products of PAHs, for example, are toxic). Unless data demonstrates that all degradation products are nontoxic (as is the case for BEHP), the only way to address toxic degradation products in the HHRAP is to include their mass in the mass of the parent chemical; and

- the metabolic degradation products may themselves be persistent. For example, DDT is metabolized to DDD and DDE, which remain persistent.

It should also be noted that not all chemicals are metabolized at the same rate and may remain in animal tissue as the parent compound through establishment of steady state concentrations. In fact, many of the chemicals in the biotransfer data set that are well predicted by Equation A-2-21 are metabolized to other compounds. DDT is metabolized to DDE, and lindane (used to derive the regression, but not in the HHRAP) is metabolized to many different compounds. For DDT and lindane, biotransfer factors are well predicted using Equation A-2-21. Thus, just because a chemical undergoes biotransformation to other compounds, doesn't necessarily mean Equation A-2-21over-estimates its biotransfer.

Lacking sufficient data to identify all degradation products, characterize all degradation products as nontoxic, and ensure that all potentially toxic degradation products are less persistent than the parent chemical, we consider it reasonably protective to use the Ba_{beef} and Ba_{milk} values for the parent chemical as predicted, without adjustment. If a highly metabolized chemical is found to drive the risk assessment, then we recommend re-evaluating the appropriateness of the Ba_{beef} and Ba_{milk} values.

Ionizing Organic Compounds

To improve BTF estimates for organic acids, we used the first-order dissociation constant (pKa) to account for chemical ionization. For these chemicals, K_{ow} is a weighted value calculated based on the fraction of the chemical in the neutral form such that:

$$K_{ow} = K_{ow}n \times (FracNeutral) + K_{ow}i \times (1 - FracNeutral) \qquad \text{Equation A-2-24}$$

where

$K_{ow}n$ = partition coefficient for the neutral species (unitless),
$K_{ow}i$ = partition coefficient for the ionized species (mol/L),
FracNeutral = fraction of neutral species present for organic acids (unitless).

Accounting for the fraction of ionizable organics in the neutral form is important because K_{ow} can vary considerably depending on pH. The cow's small intestine, where chemicals can be absorbed, has a near neutral pH (Umphrey and Staples, 1992). Thus, the neutral fraction is determined using a pH equal to 7 in the following equation (Lee et al., 1990):

$$FracNeutral = \frac{[HA]}{[HA] - [A-]} = \left(1 + 10 pH - pKa\right) \qquad \text{Equation A-2-25}$$

where

[HA] = equilibrium concentration of organic acid (mol/L),
[A-] = equilibrium concentration of anion (mol/L), and
pKa = acid dissociation constant (unitless).

If a value for log $K_{ow}i$ wasn't available, we estimated log $K_{ow}i$ assuming a ratio of log $K_{ow}i$ to log $K_{ow}n$ of 0.015. This ratio is a conservative value developed by EPA to apply to organic acids without data for log $K_{ow}i$ (U.S. EPA, 1996).

Metals

We obtained Ba_{beef} and Ba_{milk} values for metals (except cadmium, mercury, selenium, and zinc), on a fresh weight basis, from Baes et al. (1984). For cadmium, selenium, and zinc, U.S. EPA (1995a) cited *Ba* values derived by dividing uptake slopes [(g COPC/kg DW tissue)/(g COPC/kg DW feed)], obtained from U.S. EPA (1992b), by a daily consumption rate of 20 kg DW per day for beef and dairy cattle.

We obtained our recommended values from Baes et al. (1984) for all metals except cadmium, selenium, and zinc. We calculated *Ba* values for cadmium, selenium, and zinc using uptake slope factors and consumption rates provided in U.S. EPA (1992b) and U.S. EPA (1995a), converting the result to a fresh weight basis by assuming a moisture content of 87 percent in milk and 70 percent in beef. Moisture content in beef and milk were obtained from the following:

- U.S. EPA. 1997c. *Exposure Factors Handbook*. "Food Ingestion Factors". Volume II. EPA/600/P-95/002Fb. August.

- Pennington, J.A.T. 1994. *Food Value of Portions Commonly Used*. Sixteenth Edition. J.B. Lippincott Company, Philadelphia.

We provide the calculated fresh weight Ba_{beef} *and* Ba_{milk} values in the HHRAP database.

Mercuric Compounds

The HHRAP assumes that elemental mercury neither deposits onto soils nor transfers to aboveground plant parts. Therefore, there's no transfer of elemental mercury into animal tissue. Therefore, we recommend *Ba* values of zero for elemental mercury. This based on assumptions made regarding speciation and fate and transport of mercury from emission sources.

If field data suggests otherwise, Ba_{beef} and Ba_{milk} values for elemental mercury can be derived from the uptake slope factors provided in U.S. EPA (1992b) and U.S. EPA (1995a); using the same consumption rates discussed earlier for metals like cadmium, selenium, and zinc.

We derived the Ba_{beef} and Ba_{milk} values listed in the HHRAP database for mercuric chloride and methyl mercury from data in U.S. EPA (1997b). U.S. EPA (1997b) provided Ba_{beef} and Ba_{milk} values for mercury, but did not specify for which form of mercury. As in U.S. EPA (1997b), we assume mercury speciates into 87 percent divalent mercury and 13 percent methyl mercury in herbivore animal tissue. Also, assuming that the Ba_{beef} and Ba_{milk} values provided in U.S. EPA (1997b) were for total mercury in animal tissue, the biotransfer factors in U.S. EPA (1997b) can be apportioned in the fractions assumed to be found in animal tissue. Therefore, we based the values reported in the HHRAP database on the following:

- We converted the default Ba_{milk} value of 0.02 day/kg DW for mercury to a fresh weight basis assuming a 87 percent moisture content in milk (U.S. EPA 1997c; and Pennington 1994). We multiplied the resulting Ba_{milk} (fresh weight) value by (1) 0.13 to obtain a value for methyl mercury, and (2) 0.87 to obtain a value for mercuric chloride (divalent mercury).

- We converted the default Ba_{beef} value of 0.02 day/kg DW for mercury to a fresh weight basis assuming a 70 percent moisture content in beef (U.S. EPA 1997c; and Pennington 1994). We multiplied the resulting Ba_{beef} (fresh weight) value by (1) 0.13 to obtain a value for methyl mercury, and (2) 0.87 to obtain a value for mercuric chloride (divalent mercury).

A2-2.13.2 Biotransfer Factors for Pork (Ba_{pork})

For most compounds we derived Ba_{pork} values using the same method used to estimate Ba_{beef} values (see Section A2-2.13.1 above for a detailed description of the method & its limitations), modifying to reflect an assumed fat content of pork of 23%. Specifically, we used equation A-2-21 to generate Ba_{fat} values, which were then adjusted to account for the assumed fat content of pork as shown below:

$$Ba_{pork} = 10^{\log Ba_{fat}} \times 0.23 \qquad \text{Equation A-2-26}$$

The log K_{ow}'s of the chemicals used to derive the equation ranged from -0.67 to 8.2. Therefore we used equation A-2-21 for organic chemicals having a log Kow between -0.67 and 8.2. Compounds with log K_{ow} values less than -0.67 were assigned Ba_{pork} values corresponding to a log K_{ow} value of -0.67. At the high end of the range, compounds with log K_{ow} values greater than 8.2 were assigned Ba_{pork} values corresponding to a log K_{ow} value of 8.2. We used the K_{ow} values in the HHRAP database to calculate Ba_{pork} values.

This calculation is limited by the assumptions that:

- contaminants bioconcentrate in the fat tissues; and
- effects from differences in metabolism, digestive system, and feeding characteristics between beef cattle and pigs are minimal.

Metals For metals (except cadmium, selenium, and zinc), no data was available in the literature to calculate Ba_{pork} values.

For cadmium, selenium, and zinc, U.S. EPA (1995b) reported *Ba* values derived by dividing uptake slopes [(g COPC/kg DW tissue)/(g COPC/kg DW feed)], obtained from U.S. EPA (1992b), by a daily consumption rate of 4.7 kg DW per day for pigs provided in U.S. EPA (1995a). The dry weight Ba_{pork} values (for cadmium, selenium, and zinc) were converted to a fresh weight basis assuming a moisture content of 70 percent in pork (U.S. EPA 1997c; and Pennington 1994).

Mercuric Compounds The HHRAP assumes that elemental mercury neither deposits onto soils nor transfers to the aboveground plant parts; therefore, there's no transfer of elemental mercury into the animal tissue. Therefore, we recommend *Ba* values of zero for elemental mercury. This is based on the assumptions made regarding speciation and fate and transport of mercury from emissions.

If field data suggests otherwise, Ba_{pork} values for elemental mercury can be derived from the uptake slope factors provided in U.S. EPA (1992b) and U.S. EPA (1995a), using the same consumption rates discussed earlier for metals like cadmium, selenium, and zinc.

We derived Ba_{pork} values for mercuric chloride and methyl mercury from data in U.S. EPA (1997b). U.S. EPA (1997b) provided Ba_{pork} values for mercury, but did not specify for which form of mercury. As in U.S. EPA (1997b), the HHRAP assumes that mercury speciates into 87 percent divalent mercury and 13 percent methyl mercury in herbivore animal tissue. Also, assuming that the Ba_{pork} values provided in U.S. EPA (1997b) were for total mercury in animal tissue, biotransfer factors in U.S. EPA (1997b) can be apportioned in the fractions it is assumed to be found in animal tissue.

Therefore, we converted the U.S. EPA (1997b) Ba_{pork} value of 0.00013 day/kg DW for mercury to a fresh weight basis assuming a 70 percent moisture content in pork (U.S. EPA 1997c; and Pennington 1994).

We multiplied the resulting Ba_{pork} (fresh weight) value by (1) 0.13 to obtain a value for methyl mercury, and (2) 0.87 to obtain a value for mercuric chloride (divalent mercury).

A2-2.13.3 BTFs for Chicken ($Ba_{chicken}$) and Poultry Eggs (Ba_{egg})

Biotransfer factors for chicken ($Ba_{chicken}$) and poultry eggs (Ba_{egg}) are expressed as the ratio of the contaminant concentration in the fresh weight tissue to the contaminant intake from the feed. Biotransfer factors are calculated from bioconcentration factors for chicken and poultry eggs. BCFs are expressed as the ratio of the contaminant concentration in the fresh weight tissue to the contaminant concentration in dry weight soil.

For most compounds we derived Ba_{pork} values using the same method used to estimate Ba_{beef} values (see Section A2-2.13.1 above for a detailed description of the method & its limitations), modifying to reflect an assumed fat content of chicken of 14%, and eggs of 8%. Specifically, we used equation A-2-21 to generate Ba_{fat} values, which were then adjusted to account for the assumed fat content of chicken and eggs as shown below:

$$Ba_{chicken} = 10^{\log Ba_{fat}} \times 0.14 \qquad \text{Equation A-2-27}$$

$$Ba_{egg} = 10^{\log Ba_{fat}} \times 0.08 \qquad \text{Equation A-2-28}$$

The log K_{ow}'s of the chemicals used to derive the equations ranged from -0.67 to 8.2. Therefore we used equation A-2-21 for organic chemicals having a log Kow between -0.67 and 8.2. Compounds with log K_{ow} values less than -0.67 were assigned $Ba_{chicken}$ and Ba_{egg} values corresponding to a log K_{ow} value of -0.67. At the high end of the range, compounds with log K_{ow} values greater than 8.2 were assigned $Ba_{chicken}$ and Ba_{egg} values corresponding to a log K_{ow} value of 8.2. We used the logK_{ow} values in the HHRAP database to calculate $Ba_{chicken}$ and Ba_{egg} values.

These calculations are limited by the assumptions that:

- contaminants bioconcentrate in the fat tissues; and

- effects from differences in metabolism, digestive system, or feeding characteristics between beef cattle and chickens are minimal.

Please note that the scenario of principal concern for chicken and egg contamination is for home grown chickens. The raising of home grown chickens would be characteristic of free range and semi-free range housing conditions where poultry do come in contact with soil, and possibly vegetation, insects, and benthic organisms. The applicability of this scenario to commercial poultry operations characterized by housing conditions that do not provide chickens access to soil would need to be assessed on a case-by-case basis.

<u>Metals</u> $Ba_{chicken}$ and Ba_{egg} values for all metals except cadmium, selenium, and zinc, weren't available in the literature. For cadmium, selenium, and zinc, U.S. EPA (1995a) cited *Ba* values that were derived by dividing uptake slopes [(g COPC/kg DW tissue)/(g COPC/kg DW feed)], obtained from U.S. EPA (1992b), by a daily consumption rate of 0.2 kilograms DW per day by chicken. To obtain values listed in the HHRAP database, we converted the dry weight *Ba* value to a fresh weight value by assuming a moisture content of 75 percent in eggs and chicken (U.S. EPA 1997c; and Pennington 1994).

Mercury The HHRAP assumes that elemental mercury neither deposits onto soils nor transfers to the aboveground plant parts or grains. Therefore, there's no transfer of elemental mercury into animal tissue. Therefore, we recommend *Ba* values of zero for elemental mercury. This is based on the assumptions made regarding speciation and fate and transport of mercury from emission sources.

If field data suggests otherwise, $Ba_{chicken}$ and Ba_{egg} values for elemental mercury can be derived from the uptake slope factors provided in U.S. EPA (1992b) and U.S. EPA (1995a), using the same consumption rates discussed earlier for metals like cadmium, selenium, and zinc.

We derived $Ba_{chicken}$ and Ba_{egg} values for mercuric chloride and methyl mercury from data in U.S. EPA (1997b). U.S. EPA (1997b) provided $Ba_{chicken}$ and Ba_{egg} values for mercury, but did not specify for which form of mercury. As in U.S. EPA (1997b), the HHRAP assumes mercury speciates into 87 percent divalent mercury and 13 percent methyl mercury in herbivore animal tissue. Also, assuming that the $Ba_{chicken}$ and Ba_{egg} values provided in U.S. EPA (1997b) were for total mercury in animal tissue, then biotransfer factors in U.S. EPA (1997b) can be apportioned in the fractions it is assumed to be found in animal tissue.

Therefore, we based the values reported in the HHRAP database on the following:

- We converted the U.S. EPA (1997b) $Ba_{chicken}$ value of 0.11 day/kg DW for mercury to a fresh weight basis assuming a 75 percent moisture content in chicken (U.S. EPA 1997c; and Pennington 1994). We multiplied the resulting $Ba_{chicken}$ (fresh weight) value by (1) 0.13 to obtain a value for methyl mercury, and (2) 0.87 to obtain a value for mercuric chloride (divalent mercury).

- We converted the U.S. EPA (1997b) Ba_{egg} value of 0.11 day/kg DW for mercury to a fresh weight basis assuming a 75 percent moisture content in eggs (U.S. EPA 1997c; and Pennington 1994). We multiplied the resulting Ba_{egg} (fresh weight) value by (1) 0.13 to obtain a value for methyl mercury, and (2) 0.87 to obtain a value for mercuric chloride (divalent mercury).

A2-2.13.4 Bioconcentration and Bioaccumulation Factors for Fish

Bioconcentration and bioaccumulation factors for fish are used for various compounds, depending on the K_{ow} value of the compound. We recommend using bioconcentration factors for fish (BCF_{fish}) for compounds (except PCDDs, PCDFs, and PCBs) with a log K_{ow} value less than 4.0; and for metals (except lead and mercury). We recommend using bioaccumulation factors for fish (BAF_{fish}) for compounds (except PCDDs, PCDFs, and PCBs) with a log K_{ow} value greater than 4.0, lead, and mercuric compounds. We recommend also using biota-sediment accumulation factors for fish ($BSAF_{fish}$) for PCDDs, PCDFs, and PCBs.

A2-2.13.4.1 Bioconcentration Factors for Fish (BCF_{fish})

BCF_{fish} is the ratio of the contaminant concentration in fish to the contaminant concentration in the water column where the fish is exposed. It accounts for uptake of contaminants by fish from water passing across the gills. We derived *BCF* values for fish for all organic compounds with a log K_{ow} of less than 4.0 (cutoff value with BAF_{fish}) and for all metals, except lead and mercury, as cited in U.S. EPA (1995b). This implies that the concentration of contaminant in the fish is only due to water intake by the fish, and compounds with a log K_{ow} of less than 4.0 are assumed not to significantly bioaccumulate.

It is generally assumed that field measured BCFs are based on total (dissolved and suspended) water column concentrations, and laboratory measured BCFs are based on dissolved water column concentrations. This distinction is important for compounds with a log K_{ow} of greater than or equal to 4.0, because significant amounts of a contaminant can partition into the suspended sediment organic carbon (or particulate phase) of the water column. For compounds with a log K_{ow} of less than 4.0, most of the contaminant is associated with the dissolved phase of the water column and negligible amounts of the contaminant is associated with the suspended sediment phase in the water column. Therefore, for compounds with a log K_{ow} of less than 4.0, BCF values based on dissolved contaminant water concentrations in the water column are essentially the same as BCF values based on total (dissolved + suspended) contaminant water concentrations in the water column.

Our recommended BCF values don't recognize differences in total versus dissolved water concentrations when calculating fish concentrations from BCF_{fish} values for compounds with a log K_{ow} of less than 4.0. Since, dissolved water concentrations is the major contributing factor from compounds with a log K_{ow} of less than 4.0, all BCF_{fish} values (regardless of whether they were derived using total or dissolved water concentrations) can be multiplied by the contaminant concentration in the dissolved water column (C_{dw}) to calculate fish concentrations. This assumption is necessary because (1) literature data is often unclear if the water concentrations are dissolved or total concentrations, and (2) most of the literature reviewed indicated that laboratory experiments were conducted using filtered or distilled water; or the experiments were conducted using fresh water, but were filtered before analyses for water concentrations.

For most compounds, we calculated *BCF* values using best fit log K_{ow} correlation equations derived by Meylan et al. (1999):

- Meylan, W., P.H. Howard, R.S. Boethling, D. Aronson, H. Printup, and S. Gouchie. 1999. "Improved Method for Estimating Bioconcentration/Bioaccumulation Factor from Octanol/Water Partition Coefficient". *Environmental Toxicology and Chemistry*. 18(4):664-672.

Meylan et al. (1999) collected information on measured BCF and other key experimental details for 694 chemicals. Log *BCF* was then regressed against log K_{ow} , and chemicals with significant deviations from the line of best fit were analyzed by chemical structure. The resulting algorithm classifies a substance as either nonionic or ionic, the latter group including carboxylic acids, sulfonic acids and their salts, and quaternary N compounds. Log *BCF* for nonionics was estimated from log K_{ow} and a series of correction factors if applicable; different equations apply for log K_{ow} 1.0 to 7.0 and >7.0 (Meylan et al. 1999). These equations are as follows:

For log K_{ow} < 1 :

$$\log BCF = 0.50 \qquad \text{Equation A-2-27}$$

For log K_{ow} 1 to 7:

$$\log BCF = 0.77 \log K_{ow} - 0.70 + \Sigma \, \textit{Correction Factors} \qquad \text{Equation A-2-28}$$

For log K_{ow} > 7:

$$\log BCF = -1.37 \log K_{ow} + 14.4 + \Sigma \, \textit{Correction Factors} \qquad \text{Equation A-2-29}$$

For log K_{ow} >10.5:

$$\log BCF = 0.50 \qquad \text{Equation A-2-30}$$

For ionic compounds (carboxylic acids, sulfonic acids and salts, compounds with N of +5 valence), were categorized by log K_{ow}, and a log BCF in the range 0.5 to 1.75 was assigned as follows:

For log $K_{ow} < 5$:

$$\log BCF = 0.50 \qquad \text{Equation A-2-31}$$

For log K_{ow} *5 to 6*:

$$\log BCF = 0.75 \qquad \text{Equation A-2-32}$$

For log K_{ow} *6 to 7*:

$$\log BCF = 1.75 \qquad \text{Equation A-2-33}$$

For log K_{ow} *7 to 9*:

$$\log BCF = 1.00 \qquad \text{Equation A-2-34}$$

For log $K_{ow} > 9$:

$$\log BCF = 0.50 \qquad \text{Equation A-2-35}$$

We assume that BCF_{fish} values calculated using the above correlation equations were (1) based on dissolved water concentrations, and (2) not lipid-normalized.

Metals We obtained measured BCF_{fish} values for metals (except lead and mercury) from various literature studies, as cited in U.S. EPA (1999a). Values not available in U.S. EPA (1999a) we estimated using the correlation equations above. BCF_{fish} values obtained from U.S. EPA (1999a) were characterized as follows:

- Field-measured or laboratory-measured values from various experimental studies were evaluated by U.S. EPA (1999a). This information is summarized in the following document:

  > U.S. EPA. 1999a. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Peer Review Draft. August.

  Field-measured data is only (1) available for a limited number of compounds, and (2) based on a single study. In such cases, the field-measured value or the geometric mean of field-measured values were compared with the geometric mean of laboratory-measured values, and the higher one used. A detailed discussion on sources of *BCF* values and methodology followed were provided in Appendix C of U.S. EPA (1999a).

Measured values from various experimental studies were evaluated by U.S. EPA (1999a). Detailed discussion and sources of measured values were provided in U.S. EPA (1999a). For lead and some mercury compounds, a *BAF* was determined to be more applicable than a *BCF*, since those compounds tends to bioaccumulate.

Mercury As in U.S. EPA (1997b), elemental mercury isn't expected to deposit significantly onto soils and surface water. Therefore, it's assumed that there is no transfer of elemental mercury into fish. Fish are assumed to be exposed only to the divalent and organic forms of mercury that exist in the water and soil/sediment media. However, as with the conservative approach adopted by previous guidance documents, all of mercury in the fish is assumed to exist or be converted to the methyl mercury (organic) form after uptake into the fish tissue. Therefore, *BCF* and *BAF* values are not considered applicable for divalent mercury (mercuric chloride). For methyl mercury, a *BAF* was determined to be more applicable than a *BCF*, since mercury tends to bioaccumulate.

A2-2.13.4.2 Bioaccumulation Factors for Fish (BAF_{fish})

BAF_{fish} is the ratio of the contaminant concentration in fish to the contaminant concentration in the water body where the fish are exposed. The BAF_{fish} accounts for uptake of contaminants by fish from water and sediments passing across the gills, and from consumption of various foods including plankton, daphnids, and other fish.

For compounds with a log K_{ow} of greater than or equal to 4.0, contaminants can significantly partition into the suspended sediment organic carbon (or particulate phase) of the water column. Therefore, *BAF* values should generally be based on total (dissolved and suspended) water column concentrations. *BAF* values for fish reported in the HHRAP database were used for organic compounds (except PCBs, PCDDs, and PCDFs) with a log K_{ow} greater than 4.0, lead and some mercuric compounds.

In general we assumed that:

- Field-measured BAFs were based on total (dissolved and suspended) water column concentrations;
- Laboratory-measured BCFs, and therefore the BAFs predicted from them, were based on dissolved water column concentrations; and
- BCFs estimated using correlation Equations A-2-24 through A-2-32, and therefore the BAFs predicted from them, were based on dissolved water column concentrations.

In addition, we assumed that field-measured BCFs for compounds with a log K_{ow} greater than 4.0 were equal to BAFs, because the tissue concentrations are a result of uptake of water (dissolved and suspended), sediment, and various trophic level food.

For most organic compounds (except PCBs, PCDDs and PCDFs) with a log K_{ow} greater than 4.0, the FCM, which accounts for accumulation through the food chain in addition to water, becomes greater than 1. Therefore, a BAF_{fish}, which takes the food chain into consideration, is more appropriate than a BCF_{fish} (U.S. EPA 1995c; 1998). As in U.S. EPA (1995c; 1998), we assumed that *BAF* values for inorganic chemicals equal *BCFs* (i.e., the *FCM* is 1.0), unless chemical specific biomagnification data support using a *FCM* other than 1.0.

For consistency, we adjusted all field-measured BAF (or BCF) values using the methodology suggested in U.S. EPA (1998), to include only the dissolved water column fractions; (i.e., the BAFs based on total water concentrations were converted to BAFs based on dissolved water concentrations). This was done so that all BAF_{fish} values (based on dissolved water concentrations) can be multiplied by the contaminant concentration in the dissolved water column (C_{dw}) to calculate fish concentrations.

We used the following equation cited in U.S. EPA (1998) to convert the *BAF* based on total water concentrations to a *BAF* based on dissolved water concentrations:

$$f_{fd} = \frac{1}{1 + \frac{(DOC)\ (K_{ow})}{10} + (POC)\ (K_{ow})} \qquad \text{Equation A-2-36}$$

where

f_{fd} = fraction of COPC that is freely dissolved in water
DOC = concentration of dissolved organic carbon, kg organic carbon / L water
POC = concentration of particulate organic carbon, kg organic carbon / L water

Since, the Gobas (1993) model was derived from a study conducted at Lake Ontario, DOC and POC values for Lake Ontario were used. Values cited in U.S. EPA (1998) were:

DOC = 2×10^{-6} kg/L
POC = 7.5×10^{-9} kg/L

A *BAF* based on dissolved water concentrations can be calculated from a *BAF* based on total water concentrations as follows:

$$BAF\ (dissolved) = \frac{BAF\ (total)}{f_{fd}} - 1 \qquad \text{Equation A-2-37}$$

Metals (lead) For lead, the food-chain multiplier becomes greater than 1; therefore, a BAF is more appropriate. We obtained the BAF_{fish} value reported in the HHRAP database for lead as a geometric mean from various literature sources described in U.S. EPA (1999a). Since we recommend assuming that metals are insoluble under neutral conditions, the dissolved and total water concentrations are almost equal. However, for consistency, we adjusted the BAF_{fish} value for lead for dissolved fractions.

Mercuric Compounds As in U.S. EPA (1997b), we don't expect elemental mercury to deposit significantly onto soils and surface water. Therefore, there it is assumed that there is no transfer of elemental mercury into fish. Fish are assumed to be exposed only to the divalent and organic forms of mercury that exists in the water and soil/sediment media. However, consistent with the conservative approach adopted by previous guidance documents, all of mercury in the fish is assumed to exist or be converted to the methyl mercury (organic) form after uptake into the fish tissue. Therefore, we obtained the BAF_{fish} value for methyl mercury from U.S. EPA (1997b) for a trophic level 4 fish.

A2-2.13.4.3 Biota-Sediment Accumulation Factor for Fish ($BSAF_{fish}$)

PCDDs, PCDFs, and PCBs $BSAF_{fish}$ accounts for the transfer of contaminants from the bottom sediment to the lipid in fish. U.S. EPA (2000) and (1993d) recommended using $BSAF_{fish}$ values for dioxin-like compounds, including PCBs, because of their lipophilic nature. U.S. EPA (1995b) also stated that $BSAF_{fish}$ values (1) were used for 2,3,7,8-TCDD and PCBs to estimate protective sediment concentrations instead of surface water concentrations, and (2) were a more reliable measure of bioaccumulation potential because of the analytical difficulties in measuring dissolved concentrations in surface water. Therefore, for PCDDs, PCDFs, and PCBs, The HHRAP database includes $BSAF_{fish}$ values instead of BAFs for fish. We obtained the $BSAF_{fish}$ values from U.S. EPA (2000).

U.S. EPA (2000) provided exposure scenarios for conducting site-specific assessments to dioxin-like compounds. For each scenario, U.S. EPA (2000) recommended the use of the following $BSAF_{fish}$ values based on the amount of chlorination of the PCDD or PCDF:

- For TetraCDDs and TetraCDFs, $BSAF_{fish} = 9.0 \times 10^{-02}$
- For PentaCDDs and PentaCDFs, $BSAF_{fish} = 9.0 \times 10^{-02}$

C For HexaCDDs and HexaCDFs, $BSAF_{fish}$ = 4.0×10^{-02}

C For HeptaCDDs and HeptaCDFs, $BSAF_{fish}$ = 5.0×10^{-03}

C For OctaCDDs and OctaCDFs, $BSAF_{fish}$ = 1.0×10^{-04}

Homologue group $BSAF_{fish}$ values obtained from U.S. EPA (2000) were either measured or estimated values that were based on a whole fish lipid content of 7 percent and an organic carbon content of 3 percent.

A2-3 TOXICITY BENCHMARKS

The following sections discuss carcinogenic and noncarcinogenic toxicity benchmarks of compounds. The toxicity information provided in the HHRAP is for informational purposes to help permitting authorities explain the basis for selecting contaminants of concern. Since toxicity benchmarks and slope factors may change as additional toxicity research is conducted, we highly recommend consulting with the most current versions of the sources listed below before completing a risk assessment, to ensure that the toxicity data used in the risk assessment is based upon the most current Agency consensus.

A2-3.1 Prioritization of Data Sources for Chronic Toxicity Benchmarks

In December 2003, the Office of Superfund Remediation and Technology Innovation distributed OSWER Directive 9285.7-53, (U.S. EPA 2004a) updating the hierachy of sources of human health toxicity values originally distributed in Risk Assessment Guidance for Superfund Volume I, Part A, Human Health Evaluation Manual (RAGS) (U.S. EPA 1989). We recommend using the same heirarchy of sources for acquiring human health toxicity data to be used in performing risk assessments of hazardous waste combustion facilities.

We gathered the following parameter values for the database:

- Chronic Oral Toxicity Benchmarks:
  - Oral Reference Doses (*RfD*s) for effects other than cancer (non-cancer)
  - Oral Cancer Slope Factors (*CSF*s) for cancer

- Chronic Inhalation Toxicity Benchmarks:
  - Inhalation Reference Concentrations (*RfC*s) for effects other than cancer (non-cancer)
  - Unit Risk Factors (*URF*s) for cancer

As with other Agency applications (i.e., U.S. EPA Superfund, Resource Conservation and Recovery Act [RCRA] , and Air Toxics), toxicity benchmarks available in the U.S. EPA Integrated Risk Information System (IRIS) (U.S. EPA 2005) were given first priority. For contaminants lacking current IRIS assessments, we obtained toxicity benchmark values from one of the following data sources:

- Provisional Peer-Reviewed Toxicity Values (PPRTVs)

- Other Peer Reviewed Values:

- California Environmental Protection Agency (CalEPA) chronic Reference Exposure Levels (RELs) and Unit Risk Estimate (UREs)

- U.S. Agency for Toxic Substances and Disease Registry (ATSDR) chronic Minimum Risk Levels (MRLs)

- U.S. EPA. 1997a. Health Effects Assessment Summary Tables (HEAST) and older health effects assessment documents not incorporated into HEAST - U.S. EPA Office of Research and Development (ORD)-National Center for Environmental Assessment (EPA-NCEA)

Dose-response assessments that have achieved full intra-agency consensus are incorporated in IRIS, which is regularly updated and available on-line (U.S. EPA 2005). As of 1996, benchmark values placed on IRIS have undergone external peer review and Agency consensus review. Both the U.S. EPA Superfund and RCRA programs accept the primacy of human health toxicity values contained in U.S. EPA's IRIS. The primacy of IRIS toxicity values is also identified in Risk Assessment Guidance for Superfund (RAGS) hierarchy (U.S. EPA 1989).

A Provisional Peer-Reviewed Toxicity Value (PPRTV) is a toxicity value derived for use in the U.S. EPA Superfund Program when such value is not available in U.S. EPA's IRIS (PPRTVs currently represent the second tier of human health toxicity values for the U.S. EPA Superfund and RCRA hazardous waste programs). PPRTVs are derived after a review of the relevant scientific literature using the methods, sources of data, and guidance for value derivation used by the U.S. EPA IRIS Program. All provisional toxicity values receive internal review by two EPA scientists and external peer review by at least two scientific experts. A third scientific review is performed if there is a conflict between the two original external reviewers. PPRTVs differ in part from IRIS values in that PPRTVs do not receive the multi-program consensus review provided for IRIS values. This is because IRIS values are generally intended to be used in all U.S. EPA programs, while PPRTVs are developed specifically for the Superfund Program. In general, the need for a PPRTV is eliminated once an analogous IRIS value becomes available. Once IRIS values become available, PPRTVs are generally removed from the PPRTV database.

U.S. EPA ORD is concerned that PPRTV’s may be seen (and used) as equivalent to IRIS values, and has restricted ***direct*** access to the PPRTV values database to only EPA employees. EPA employees can access the PPRTV database directly via the EPA intranet web site. All others wishing to gain access to PPRTV values need to register as approved users.

If you are on an approved or registered user list, and contact the EPA Center which derives PPRTVs they would email PPRTV assessments you request to you. Plus, if you are on this list, they would periodically email you a list of what contaminants are in the PPRTV database. If you think that you would like to be on this list, please contact a risk assessor or toxicologist in an EPA Superfund or hazardous waste program in the EPA Regional Office containing sites you are working on or expect to work on. You may also apply to be put on the list by sending an email, with “PPRTV Approved User Application” in the subject line, to HHRAPFeedback@EPA.GOV. Please include the following information in the body of the email:

- Your name
- Your affiliation (e.g. company or regulatory agency you work for, or civic/public organization, or other entity interested in a risk assessment using PPRTVs)
- Email address (if it’s different than the address form which you send the application)

As described on the U.S. EPA ORD Air Toxics web site (http://www.epa.gov/ttn/atw), the California Environmental Protection Agency (CalEPA) Office of Environmental Health Hazard Assessment has developed dose-response assessments for many substances, based both on carcinogenicity and health effects other than cancer. The process for developing these assessments is similar to that used by U.S. EPA to develop IRIS values and incorporates significant external scientific peer review. The non-cancer information includes available inhalation health risk guidance values expressed as chronic inhalation and oral reference exposure levels (RELs). CalEPA defines the REL as a concentration level at (or below) which no health effects are anticipated, a concept that is substantially similar to U.S. EPA's non-cancer dose-response assessment perspective. The HHRAP database shows the chronic REL when no IRIS RfC/RfD or PPRTV exists.

CalEPA's quantitative dose-response information on carcinogenicity by inhalation exposure is expressed in terms of the URE, defined similarly to EPA's URE. The HHRAP database shows specific CalEPA UREs where no IRIS or PPRTV values exist. CalEPA's dose response assessments for carcinogens and noncarcinogens are available on-line.

As with U.S. EPA ORD Air Toxics, ATSDR chronic MRL values were listed in the HHRAP database for assessing noncancer effects. As described on the U.S. EPA ORD Air Toxics web site (http://www.epa.gov/ttn/atw), U.S. Agency for Toxic Substances and Disease Registry (ATSDR) develops and publishes Minimum Risk Levels (MRLs) for many toxic substances. The MRL is defined as an estimate of daily human exposure to a substance that is likely to be without an appreciable risk of adverse effects (other than cancer) over a specified duration of exposure. MRLs can be derived for chronic exposures by the inhalation and oral routes. ATSDR describes MRLs as substance-specific estimates to be used by health assessors to select environmental contaminants for further evaluation. MRLs are presented with only 1 significant figure and are considered to be levels below which contaminants are unlikely to pose a health threat. Exposures above an MRL do not necessarily represent a threat, and MRLs are therefore not intended for use as predictors of adverse health effects or for setting cleanup levels. ATSDR publishes MRLs as part of pollutant-specific toxicological profile documents, and also in a regularly updated on-line table.

Health Effects Assessment Summary Tables (HEAST) (U.S. EPA. 1997a) consist of dose-response assessments for some substances that were prepared by the EPA Office of Research and Development (ORD), but not submitted for internal U.S. EPA consensus. For some substances, older health effects assessment documents omitted from HEAST also exist and are generally cited in Table A-1 as ORD-National Center for Environmental Assessment (EPA-NCEA) values. In 1989 and for several years thereafter, HEAST was updated on a quarterly basis. HEAST is no longer updated with new values, but existing values are deleted as revised values become available on IRIS.

A2-3.2 Explanation of Calculated Toxicity Benchmark Values

In the assessment of risk from contaminants, EPA-derived or reviewed toxicity benchmarks values are recommended. However, for numerous compounds, a complete set of inhalation and oral EPA-derived toxicity benchmarks are not available. In such cases, toxicity benchmark values were calculated based on available EPA-derived benchmarks values. For instance, if the oral *RfD* (mg/kg/day) was available and the inhalation *RfC* (mg/m^3) was not; the *RfC* was calculated by dividing the *RfD* by an average human inhalation rate of 20 m^3/day and multiplying by the average human body weight of 70 kg. This conversion is based on a route-to-route extrapolation, which assumes that the toxicity of the given chemical is equivalent over all routes of exposure.

Although conversion based on a route-to-route extrapolation of oral dose-response or inhalation information is not optimal risk assessment practice, the alternative would be to omit these substances altogether from any quantitative risk estimate, and thereby, providing a de facto false negative result of potential risk for the considered route or pathway. Therefore, using route-to-route extrapolation of oral dose-response or inhalation information when applying the HHRAP is preferable when no toxicological benchmark value is available in the peer reviewed data sources listed. However, assumptions and uncertainties involved when using toxicity benchmarks calculated based on route-to-route extrapolation should limit their use to screening-level or priority type risk assessments.

The following methodology was used to calculate missing toxicity benchmark values using available benchmarks that are based on route-to-route extrapolation:

1) *Oral RfD*s presented in the peer reviewed sources cited above were used if available. Missing *Oral RfD*s were calculated from the *RfC* assuming route-to-route extrapolation using the following equation:

$$Oral\ RfD = \frac{RfC \cdot 20\ m^3/d}{70\ kg\ BW} \qquad \text{Equation A-2-38}$$

*RfC*s presented in the peer reviewed sources cited above were used when available. If *RfC*s were not available they were calculated from the *RfD* assuming route-to-route extrapolation and using the inverse of the equation above.

2) For *Inhalation URF*s, values were obtained from the peer reviewed sources cited above. If the *Inhalation URF*s were not available they were calculated from *Oral CSF*, using the following equation:

$$Inhal.\ URF = \frac{Oral\ CSF \cdot 20\ m^3/d}{70\ kg \times 1000\ \mu g/mg} \qquad \text{Equation A-2-39}$$

*Oral CSF*s presented in the peer reviewed sources cited above were used when available. Missing *Oral CSFs* were calculated from *Inhalation URF* values assuming route-to-route extrapolation and using the inverse of the equation above.

A2-3.3 Uncertainties Involved when using Toxicity Benchmarks Calculated based on Route-to-Route Extrapolation

The conversion based on a route-to-route extrapolation, which assumes that the toxicity of the given chemical is equivalent over all routes of exposure, does introduce uncertainty into the assessment. By using this method, it is assumed that the qualitative data supporting the benchmark value for a certain route also applies to the route in question. For example, if an *RfD* is available and the *RfC* is calculated from that value, the assumption is made that the toxicity seen following oral exposure will be equivalent to toxicity following inhalation exposure. This assumption could overestimate or underestimate the toxicity of the given chemical following inhalation exposure.

Additional information regarding what to consider in the evaluation of route-to-route extrapolations can be found in the U.S. EPA (1994c) document *Methods for Derivation of Inhalation Reference Concentrations and Application of Inhalation Dosimetry*.

A2-3.4 Prioritization of Data Sources for Acute Toxicity Benchmarks

Acute inhalation exposure guidelines and criteria are (1) designed to protect a variety of exposure groups including occupational workers, military personnel, and the general public, (2) based on varying exposure durations up to 24 hours in length, and (3) intended to protect against a variety of toxicity endpoints ranging from discomfort or mild adverse health effects to serious, debilitating, and potentially life-threatening effects, up to and including death.

Because of the daily operations of most emission sources and the potential for upset conditions to sometimes occur during operations, acute values developed to include intermittent exposures were considered more appropriate and more protective than values which are based on the strict assumption that acute exposures will be one-time only. Therefore, we recommend the Acute Reference Exposure Levels (Acute REL) developed by Cal/EPA, as the first choice for acute inhalation values (Cal/EPA 1999). Additional acute reference values which also do not exclude intermittent exposures are anticipated to be available from the EPA Office of Research and Development. When available, we recommend using those values (referred to as Acute Reference Concentrations [A-RfCs]) as the first choice, with the Cal/EPA acute RELs second in the hierarchy. Please note that the values we recommend using from all of the sources are based on one-hour exposures.

1. **Cal/EPA Acute RELs** – an acute REL represents the concentration in air at or below which no adverse health effects are anticipated in the general population, including sensitive individuals, for a specified exposure period (Cal/EPA 1999)
2. **Acute inhalation exposure guidelines (AEGL-1)** – an AEGL-1 value represents “the airborne concentration of a substance above which it is predicted that the general population, including susceptible individuals, could experience notable discomfort, irritation, or certain asymptomatic nonsensory effects. However, the effects are not disabling and are transient and reversible upon cessation of exposure.” (NAC 2002)

3. **Level 1 emergency planning guidelines (ERPG-1)** – an ERPG-1 value represents “the maximum concentration in air below which it is believed nearly all individuals could be exposed for up to one hour without experiencing other than mild transient adverse health effects or perceiving a clearly defined objectionable odor.” (AIHA 2004; SCAPA 2004)

4. **Temporary emergency exposure limits (TEEL-1)** – a TEEL-1 value represents “the maximum concentration in air below which it is believed nearly all individuals could be exposed without experiencing other than mild transient adverse health effects or perceiving a clearly defined odor.” (SCAPA 2004)

It should be noted that the TEEL-1 values (SCAPA 2004) are calculated based on an assumed exposure period of 15-minutes. To evaluate risks due to acute exposure based on the highest 1-hour average air concentrations, the TEEL-1 values were extrapolated from a 15-minute to a 1-hour exposure basis using a modification to Haber’s Rule developed by ten Berge (1986) and used by Cal/EPA to develop acute RELs (Cal/EPA 1999), as shown below.

$$C^n \times T = K$$ Equation A-2-40

where
C = Concentration (mg/m^3)
n = Constant greater than zero (unitless)
T = Time of exposure (hour)
K = Constant level or severity of response (unitless)

Where available, chemical-specific values for the parameter *n* as selected and recommended by Cal/EPA's Office of Environmental Health Hazard Assessment (OEHHA) based on the work of ten Berge (1986) and the open literature were used to make the extrapolations (Cal/EPA 1999). For chemicals for which a chemical-specific value of n was not available, extrapolations were made using a value of n = 1, as recommended by OEHHA, because the extrapolations were all based on an initial exposure period (15-minutes) of less than 1 hour duration (Cal/EPA 1999).

Use of the modified form of Haber's Rule allows contributions by both concentration and time to the overall severity of effect to be considered. However, the extrapolated air concentrations are to be interpreted carefully, and should not be interpreted to be absolutes. For example, chemical-specific values of the exponent n are sometimes based on a relatively limited set of dose-response data. Also, the majority of extrapolated TEEL-1 values were calculated using default exponent values and, therefore, are likely to be even less certain than exponent values based on limited data sets.

The hierarchy is presented in order of preference, from 1 (most preferred) to 4 (least preferred). If no acute REL value is available for a given contaminant, one can work down the list in order. If no AEGL-1 value is available, but an AEGL-2 value is available, the AEGL-2 can be selected as the AIEC only if it is a more protective value (lower in concentration) than an ERPG-1 or a TEEL-1 value if either of these values is available.

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CHEMICALS OF POTENTIAL CONCERN INCLUDED IN THE COMPANION DATABASE

CAS#	Compound
83-32-9	Acenaphthene
75-07-0	Acetaldehyde
67-64-1	Acetone
75-05-8	Acetonitrile
98-86-2	Acetophenone
107-02-8	Acrolein
107-13-1	Acrylonitrile
309-00-2	Aldrin
62-53-3	Aniline
120-12-7	Anthracene
7440-36-0	Antimony compounds
12674-11-2	Aroclor 1016
11097-69-1	Aroclor 1254
7440-38-2	Arsenic Compounds
1912-24-9	Atrazine
7440-39-3	Barium
100-52-7	Benzaldehyde
71-43-2	Benzene
56-55-3	Benzo(a)anthracene
50-32-8	Benzo(a)pyrene
205-99-2	Benzo(b)fluoranthene
207-08-9	Benzo(k)fluoranthene
65-85-0	Benzoic acid
100-47-0	Benzonitrile
100-51-6	Benzyl alcohol
100-44-7	Benzyl chloride
7440-41-7	Beryllium compounds
319-84-6	BHC, alpha- (alpha-Hexachlorocyclohexane)
319-85-7	BHC, beta- (beta-Hexachlorocyclohexane)
111-44-4	Bis(2-chlorethyl)ether
75-27-4	Bromodichloromethane
75-25-2	Bromoform (Tribromomethane)
101-55-3	Bromophenyl-phenylether, 4-
7440-43-9	Cadmium compounds
75-15-0	Carbon disulfide
56-23-5	Carbon tetrachloride
57-74-9	Chlordane
7782-50-5	Chlorine
59-50-7	Chloro-3-methylphenol, 4-
106-47-8	Chloroaniline, p-
108-90-7	Chlorobenzene

CAS#	Compound
510-15-6	Chlorobenzilate
75-45-6	Chlorodifluoromethane
75-00-3	Chloroethane
67-66-3	Chloroform (Trichloromethane)
39638-32-9	Chloroisopropyl ether, bis-1,2-
91-58-7	Chloronaphthalene,2-
95-57-8	Chlorophenol, 2-
7005-72-3	Chlorophenyl-phenylether, 4-
2921-88-2	Chlorpyrifos
7440-47-3	Chromium compounds
18540-29-9	Chromium, hexavalent
218-01-9	Chrysene
108-39-4	Cresol, m- (3-Methylphenol)
95-48-7	Cresol, o-
106-44-5	Cresol, p-
98-82-8	Cumene (Isopropylbenzene)
57-12-5	Cyanide Compounds
72-54-8	DDD, 4,4'-
72-55-9	DDE, 4,4'-
50-29-3	DDT, 4-4'-
333-41-5	Diazinon
53-70-3	Dibenz(a,h)anthracene
96-12-8	Dibromo-3-chloropropane, 1,2-
124-48-1	Dibromochloromethane
95-50-1	Dichlorobenzene, 1,2-
541-73-1	Dichlorobenzene, 1,3-
106-46-7	Dichlorobenzene,1,4-
91-94-1	Dichlorobenzidine, 3,3'-
75-71-8	Dichlorodifluoromethane
75-34-3	Dichloroethane 1,1-
107-06-2	Dichloroethane, 1,2- (Ethylene Dichloride)
75-35-4	Dichloroethylene 1,1-
156-59-2	Dichloroethylene, cis-1,2-
156-60-5	Dichloroethylene-1,2 (trans)
120-83-2	Dichlorophenol, 2,4-
78-87-5	Dichloropropane, 1,2-
542-75-6	Dichloropropene, 1,3-
62-73-7	Dichlorvos
60-57-1	Dieldrin
84-66-2	Diethyl phthalate
131-11-3	Dimethyl phthalate
105-67-9	Dimethylphenol, 2,4-
119-90-4	Dimethyoxybenzidine, 3,3'-

CHEMICALS OF POTENTIAL CONCERN INCLUDED IN THE COMPANION DATABASE

CAS#	Compound
84-74-2	Di-n-butyl phthalate
99-65-0	Dinitrobenzene, 1,3-
51-28-5	Dinitrophenol, 2,4-
121-14-2	Dinitrotoluene, 2,4-
606-20-2	Dinitrotoluene, 2,6-
117-84-0	Di-n-octylphthalate
123-91-1	Dioxane, 1,4-
646-06-0	DIOXOLANE,1,3-
122-66-7	Diphenylhydrazine,1,2-
298-04-4	Disulfoton
115-29-7	Endosulfan I
72-20-8	Endrin
106-89-8	Epichlorohydrin (1-Chloro-2,3- epoxypropane)
97-63-2	Ethyl methacrylate
62-50-0	Ethyl methanesulfonate
100-41-4	Ethylbenzene
106-93-4	Ethylene Dibromide
75-21-8	Ethylene oxide
117-81-7	Ethylhexyl phthalate, bis-2-
206-44-0	Fluoranthene
86-73-7	Fluorene
50-00-0	Formaldehyde
64-18-6	Formic acid
35822-46-9	HeptaCDD, 1,2,3,4,6,7,8-
67562-39-4	HeptaCDF, 1,2,3,4,6,7,8-
55673-89-7	HeptaCDF, 1,2,3,4,7,8,9-
76-44-8	Heptachlor
1024-57-3	Heptachlor epoxide
39227-28-6	HexaCDD, 1,2,3,4,7,8-
57653-85-7	HexaCDD, 1,2,3,6,7,8-
19408-74-3	HexaCDD, 1,2,3,7,8,9-
70648-26-9	HexaCDF, 1,2,3,4,7,8-
57117-44-9	HexaCDF, 1,2,3,6,7,8-
72918-21-9	HexaCDF, 1,2,3,7,8,9-
60851-34-5	HexaCDF, 2,3,4,6,7,8-
87-68-3	Hexachloro-1,3-butadiene (Perchlorobutadiene)
118-74-1	Hexachlorobenzene
77-47-4	Hexachlorocyclopentadiene
67-72-1	Hexachloroethane (Perchloroethane)
70-30-4	Hexachlorophene
7647-01-0	Hydrogen chloride (Hydrochloric acid)
193-39-5	Indeno(1,2,3-cd) pyrene
78-59-1	Isophorone
7439-92-1	Lead compounds
121-75-5	Malathion
7487-94-7	Mercuric chloride
7439-97-6	Mercury (elemental)
126-98-7	Methacrylonitrile
67-56-1	Methanol
72-43-5	Methoxychlor
79-20-9	Methyl acetate
74-83-9	Methyl bromide (Bromomethane)
74-87-3	Methyl chloride (Chloromethane)
78-93-3	Methyl ethyl ketone (2-Butanone)
108-10-1	Methyl isobutyl ketone
22967-92-6	Methyl mercury
298-00-0	Methyl parathion
74-95-3	Methylene bromide
75-09-2	Methylene chloride
91-20-3	Naphthalene
7440-02-0	Nickel
88-74-4	Nitroaniline, 2-
99-09-2	Nitroaniline, 3-
100-01-6	Nitroaniline, 4-
98-95-3	Nitrobenzene
88-75-5	Nitrophenol, 2-
100-02-7	Nitrophenol, 4-
924-16-3	Nitroso-di-n-butylamine, n-
86-30-6	Nitrosodiphenylamine, n-
621-64-7	Nitrosodipropylamine, n-
3268-87-9	OctaCDD, 1,2,3,4,6,7,8,9-
39001-02-0	OctaCDF, 1,2,3,4,6,7,8,9-
40321-76-4	PentaCDD, 1,2,3,7,8-

CHEMICALS OF POTENTIAL CONCERN INCLUDED IN THE COMPANION DATABASE

CAS#	Compound
57117-41-6	PentaCDF, 1,2,3,7,8-
57117-31-4	PentaCDF, 2,3,4,7,8-
608-93-5	Pentachlorobenzene
82-68-8	Pentachloronitrobenzene (PCNB)
87-86-5	Pentachlorophenol
85-01-8	Phenanthrene
108-95-2	Phenol
298-02-2	Phorate
85-44-9	Phthalic anhydride (1,2-Benzene dicarboxylic anhydride)
23950-58-5	Pronamide
129-00-0	Pyrene
110-86-1	Pyridine
299-84-3	Ronnel
94-59-7	Safrole
7782-49-2	Selenium
7440-22-4	Silver
57-24-9	Strychnine
100-42-5	Styrene
1746-01-6	TetraCDD, 2,3,7,8-
51207-31-9	TetraCDF, 2,3,7,8-
95-94-3	Tetrachlorobenzene, 1,2,4,5-
630-20-6	Tetrachloroethane, 1,1,1,2-
79-34-5	Tetrachloroethane, 1,1,2,2-
127-18-4	Tetrachloroethylene (Perchloroethylene)
58-90-2	Tetrachlorophenol, 2,3,4,6-
109-99-9	Tetrahydrofuran
7440-28-0	Thallium (l)
108-88-3	Toluene
95-53-4	Toluidine, o-
87-61-6	Trichlorobenzene, 1,2,3-
120-82-1	Trichlorobenzene, 1,2,4-
71-55-6	Trichloroethane, 1,1,1-
79-00-5	Trichloroethane, 1,1,2-
79-01-6	Trichloroethylene
75-69-4	Trichlorofluoromethane (Freon 11)
95-95-4	Trichlorophenol, 2,4,5-
88-06-2	Trichlorophenol, 2,4,6-
96-18-4	Trichloropropane, 1,2,3-
108-67-8	Trimethylbenzene, 1,3,5-
99-35-4	Trinitrobenzene, 1,3,5 (sym-)
118-96-7	Trinitrotoluene, 2,4,6-
108-05-4	Vinyl acetate
75-01-4	Vinyl chloride
108-38-3	Xylene, m-
95-47-6	Xylene, o-
106-42-3	Xylene, p-
7440-66-6	Zinc

APPENDIX B

ESTIMATING MEDIA CONCENTRATION EQUATIONS AND VARIABLE VALUES

Table	Equation
	SOIL INGESTION EQUATIONS
B-1-1	Soil Concentration Due to Deposition
B-1-2	COPC Soil Loss Constant
B-1-3	COPC Loss Constant Due to Soil Erosion
B-1-4	COPC Loss Constant Due to Soil Runoff
B-1-5	COPC Loss Constant Due to Soil Leaching
B-1-6	COPC Loss Constant Due to Soil Volatilization
	PRODUCE INGESTION EQUATIONS
B-2-1	Soil Concentration Due to Deposition
B-2-2	COPC Soil Loss Constant
B-2-3	COPC Loss Constant Due to Soil Erosion
B-2-4	COPC Loss Constant Due to Soil Runoff
B-2-5	COPC Loss Constant Due to Soil Leaching
B-2-6	COPC Loss Constant Due to Soil Volatilization
B-2-7	Aboveground Produce Concentration Due to Direct Deposition
B-2-8	Aboveground Produce Concentration Due to Air-to-Plant Transfer
B-2-9	Aboveground Produce Concentration Due to Root Uptake
B-2-10	Belowground Produce Concentration Due to Root Uptake
	ANIMAL PRODUCTS INGESTION EQUATIONS
B-3-1	Soil Concentration Due to Deposition
B-3-2	COPC Soil Loss Constant
B-3-3	COPC Loss Constant Due to Soil Erosion
B-3-4	COPC Loss Constant Due to Soil Runoff
B-3-5	COPC Loss Constant Due to Soil Leaching
B-3-6	COPC Loss Constant Due to Soil Volatilization
B-3-7	Forage and Silage Concentration Due to Direct Deposition
B-3-8	Forage and Silage Concentration Due to Air-to-Plant Transfer
B-3-9	Forage/Silage/Grain Concentration Due to Root Uptake
B-3-10	Beef Concentration Due to Plant & Soil Ingestion
B-3-11	Milk Concentration Due to Plant & Soil Ingestion
B-3-12	Pork Concentration Due to Plant & Soil Ingestion
B-3-13	COPC Concentration in Eggs
B-3-14	Concentration in Chicken
	DRINKING WATER AND FISH INGESTION EQUATIONS
B-4-1	WATERSHED Soil Concentration Due to Deposition
B-4-2	COPC Soil Loss Constant
B-4-3	COPC Loss Constant Due to Soil Erosion
B-4-4	COPC Loss Constant Due to Soil Runoff
B-4-5	COPC Loss Constant Due to Soil Leaching
B-4-6	COPC Loss Constant Due to Soil Volatilization
B-4-7	Total Water Body Load
B-4-8	Deposition to Water Body
B-4-9	Impervious Runoff Load to Water Body
B-4-10	Pervious Runoff Load to Water Body
B-4-11	Erosion Load to Water Body

DRINKING WATER AND FISH INGESTION EQUATIONS (cont’d)

B-4-12 Diffusion Load to Water Body
B-4-13 Universal Soil Loss Equation (USLE)
B-4-14 Sediment Delivery Ratio
B-4-15 Total Water Body Concentration
B-4-16 Fraction in Water Column & Benthic Sediment
B-4-17 Overall Total Water Body Dissipation Rate Constant
B-4-18 Water Column Volatilization Loss Rate Constant
B-4-19 Overall COPC Transfer Rate Coefficient
B-4-20 Liquid Phase Transfer Coefficient
B-4-21 Gas Phase Transfer Coefficient
B-4-22 Benthic Burial Rate Constant
B-4-23 Total Water Column Concentration
B-4-24 Dissolved Phase Water Concentration
B-4-25 COPC Concentration Sorbed to Bed Sediment
B-4-26 Fish Concentration From Bioconcentration Factors Using Dissolved-Phase Water Concentration
B-4-27 Fish Concentration From Bioaccumulation Factors Using Dissolved-Phase Water Concentration
B-4-28 Fish Concentration From Biota-to-Sediment Accumulation Factors Using COPC Sorbed to Bed Sediment

DIRECT INHALATION EQUATION

B-5-1 Air Concentration

ACUTE EQUATION

B-6-1 Acute Air Concentration

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

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Description

Use the equations in this table to calculate an average COPC soil concentration resulting from wet and dry deposition of particles and vapors to soil over the exposure duration. We recommend assuming that COPCs are incorporated only to a finite depth (the soil mixing zone depth, Z_s). Use the COPC soil concentration averaged over the exposure duration, represented by *Cs*, for carcinogenic COPCs, where risk is averaged over the lifetime of an individual. Because the hazard quotient associated with noncarcinogenic COPCs is based on a reference dose rather than a lifetime exposure, we recommend using the highest annual average COPC soil concentration occurring during the exposure duration period for noncarcinogenic COPCs. The highest annual average COPC soil concentration would most likely occur at the end of the time period of combustion and is represented by Cs_{tD}.

The following uncertainties are associated with this variable:

(1) We assume that the time period for deposition of COPCs resulting from hazardous waste combustion is a conservative, long-term value. This assumption may overestimate *Cs* and Cs_{tD}.

(2) Exposure duration values (T_2) are based on historical mobility studies and won't necessarily remain constant. Specifically, mobility studies indicate that most receptors that move remain in the vicinity of the combustion unit; however, it is impossible to accurately predict the probability that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants.

(3) A value of zero for T_1 doesn't account for exposure that may have occurred from historic operations and emissions from hazardous waste combustion. This may underestimate *Cs* and Cs_{tD}.

(4) For soluble COPCs, leaching might lead to movement below the 2 centimeters, resulting in lower concentrations within the mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}.

(5) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This may underestimate *Cs* and Cs_{tD}.

Equation for Carcinogens

Soil Concentration Averaged Over Exposure Duration

$$Cs = \frac{\left(\frac{Ds \cdot tD - Cs_{tD}}{ks}\right) + \left(\frac{Cs_{tD}}{ks} \cdot \left[1 - \exp\left(-ks\left(T_2 - tD\right)\right)\right]\right)}{\left(T_2 - T_1\right)} \quad for\ T_1 < tD < T_2$$

$$Cs = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left(\left[tD + \frac{\exp(-\ ks \cdot tD\ )}{ks}\right] - \left[T_1 + \frac{\exp(-\ ks \cdot T_1)}{ks}\right]\right) \quad for\ T_2 \le tD$$

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 2 of 9)

Equation for Noncarcinogens

Highest Annual Average Soil Concentration

$$Cs_{tD} = \frac{Ds \cdot [1 - \exp(-ks \cdot tD)]}{ks}$$

where

$$Ds = \frac{100 \cdot Q}{Z_s \cdot BD} \cdot [F_v\ (Dydv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)]$$

For mercury modeling

$$Ds_{(Mercury)} = \frac{100 \cdot [0.48Q_{(Total)}]}{Z_s \cdot BD} \cdot [F_{v_{(Hg^{2+})}}\ (Dydv + Dywv) + (Dydp + Dywp) \cdot [1 - F_{v_{(Hg^{2+})}}]$$

Use $0.48Q$ for total mercury and $F_v = 0.85$ in the mercury modeling equation to calculate Ds. Apportion the calculated Ds value into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the assumed 98% Hg^{2+} and 2% MHg speciation split in soils (see Chapter 2). Elemental mercury (Hg^0) occurs in very small amounts in the vapor phase and does not exist in the particle or particle-bound phase. Therefore, assume elemental mercury deposition onto soils is negligible or zero. Evaluate elemental mercury for the direct inhalation pathway only (Table B-5-1).

$Ds_{(Hg2+)} = 0.98\ Ds_{(Mercury)}$
$Ds_{(MHG)} = 0.02\ Ds_{(Mercury)}$
$Ds_{(Hg0)} = 0.0$

Evaluate divalent and methyl mercury as individual COPCs. Calculate Cs for divalent and methyl mercury using the corresponding (1) fate and transport parameters for mercuric chloride (divalent mercury, Hg^{2+}) and methyl mercury provided in Appendix A-2, and (2) Ds (Hg^{2+}) and Ds (MHg) as calculated above.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 3 of 9)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	
Cs_{tD}	Soil concentration at time *tD*	mg COPC/kg soil	
Ds	Deposition term	mg COPC/kg soil-yr	**Varies** U.S. EPA (1994a) and NC DEHNR (1997) recommend incorporating a deposition term into the *Cs* equation. Uncertainties associated with this variable include the following: (1) Five of the variables in the equation for *Ds* (*Q, Cywv, Dywv*, *Dydp*, and *Dywp*) are COPC- and site-specific. Values for these variables are estimated through modeling. The direction and magnitude of any uncertainties shouldn't be generalized. (2) Based on the narrow recommended ranges, we expect uncertainties associated with *Vdv*, F_v, and *BD* to be low. (3) Values for Z_s vary by about one order of magnitude. Uncertainty is greatly reduced if you know whether soils are tilled or untilled.
tD	Time period over which deposition occurs (time period of combustion)	yr	**30** U.S. EPA (1998) suggests that *tD* can be ≥ 30 years. We recommend using 30 years unless site-specific information is available indicating that this assumption is unreasonable (see Chapter 6 of the HHRAP).
ks	COPC soil loss constant due to all processes	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-1-2. The COPC soil loss constant is the sum of all COPC removal processes. Uncertainty associated with this variable includes the following: COPC-specific values for *ksg* (one of the variables in the equation in Table B-1-2) are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 4 of 9)

Variable	Description	Units	Value
T_2	Length of exposure duration	yr	**6, 30, or 40** We recommend reasonable maximum exposure (RME) values for T_2: Exposure Duration — RME — Reference Child Resident, Farmer Child, Fisher Child — 6 years — U.S. EPA (1997b) Adult Resident and Fisher — 30 years — U.S. EPA (1997b) Farmer — 40 years — U.S. EPA (1994b) U.S. EPA (1994c) recommended the following unreferenced values: Exposure Duration — Years Subsistence Farmer — 40 Adult Resident — 30 Subsistence Fisher — 30 Child Resident — 9 Uncertainties associated with this variable include the following: (1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate Cs and Cs_{tD}. (2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources. However, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate Cs and Cs_{tD}.
T_1	Time period at the beginning of combustion	yr	**0** Consistent with U.S. EPA (1994c), we recommend a value of 0 for T_1. The following uncertainty is associated with this variable: A T_1 of 0 does not account for exposure that may have occurred from historical operations or emissions from burning hazardous waste. This may underestimate Cs and Cs_{tD}.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 5 of 9)

Variable	Description	Units	Value
100	Units conversion factor	mg-cm^2/kg-cm^2	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 of the HHRAP for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended *BD* value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 6 of 9)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss this variable and offer COPC-specific values in Appendix A-2. The range is based on the values presented in Appendix A-2. Values are also presented in U.S. EPA (1994c) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) Our F_v calculations assume a default S_T value for background plus local sources, rather than an S_T value for urban sources. If your site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Dydv	Unitized yearly average dry deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dywv	Unitized yearly average wet deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dydp	Unitized yearly average dry deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dywp	Unitized yearly average wet deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

This reference is for the statement that the equation used to calculate the fraction of air concentration in vapor phase (F_v) assumes that the variable c (the Junge constant) is constant for all chemicals. However, Bidleman (1988) notes that the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. The following equation, presented in Bidleman (1988), is cited by U.S. EPA (1994b) and NC DEHNR (1997) for calculating the variable F_v:

where

F_v = Fraction of chemical air concentration in vapor phase (unitless)
c = Junge constant = 1.7 x 10^{-04} (atm-cm)
S_T = Whitby's average surface area of particulates = 3.5 x 10^{-06} cm^2/cm^3 air (corresponds to background plus local sources)
= Liquid-phase vapor pressure of chemical (atm) (see Appendix A-2)

If the chemical is a solid at ambient temperatures, the solid-phase vapor pressure is converted to a liquid-phase vapor pressure as follows:
where

$$\ln \frac{P^{\circ}_L}{P^{\circ}_S} = \frac{\Delta S_f}{R} \cdot \frac{(T_m - T_a)}{T_a}$$

= Solid-phase vapor pressure of chemical (atm) (see Appendix A-2)

= Entropy of fusion over the universal gas constant = 6.79 (unitless)

T_m = Melting point of chemical (K) (see Appendix A-2)
T_a = Ambient air temperature = 284 K (11°C)

Brzuzy, L.P. and R.A. Hites. 1995. "Estimating the Atmospheric Deposition of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans from Soils." *Environmental Science and Technology*. Volume 29. Pages 2090-2098.

This reference presents soil profiles for dioxin measurements.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

Cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, *BD*, of 1.5 g soil/cm^3 soil for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Press, Inc. New York.

Cited by U.S. EPA (1998) for the statement that *BD* is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-1-1. This document also recommends using (1) a deposition term, *Ds*, and (2) COPC-specific F_v values.

Research Triangle Institute (RTI). 1992. *Preliminary Soil Action Level for Superfund Sites.* Draft Interim Report. Prepared for U.S. EPA Hazardous Site Control Division, Remedial Operations Guidance Branch. Arlington, Virginia. EPA Contract 68-W1-0021. Work Assignment No. B-03, Work Assignment Manager Loren Henning. December.

This document is a reference source for COPC-specific F_v values.

U.S. EPA. 1992. *Estimating Exposure to Dioxin-Like Compounds.* Draft Report. Office of Research and Development. Washington, D.C. EPA/600/6-88/005b.

The External Review Draft of the MPE document (the final is U.S. EPA 1998) cites this document as the source of values for soil mixing zone depth, Z_s, for tilled and untilled soils.

U.S. EPA. 1993. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste. Office of Research and Development. Washington, D.C. September.

This document is a reference for the equation in Table B-1-1. It recommends using a deposition term, *Ds*, and COPC-specific F_v values in the *Cs* equation.

U.S. EPA 1994a. *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. April.

This document is a reference for the equation in Table B-1-1; it recommends using the following in the *Cs* equation: (1) a deposition term, *Ds*, and (2) a default soil bulk density value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document recommends values for length of exposure duration, T_2, for the farmer.

TABLE B-1-1

SOIL CONCENTRATION DUE TO DEPOSITION (SOIL INGESTION EQUATIONS)

(Page 9 of 9)

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes*. Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- Values for the length of exposure duration, T_2
- Value of 0 for the time period of the beginning of combustion, T_1
- F_v values that range from 0.27 to 1 for organic COPCs
- Default soil bulk density value of 1.5 g soil/cm^3 soil, based on a mean for loam soil from Carsel et al. (1988)

U.S. EPA. 1997a. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1997b. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002Fc. August.

This document is a reference source for values for length of exposure duration, T_2.

U.S. EPA. 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (MPE).* Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

TABLE B-1-2

COPC SOIL LOSS CONSTANT
(SOIL INGESTION EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC soil loss constant, which accounts for the loss of COPCs from soil by several mechanisms.

Uncertainties associated with this equation include the following:

(1) COPC-specific values for *ksg* are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

(2) The source of the equations in Tables B-1-3 through B-1-5 have not been identified.

Equation

$$ks = ksg + kse + ksr + ksl + ksv$$

Variable	Description	Units	Value
ks	COPC soil loss constant due to all processes	yr^{-1}	
ksg	COPC loss constant due to biotic and abiotic degradation	yr^{-1}	**Varies** This variable is COPC-specific. Values are available in the COPC tables in Appendix A-2. "Degradation rate" values are also presented in NC DEHNR (1997); however, no reference or source is provided for the values. U.S. EPA (1994a) and U.S. EPA (1994b) state that *ksg* values are COPC-specific; however, all *ksg* values are presented as zero (U.S. EPA 1994a) or as "NA" (U.S. EPA 1994b); the basis of these assumptions is not addressed. The following uncertainty is associated with this variable: COPC-specific values for *ksg* are determined empirically from field studies; no information is available on applying these values to the site-specific conditions associated with affected facilities.

TABLE B-1-2

COPC SOIL LOSS CONSTANT
(SOIL INGESTION EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** This variable is COPC- and site-specific, and is further discussed in Table B-1-3. Consistent with U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), we recommend a default value of zero for *kse* because contaminated soil erodes both onto the site and away from the site. Uncertainties associated with this variable include the following: (1) The source of the equation in Table B-1-3 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *kse*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *kse*.
ksr	COPC loss constant due to surface runoff	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-1-4. No reference document is cited for this equation; however, using this equation is consistent with U.S. EPA (1998). U.S. EPA (1994a) assumed that all *ksr* values are zero but didn't explain the basis for this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-1-4) include the following: (1) The source of the equation in Table B-1-4 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *ksr*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksr*.
ksl	COPC loss constant due to leaching	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-1-5. Using this equation is consistent with U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997). U.S. EPA (1994a) assumed that *ksl* is zero but didn't explain the basis of this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-1-5) include the following: (1) The source of the equation in Table B-1-5 has not been identified. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

TABLE B-1-2

COPC SOIL LOSS CONSTANT
(SOIL INGESTION EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-1-6. This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from U.S. EPA (1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986). Uncertainties associated with this equation include the following: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

REFERENCES AND DISCUSSION

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the reference documents for the equations in Tables B-1-4 and B-1-5. This document is also cited as (1) the source for a range of COPC-specific degradation rates (*ksg*), and (2) one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because of contaminated soil eroding both onto the site and away from the site.

U.S. EPA. 1994a. *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.

This document is cited as a source for the assumptions that losses resulting from erosion (*kse*), surface runoff (*ksr*), degradation (*ksg*), leaching (*ksl*), and volatilization (*ksv*) are all zero.

U.S. EPA. 1994b. *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.

This document is one of the reference documents for the equations in Tables B-1-4 and B-1-5. This document is also cited as one of the sources that recommend using the assumption that the loss resulting from erosion (*kse*) is zero and the loss resulting from degradation (*ksg*) is "NA" or zero for all compounds.

TABLE B-1-2

COPC SOIL LOSS CONSTANT (SOIL INGESTION EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference documents for the equations for *ksr*, *ksl*, and *ksv*.

TABLE B-1-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (SOIL INGESTION EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the constant for COPC loss resulting from erosion of soil. Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend a default value of zero for *kse* because of contaminated soil eroding onto the site and away from the site. In site-specific cases where the permitting authority considers it appropriate to calculate a *kse*, we recommend using the equation presented in this table along with associated uncertainties. You can find additional discussion on determining *kse* in U.S. EPA (1998). Uncertainties associated with this equation include:

(1) For soluble COPCs, leaching might lead to movement below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *kse*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) in comparison to that of other residues. This uncertainty may underestimate *kse*.

Equation

$$kse = \frac{0.1 \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s} \left(\frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)} \right)$$

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend that the default value assumed for *kse* is zero because contaminated soil erodes both onto the site and away from the site. uncertainty may overestimate *kse*.
0.1	Units conversion factor	g-kg/cm^2-m^2	
X_e	Unit soil loss	kg/m^2-yr	**Varies** This variable is site-specific and is calculated by using the equation in Table B-4-13. The following uncertainty is associated with this variable: All of the equation variables are site-specific. Using default values rather than site-specific values for any or all of these variables will result in unit soil loss (X_e) estimates that are under- or overestimated to some degree. Based on default values, we expect X_e estimates to vary over a range of less than two orders of magnitude.

TABLE B-1-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (SOIL INGESTION EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
SD	Sediment delivery ratio	unitless	**Varies** This value is site-specific, and is calculated by using the equation in Table B-4-14. Uncertainties associated with this variable include the following: (1) The recommended default values for the empirical intercept coefficient, *a*, are average values based on studies of sediment yields from various watersheds. Therefore, those default values may not accurately represent site-specific watershed conditions. As a result, using these default values may under- or overestimate *SD*. (2) The recommended default value for the empirical slope coefficient, *b*, is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, using the default value may under- or overestimate *SD*.
ER	Soil enrichment ratio	unitless	**Inorganics: 1** **Organics: 3** COPC enrichment occurs because (1) lighter soil particles erode more quickly than heavier soil particles, and (2) concentration of organic COPCs—which is a function of organic carbon content of sorbing media—is expected to be higher in eroded material than in *in-situ* soil (U.S. EPA 1998). In the absence of site-specific data, we recommend a default value of 3 for organic COPCs and 1 for inorganic COPCs. This is consistent with other Agency guidance (1998), which recommends a range of 1 to 5 and a value of 3 as a “reasonable first estimate.” This range has been used for organic matter, phosphorus, and other soil-bound COPCs (U.S. EPA 1998); however, no sources or references were provided for this range. *ER* is generally higher in sandy soils than in silty or loamy soils (U.S. EPA 1998). The following uncertainty is associated with this variable: The default *ER* value may not accurately reflect site-specific conditions; therefore, *kse* may be over- or underestimated to an unknown extent. Using county-specific *ER* values will reduce the extent of any uncertainties.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended *BD* value may not accurately represent site-specific soil conditions. It may under- or overestimate site-specific soil conditions to an unknown degree.

TABLE B-1-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (SOIL INGESTION EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil / Depth (cm) / Reference Untilled / 2 / Brzuzy et al. (1995) Tilled / 20 / U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	ml water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
θ_{sw}	Soil volumetric water content	ml water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *kse* may be under- or overestimated to a small extent, based on the limited range of values.

TABLE B-1-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (SOIL INGESTION EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density, *BD*, value of 1.5 g soil/cm^3 soil for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because contaminated soil erodes both onto the site and away from the site.

U.S. EPA. 1994. *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.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *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.

This document recommends (1) a default soil bulk density value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil that is taken from Carsel et al. (1988), and (2) a default soil volumetric water content, θ_{sw}, value of 0.2 ml water/cm^3 soil.

TABLE B-1-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (SOIL INGESTION EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the source of a range of COPC enrichment ratio, *ER*, values. The recommended range, 1 to 5, was used for organic matter, phosphorous, and other soul-bound COPCs. This document recommends a value of 3 as a "reasonable first estimate," and states that COPC enrichment occurs because lighter soil particles erode more quickly than heavier soil particles. Lighter soil particles have higher ratios of surface area to volume and are higher in organic matter content. Therefore, concentration of organic COPCs, which is a function of the organic carbon content of sorbing media, is expected to be higher in eroded material than in *in situ* soil.

This document is also a source of the following:

- A range of soil volumetric water content (θ_{sw}) values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil
- The equations in Tables B-1-3 and B-1-5.

TABLE B-1-4

COPC LOSS CONSTANT DUE TO RUNOFF (SOIL INGESTION EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant due to runoff of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might result in movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate *ksr*.

Equation

$$ksr = \frac{RO}{\theta_{sw} \cdot Z_s} \cdot \left(\frac{1}{1 + \left(Kd_s \cdot BD / \theta_{sw} \right)} \right)$$

Variable	Description	Units	Value
ksr	COPC loss constant due to runoff	yr^{-1}	
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures for estimating the amount of surface runoff, such as those based on the U.S. Soil Conservation Service curve number equation (CNE). U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksr* may be under- or overestimated to an unknown degree.

TABLE B-1-4

COPC LOSS CONSTANT DUE TO RUNOFF (SOIL INGESTION EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	ml water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils), recommended by U.S. EPA (1998) (no source or reference is provided for this range), and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksr* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.
Kd_s	Soil-water partition coefficient	ml water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

TABLE B-1-4

COPC LOSS CONSTANT DUE TO RUNOFF (SOIL INGESTION EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994), and NC DEHNR (1997) as a reference to calculate average annual runoff, *RO*. This reference provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these values are total contributions and not only surface runoff, U.S. EPA (1994) recommends that the volumes be reduced by 50 percent in order to estimate surface runoff.

Hillel, D. 1980. Fundamentals of Soil Physics. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the source documents that recommends using Table B-1-4; however, this document is not the original source of this equation (the source is unknown). This document also recommends the following:

- Estimating annual current runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service curve number equation (CNE); U.S. EPA (1985) is cited as an example of such a procedure.
- Default value of 0.2 (ml water/cm^3 soil) for soil volumetric water content (θ_{sw})

TABLE B-1-4

COPC LOSS CONSTANT DUE TO RUNOFF
(SOIL INGESTION EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water—Part I (Revised. 1985)*. Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate site-specific surface runoff.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents a range of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *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. Offices of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- Estimation of average annual runoff, *RO*, by using the *Water Atlas of the United States* (Geraghty et al. 1973)
- Default soil bulk density, *BD*, value of 1.5 g soil/cm^3 soil, based on the mean for loam soil that is taken from Carsel et al. (1988)
- Default soil volumetric water content, θ_{sw}, value of 0.2 (ml water/cm^3 soil)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of soil volumetric water content, θ_{sw}, values of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) (the original source of, or reference for, these values is not identified)
- A range of values for soil mixing depth, Z_s, for tilled and untilled soil (the original source of, or reference for, these values is not identified)
- Using the *Water Atlas of the United States* (Geraghty et al. 1973) to calculate average annual runoff, *RO*

TABLE B-1-5

COPC LOSS CONSTANT DUE TO LEACHING
(SOIL INGESTION EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the constant for COPC loss resulting from leaching of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksl*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to that of other residues. This uncertainty may underestimate *ksl*.
(3) The original source of this equation has not been identified. U.S. EPA (1998) presents the equation as shown here. U.S. EPA (1994b) and NC DEHNR (1997) replaced the numerator as shown with "*q*", defined as average annual recharge (cm/yr).

Equation

$$ksl = \frac{P + I - OR - E_v}{\theta_{sw} \cdot Z_s \cdot \left[1.0 + \left(BD \cdot Kd_s / \theta_{sw}\right)\right]}$$

Variable	Description	Units	Value
ksl	COPC loss constant due leaching	yr^{-1}	
P	Average annual precipitation	cm/yr	**18.06 to 164.19** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (U.S. Bureau of Census 1987; Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. We recommend using site-specific data. The following uncertainty is associated with this variable: To the extent that a site is not located near an established meteorological data station, and site-specific data are not available, default average annual precipitation data may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated. However, average annual precipitation data are reasonably available; therefore, we expect uncertainty introduced by this variable to be minimal.

TABLE B-1-5

COPC LOSS CONSTANT DUE TO LEACHING (SOIL INGESTION EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
I	Average annual irrigation	cm/yr	**0 to 100** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual irrigation information is not available, default values (generally based on the closest comparable location) may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures, such as those based on the U.S. Soil Conservation Service CNE. U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
E_v	Average annual evapotranspiration	cm/yr	**35 to 100** This variable is site-specific. This range is based on information presented in U. S. EPA (1998), representing data from 69 selected cities. The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual evapotranspiration information is not available, default values may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.

TABLE B-1-5

COPC LOSS CONSTANT DUE TO LEACHING (SOIL INGESTION EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	ml water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksl* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.

TABLE B-1-5

COPC LOSS CONSTANT DUE TO LEACHING (SOIL INGESTION EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default BD value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen and R.W. Shor. 1984. "A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture." Prepared for the U.S. Department of Energy under Contract No. DEAC05-840R21400.

For the continental United States, as cited in U.S. EPA (1998), this document is the source of a series of maps showing: (1) average annual precipitation (*P*), (2) average annual irrigation (*I*), and (3) average annual evapotranspiration isolines.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, *BD*, of 1.5 g soil/cm^3 soil for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997) as a reference for calculating average annual runoff, *RO*. This document provides maps with isolines of annual average surface runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these volumes are total contributions and not only surface runoff, U.S. EPA (1994b) recommends that the volumes be reduced by 50 percent in order to estimate average annual surface runoff.

TABLE B-1-5

COPC LOSS CONSTANT DUE TO LEACHING (SOIL INGESTION EQUATIONS)

(Page 5 of 5)

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the source documents that cites the use of the equation in Table B-1-5. However, the document is not the original source of this equation. This document also recommends the following:

- Estimation of average annual surface runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service CNE; U.S. EPA 1985 is cited as an example of such a procedure.
- A default value of 0.2 (ml water/cm^3 soil) for soil volumetric water content, θ_{sw}

U.S. Bureau of the Census. 1987. *Statistical Abstract of the United States:* 1987. 107th edition. Washington, D.C.

This document is a source of average annual precipitation (*P*) information for 69 selected cites, as cited in U.S. EPA (1990); these 69 cities are not identified.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater*. Part I (Revised 1985). Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate *RO*.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents values for soil mixing depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *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.

This document recommends (1) a default soil volumetric water content, θ_{sw}, value of 0.2 (ml water/cm^3 soil), and (2) a default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference source documents for the equation in Table B-1-5. The original source of this equation is not identified. This document also presents a range of values for soil mixing depth, *Z*, for tilled and untilled soil; the original source of these values is not identified.

TABLE B-1-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (SOIL INGESTION EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC loss constant from soil due to volatilization, and comes from *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (U.S. EPA 1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986).

Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to that of other residues. This uncertainty may underestimate *ksv*.

Equation

$$ksv = \left[\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right] \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \left(\frac{BD}{\rho_{soil}}\right) - \theta_{sw}\right]$$

Variable	Definition	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	
$3.1536 \times 10^{+7}$	Units conversion factor	s/yr	
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail, and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Values for this variable, estimated by using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, *ksv* may be under- or overestimated.

TABLE B-1-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (SOIL INGESTION EQUATIONS)

(Page 2 of 5)

Variable	Definition	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil / Depth (cm) / Reference Untilled / 2 / Brzuzy et al. (1995) Tilled / 20 / U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. Additional information on this subject can be obtained from Brzuzy and Hites (1995), which presents soil profiles for dioxin measurements. A default value of 2 cm for soil mixing depth for untilled soils is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for soil mixing depth for tilled soils is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeter in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
R	Universal gas constant	atm-m^3/mol-K	**8.205×10^{-5}** There are no uncertainties associated with this parameter.

TABLE B-1-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (SOIL INGESTION EQUATIONS)

(Page 3 of 5)

Variable	Definition	Units	Value
T_a	Ambient air temperature	K	**298** This variable is site-specific. U.S. EPA (1998) recommends a default ambient air temperature of 298 K. The following uncertainty is associated with this variable: To the extent that site-specific or local values for the variable are not available, default values may not accurately represent site-specific conditions. We expect the uncertainty associated with the selection of a single value from within the temperature range at a single location to be more significant than the uncertainty associated with choosing a single ambient temperature to represent all localities.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.
ρ_{soil}	Solids particle density	g/cm^3	**2.7** We recommend using this value, based on Blake and Hartage (1996) and Hillel (1980). The solids particle density will vary with location and soil type.
D_a	Diffusivity of COPC in air	cm^2/s	**Varies** This value is COPC-specific. We discuss this variable in detail, and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: The default D_a values may not accurately represent the behavior of COPCs under site-specific conditions. However, we expect the degree of uncertainty to be minimal.

TABLE B-1-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (SOIL INGESTION EQUATIONS)

(Page 4 of 5)

Variable	Definition	Units	Value
θ_{sw}	Soil volumetric water content	ml water/cm^3 soil	**0.2** This variable depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: (1) The default θ_{sw} values may not accurately reflect site-specific or local conditions; therefore, *ksv* may be under- or overestimated to a small extent, based on the limited range of values.

REFERENCES AND DISCUSSION

Blake, GR. and K.H. Hartge. 1996. *Particle Density. Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods.* Second Edition. Arnold Klute, Ed. American Society of Agronomy, Inc. Madison, WI., p. 381.

Carsel, R.F., R.S, Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density value, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

Miller, R.W. and D.T. Gardiner. 1998. In: *Soils in Our Environment.* J.U. Miller, Ed. Prentice Hall. Upper Saddle River, NJ. pp. 80-123.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents value for soil, mixing depth, Z_s, for tilled and untilled soil.

TABLE B-1-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (SOIL INGESTION EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends a default soil density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil; however, the source or basis for these values is not identified
- A default ambient air temperature of 298 K
- A range of soil volumetric water content, θ_{sw}

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 8)

Description

Use the equations in this table to calculate an average COPC soil concentration resulting from wet and dry deposition of particles and vapors to soil over the exposure duration. We recommend assuming that COPCs are incorporated only to a finite depth (the soil mixing zone depth, Z_s). Use the COPC soil concentration averaged over the exposure duration, represented by Cs, for carcinogenic COPCs, where risk is averaged over the lifetime of an individual. Because the hazard quotient associated with noncarcinogenic COPCs is based on a reference dose rather than a lifetime exposure, we recommend using the highest annual average COPC soil concentration occurring during the exposure duration period for noncarcinogenic COPCs. The highest annual average COPC soil concentration would most likely occur at the end of the time period of combustion and is represented by Cs_{tD}.

The following uncertainties are associated with this variable:

(1) We assume that the time period for deposition of COPCs resulting from hazardous waste combustion is a conservative, long-term value. This assumption may overestimate Cs and Cs_{tD}.

(2) Exposure duration values (T_2) are based on historical mobility studies and won't necessarily remain constant. Specifically, mobility studies indicate that most receptors that move remain in the vicinity of the combustion unit; however, it is impossible to accurately predict the probability that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants.

(3) A value of zero for T_1 doesn't account for exposure that may have occurred from historic operations and emissions from burning hazardous waste. This may underestimate Cs and Cs_{tD}.

(4) For soluble COPCs, leaching might lead to movement to below the mixing depth, resulting in lower concentrations within the mixing depth. This may overestimate Cs and Cs_{tD}.

(5) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This may underestimate Cs and Cs_{tD}.

Equation for Carcinogens

Soil Concentration Averaged Over Exposure Duration

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 8)

Equation for Noncarcinogens

Highest Annual Average Soil Concentration

where

For mercury modeling

Use $0.48Q$ for total mercury and $F_v = 0.85$ in the mercury modeling equation to calculate Ds. Apportion the calculated Ds value into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the assumed 98% Hg^{2+} and 2% MHg speciation split in soils (see Chapter 2). Elemental mercury (Hg^0) occurs in very small amounts in the vapor phase and does not exist in the particle or particle-bound phase. Therefore, assume elemental mercury deposition onto soils is negligible or zero. Evaluate elemental mercury for the direct inhalation pathway only (Table B-5-1).

$$Ds_{\,(Hg2+)} = 0.98\ Ds_{\,(Mercury)}$$
$$Ds_{\,(MHg)} = 0.02\ Ds_{\,(Mercury)}$$
$$Ds_{\,(Hg0)} = 0.0$$

Evaluate divalent and methyl mercury as individual COPCs. Calculate Cs for divalent and methyl mercury using the corresponding (1) fate and transport parameters for mercuric chloride (divalent mercury, Hg^{2+}) and methyl mercury provided in Appendix A-2, and (2) Ds (Hg^{2+}) and Ds (MHg) as calculated above.

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 8)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	
Cs_{tD}	Soil concentration at time *tD*	mg COPC/kg soil	
Ds	Deposition term	mg COPC/kg soil-yr	**Varies** U.S. EPA (1994a) and NC DEHNR (1991) recommend incorporating the use of a deposition term into the *Cs* equation. Uncertainties associated with this variable include the following: (1) Five of the variables in the equation for *Ds* (*Q, Cyv, Dywv*, *Dywp*, and *Dydp*) are COPC- and site-specific. Values of these variables are estimated through modeling. The direction and magnitude of any uncertainties shouldn't be generalized. (2) Based on the narrow recommended ranges, we expect uncertainties associated with *Vdv*, F_v, and *BD* to be low. (3) Values for Z_s vary by about one order of magnitude. Uncertainty is greatly reduced if you know whether soils are tilled or untilled.
tD	Time period over which deposition occurs (time period of combustion)	yr	**30** U.S. EPA (1998) suggests that this period of time can be ≥ 30 years. We recommend using 30 years unless site-specific information is available indicating that this assumption is unreasonable (see Chapter 6 of the HHRAP).
ks	COPC soil loss constant due to all processes	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-1-2. The COPC soil loss constant is the sum of all COPC removal processes. Uncertainty associated with this variable includes the following: COPC-specific values for *ksg* (one of the variables in the equation in Table B-1-2) are empirically determined from field studies. No information is available on applying these values to the site-specific conditions associated with affected facilities.

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 8)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td>T_2</td><td>Length of exposure duration</td><td>yr</td><td>

6, 30, or 40

We recommend the following reasonable maximum exposure (RME) values for T_2:

<table>
<tr><td>Exposure Duration</td><td>RME</td><td>Reference</td></tr>
<tr><td>Child Resident
Farmer Child
Fisher Child</td><td>6 years</td><td>U.S. EPA (1997b)</td></tr>
<tr><td>Adult Resident and Fisher</td><td>30 years</td><td>U.S. EPA (1997b)</td></tr>
<tr><td>Farmer</td><td>40 years</td><td>U.S. EPA (1994b)</td></tr>
</table>

U.S. EPA (1994c) recommended the following unreferenced values:

<table>
<tr><td>Exposure Duration</td><td>Years</td></tr>
<tr><td>Subsistence Farmer</td><td>40</td></tr>
<tr><td>Adult Resident</td><td>30</td></tr>
<tr><td>Subsistence Fisher</td><td>30</td></tr>
<tr><td>Child Resident</td><td>9</td></tr>
</table>

Uncertainties associated with this variable include the following:

(1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate Cs and Cs_{tD}.

(2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources; however, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate Cs and Cs_{tD}.

</td></tr>
<tr><td>T_1</td><td>Time period at the beginning of combustion</td><td>yr</td><td>

0

Consistent with U.S. EPA (1994bc), we recommend a value of 0 for T_1.

The following uncertainty is associated with this variable:

A T_1 of zero doesn't account for exposure that may have occurred from historical operation or emissions from the combustion of hazardous waste. This may underestimate Cs and Cs_{tD}.

</td></tr>
<tr><td>100</td><td>Units conversion factor</td><td>mg-cm^2/kg-cm^2</td><td></td></tr>
</table>

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 5 of 8)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td>Q</td><td>COPC emission rate</td><td>g/s</td><td>Varies
This variable is COPC- and site-specific. See Chapters 2 and 3 of the HHRAP for guidance regarding the calculation of this variable.</td></tr>
<tr><td>Z_s</td><td>Soil mixing zone depth</td><td>cm</td><td>2 to 20
We recommend the following values for Z_s:
<table><tr><td>Soil</td><td>Depth (cm)</td><td>Reference</td></tr><tr><td>Untilled</td><td>2</td><td>Brzuzy et al. (1995)</td></tr><tr><td>Tilled</td><td>20</td><td>U.S. EPA (1998)</td></tr></table>
U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998).

The following uncertainties are associated with this variable:
(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate Cs and Cs_{tD}.</td></tr>
<tr><td>BD</td><td>Soil bulk density</td><td>g soil/cm^3 soil</td><td>1.5
This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default <i>BD</i> value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable.

The following uncertainty is associated with this variable:
The recommended <i>BD</i> value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.</td></tr>
</table>

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 6 of 8)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss this variable and offer COPC-specific values in Appendix A-2. This range is based on the values presented in Appendix A-2. Values are also presented in U.S. EPA (1994c) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) Our F_v calculations assume a default S_T value for background plus local sources, rather than an S_T value for urban sources. If your site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
$Dydv$	Unitized yearly average dry deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
$Dywv$	Unitized yearly average wet deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
$Dydp$	Unitized yearly average dry deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
$Dywp$	Unitized yearly average wet deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 7 of 8)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Brzuzy, L.P. and R.A. Hites. 1995. "Estimating the Atmospheric Deposition of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans from Soils." *Environmental Science and Technology.* Volume 29. Pages 2090-2098.

This reference presents soil profiles for dioxin measurements.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This reference is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, BD, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Press, Inc. New York.

Cited by U.S. EPA (1998) for the statement that BD is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-1-1. This document also recommends using (1) a deposition term, Ds, and (2) COPC-specific F_v values.

Research Triangle Institute (RTI). 1992. *Preliminary Soil Action Level for Superfund Sites.* Draft Interim Report. Prepared for U.S. EPA Hazardous Site Control Division, Remedial Operations Guidance Branch. Arlington, Virginia. EPA Contract 68-W1-0021. Work Assignment No. B-03, Work Assignment Manager Loren Henning. December.

This document is a reference source for COPC-specific F_v values.

U.S. EPAU.S. EPA. 1992. *Estimating Exposure to Dioxin-Like Compounds.* Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005b.

The External Review Draft of the MPE document (the final is U.S. EPA 1998) cites this document as the source of values for soil mixing zone depth, Z_s, for tilled and untilled soils.

TABLE B-2-1

SOIL CONCENTRATION DUE TO DEPOSITION
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 8 of 8)

U.S. EPA. 1993b. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste. Office of Research and Development. Washington, D.C. September 24.

This document is a reference for the equation in Table B-2-1. It recommends using a deposition term, Ds, and COPC-specific F_v values in the Cs equation.

U.S. EPA. 1994a. *Draft Guidance for Performing Screening Level Risk Analysis 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. April 15.

This document is a reference for the equation in Table B-2-1; it recommends that the following be used in the Cs equation: (1) a deposition term, Ds, and (2) a default soil bulk density value of 1.5 g/cm^3, based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document recommends values for length of exposure duration, T_2, for the farmer.

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes.* Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- Values for the length of exposure duration, T_2
- Value of 0 for the time period of the beginning of combustion, T_1
- F_v values that range from 0.27 to 1 for organic COPCs
- Default soil bulk density value of 1.5 g/cm^3, based on a mean for loam soil from Carsel et al. (1988)

U.S. EPA. 1997a. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1997b. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002Fc. August.

This document is a reference source for values for length of exposure duration, T_2.

U.S. EPA. 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (MPE).* Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

TABLE B-2-2

COPC SOIL LOSS CONSTANT (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC soil loss constant, which accounts for the loss of COPCs from soil by several mechanisms.

Uncertainties associated with this equation include the following:

(1) COPC-specific values for *ksg* are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.
(2) The source of the equations in Tables B-2-3 through B-2-5 has not been identified.

Equation

Variable	Description	Units	Value
ks	COPC soil loss constant due to all processes	yr^{-1}	
ksg	COPC loss constant due to biotic and abiotic degradation	yr^{-1}	**Varies** This variable is COPC-specific. Values are available in the COPC tables in Appendix A-2. "Degradation rate" values are also presented in NC DEHNR (1997); however, no reference or source is provided for the values. U.S. EPA (1994a) and U.S. EPA (1994b) state that *ksg* values are COPC-specific; however, all *ksg* values are presented as zero (U.S. EPA 1994a) or as "NA" (U.S. EPA 1994b); the basis of these assumptions is not addressed. The following uncertainty is associated with this variable: COPC-specific values for *ksg* are empirically determined from field studies; no information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

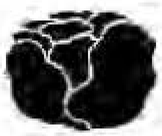
TABLE B-2-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** This variable is COPC- and site-specific, and is further discussed in Table B-2-3. Consistent with U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), we recommend a default value of zero for *kse* because contaminated soil erodes both onto the site and away from the site. Uncertainties associated with this variable include the following: (1) The source of the equation in Table B-2-3 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *kse*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *kse*.
ksr	COPC loss constant due to surface runoff	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-2-4. No reference document is cited for this equation. Using this equation is consistent with U.S. EPA (1998) and NC DEHNR (1997). U.S. EPA (1994a) assumes that all *ksr* values are zero but does not explain the basis of this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-2-4) include the following: (1) The source of the equation in Table B-2-4 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksr*.
ksl	COPC loss constant due to leaching	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-2-5. Using this equation is consistent with U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997). U.S. EPA (1994a) states that *ksl* is zero but doesn't explain the basis of this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-2-5) include the following: (1) The source of the equation in Table B-2-5 wasn't identified. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

TABLE B-2-2



COPC SOIL LOSS CONSTANT (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-2-6. This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from U.S. EPA (1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986). Uncertainties associated with this equation include the following: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

REFERENCES AND DISCUSSION

Hwang S. T. and Falco, J. W. 1986. “Estimation of multimedia exposures related to hazardous waste facilities”, In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the reference documents for the equations in Tables B-2-4 and B-2-5. This document is also cited as (1) the source for a range of COPC-specific degradation rates (*ksg*), and (2) one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because of contaminated soil eroding both onto the site and away from the site.

U.S. EPA. 1994a. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

This document is cited as a source for the assumptions that losses resulting from erosion (*kse*), surface runoff (*ksr*), degradation (*ksg*), leaching (*ksl*), and volatilization (*ksv*) are all zero.

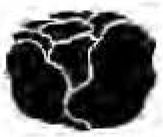
U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* Office of Emergency and Remedial Response. Office of Solid Waste. December 14.

This document is one of the reference documents for the equations in Tables B-2-4 and B-2-5. This document is also cited as one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero and the loss resulting from degradation (*ksg*) is “NA” or zero for all compounds.

TABLE B-2-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 4)



U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference documents for the equations for *ksr*, *ksl*, and *ksv*.

TABLE B-2-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the constant for COPC loss resulting from erosion of soil. Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend a default value of zero for *kse* because of contaminated soil eroding onto the site and away from the site. In site-specific cases where the permitting authority considers it appropriate to calculate a *kse*, we recommend using the equation presented in this table along with associated uncertainties. You can find additional discussion on determining *kse* in U.S. EPA (1998). Uncertainties associated with this equation include:

(1) For soluble COPCs, leaching might lead to movement below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *kse*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) in comparison to that of other residues. This uncertainty may underestimate *kse*.

Equation

$$kse = \frac{0.1 \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s} \cdot \left(\frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)} \right)$$

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend assuming a default value for *kse* of zero because contaminated soil erodes both onto the site and away from the site. Uncertainty may overestimate *kse*.
0.1	Units conversion factor	g-kg/cm^2-m^2	
X_e	Unit soil loss	kg/m^2-yr	**Varies** This variable is site-specific and is calculated by using the equation in Table B-4-13. The following uncertainty is associated with this variable: All of the equation variables are site-specific. Using default values rather than site-specific values for any or all of these variables will result in unit soil loss (X_e) estimates that are under- or overestimated to some degree. Based on default values, X_e estimates can vary over a range of less than two orders of magnitude.

TABLE B-2-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
SD	Sediment delivery ratio	unitless	**Varies** This value is site-specific and is calculated by using the equation in Table B-4-14. Uncertainties associated with this variable include the following: (1) The recommended default values for the empirical intercept coefficient, *a*, are average values that are based on studies of sediment yields from various watersheds. Therefore, those default values may not accurately represent site-specific watershed conditions. As a result, use of these default values may under- or overestimate *SD*. (2) The recommended default value for the empirical slope coefficient, *b*, is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, use of this default value may under- or overestimate *SD*.
ER	Soil enrichment ratio	unitless	**Inorganics: 1** **Organics: 3** COPC enrichment occurs because (1) lighter soil particles erode more quickly than heavier soil particles, and (2) concentration of organic COPCs—which is a function of organic carbon content of sorbing media—is expected to be higher in eroded material than in *in-situ* soil (U.S. EPA 1998). In the absence of site-specific data, we recommend a default value of 3 for organic COPCs and 1 for inorganic COPCs. This is consistent with other Agency guidance (1998), which recommends a range of 1 to 5 and a value of 3 as a "reasonable first estimate." This range has been used for organic matter, phosphorus, and other soil-bound COPCs (U.S. EPA 1998); however, no sources or references were provided for this range. *ER* is generally higher in sandy soils than in silty or loamy soils (U.S. EPA 1998). The following uncertainty is associated with this variable: The default *ER* value may not accurately reflect site-specific conditions; therefore, *kse* may be over- or underestimated to an unknown extent. Using county-specific *ER* values will reduce the extent of any uncertainties.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended *BD* value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.

TABLE B-2-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 5)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td>Z_s</td><td>Soil mixing zone depth</td><td>cm</td><td>

2 to 20

We recommend the following values for Z_s:

<table>
<tr><th>Soil</th><th>Depth (cm)</th><th>Reference</th></tr>
<tr><td>Untilled</td><td>2</td><td>Brzuzy et al. (1995)</td></tr>
<tr><td>Tilled</td><td>20</td><td>U.S. EPA (1998)</td></tr>
</table>

U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998).

The following uncertainties are associated with this variable:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}.

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.

</td></tr>
<tr><td>Kd_s</td><td>Soil-water partition coefficient</td><td>ml water/g soil
(or cm^3 water/g soil)</td><td>

Varies

This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2.

The following uncertainty is associated with this variable:

Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

</td></tr>
<tr><td>θ_{sw}</td><td>Soil volumetric water content</td><td>ml water/cm^3 soil</td><td>

0.2

This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b).

The following uncertainty is associated with this variable:

The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *kse* may be under- or overestimated to a small extent, based on the limited range of values.

</td></tr>
</table>

TABLE B-2-3

COPC LOSS CONSTANT DUE TO SOIL EROSION
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. “Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils.” *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because contaminated soil erodes both onto the site and away from the site.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil, as cited in U.S. EPA (1993).

U.S. EPA. 1994b. *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.

This document recommends (1) a default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988), and (2) a default soil volumetric water content, θ_{sw}, value of 0.2 (ml water/cm^3 soil), based on U.S. EPA (1993).

TABLE B-2-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1998. “*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*.” Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the source of a range of COPC enrichment ratio, *ER*, values. The recommended range, 1 to 5, was used for organic matter, phosphorous, and other soul-bound COPCs. This document recommends a value of 3 as a “reasonable first estimate,” and states that COPC enrichment occurs because lighter soil particles erode more quickly than heavier soil particles. Lighter soil particles have higher ratios of surface area to volume and are higher in organic matter content. Therefore, concentration of organic COPCs, which is a function of the organic carbon content of sorbing media, is expected to be higher in eroded material than in *in situ* soil.

This document is also a source of the following:

- A range of soil volumetric water content (θ_{sw}) values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil
- The equations in Tables B-1-3 and B-1-5.

TABLE B-2-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant due to runoff of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might result in movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate *ksr*.

Equation

$$ksr = \frac{RO}{\theta_{sw} \cdot Z_s} \cdot \left(\frac{1}{1 + \left(Kd_s \cdot BD / \theta_{sw} \right)} \right)$$

Variable	Description	Units	Value
ksr	COPC loss constant due to runoff	yr^{-1}	
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures for estimating the amount of surface runoff, such as those based on the U.S. Soil Conservation Service curve number equation (CNE). U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.

TABLE B-2-4

COPC LOSS CONSTANT DUE TO RUNOFF
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 4)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td>θ_{sw}</td><td>Soil volumetric water content</td><td>ml water/cm^3 soil</td><td>0.2
This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil’s field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils), recommended by U.S. EPA (1998) (no source or reference is provided for this range), and is consistent with U.S. EPA (1994b) and NC DEHNR (1997).

The following uncertainty is associated with this variable:
The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, <i>ksr</i> may be under- or overestimated to a small extent, based on the limited range of values.</td></tr>
<tr><td>Z_s</td><td>Soil mixing zone depth</td><td>cm</td><td>2 to 20
We recommend the following values for Z_s:

<table><tr><th>Soil</th><th>Depth (cm)</th><th>Reference</th></tr><tr><td>Untilled</td><td>2</td><td>Brzuzy et al. (1995)</td></tr><tr><td>Tilled</td><td>20</td><td>U.S. EPA (1998)</td></tr></table>
U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998).

The following uncertainties are associated with this variable:
(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate <i>Cs</i> and Cs_{tD}.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate <i>Cs</i> and Cs_{tD}.</td></tr>
<tr><td>Kd_s</td><td>Soil-water partition coefficient</td><td>ml water/g soil (or cm^3 water/g soil)</td><td>Varies
This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2.

The following uncertainty is associated with this variable:
Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.</td></tr>
</table>

TABLE B-2-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994), and NC DEHNR (1997) as a reference to calculate average annual runoff, *RO*. This reference provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these values are total contributions and not only surface runoff, U.S. EPA (1994) recommends that the volumes be reduced by 50 percent in order to estimate surface runoff.

Hillel, D. 1980. Fundamentals of Soil Physics. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the source documents that recommends using Table B-2-4; however, this document is not the original source of this equation (this source is unknown). This document also recommends the following:

- Estimating annual current runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service curve number equation (CNE); U.S. EPA (1985) is cited as an example of such a procedure.
- Default value of 0.2 (ml water/cm^3 soil) for soil volumetric water content (θ_{sw})

TABLE B-2-4

COPC LOSS CONSTANT DUE TO RUNOFF
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water—Part I (Revised. 1985)*. Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate site-specific surface runoff.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc June..

This document presents a range of values for soil mixing zone depth, Z_s, for tilled and untilled soil as cited in U.S. EPA (1993).

U.S. EPA. 1994b. *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. Offices of Emergency and Remedial Response. Office of Solid Waste. December 14.

This document recommends the following:

- Estimation of average annual runoff, *RO*, by using the *Water Atlas of the United States* (Geraghty et al. 1973)
- Default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on the mean for loam soil that is taken from Carsel et al. (1988)
- Default soil volumetric water content, θ_{sw}, value of 0.2 (ml water/cm^{3}soil), based on U.S. EPA (1993)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

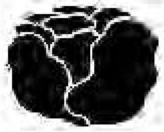
This document recommends the following:

- A range of soil volumetric water content, θ_{sw}, values of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) (the original source of, or reference for, these values is not identified)
- A range of values for soil mixing depth, Z_s, for tilled and untilled soil (the original source of, or reference for, these values is not identified)
- Using the *Water Atlas of the United States* (Geraghty et al. 1973) to calculate average annual runoff, *RO*

TABLE B-2-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 5)



Description

This equation calculates the constant for COPC loss resulting from leaching of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksl*.

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

(3) The original source of this equation hasn't been identified. U.S. EPA (1998) presents the equation as shown here. U.S. EPA (1994b) and NC DEHNR (1997) replaced the numerator as shown with "*q*", defined as average annual recharge (cm/yr).

Equation

$$ksl = \frac{P + I - OR - E_v}{\theta_{sw} \cdot Z_s \cdot \left[1.0 + \left(BD \cdot Kd_s / \theta_{sw}\right)\right]}$$

Variable	Description	Units	Value
ksl	COPC loss constant due to leaching	yr^{-1}	
P	Average annual precipitation	cm/yr	**18.06 to 164.19** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (U.S. Bureau of Census 1987; Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. We recommend using site-specific data. The following uncertainty is associated with this variable: (1) To the extent that a site is not located near an established meteorological data station, and site-specific data are not available, default average annual precipitation data may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated. However, average annual precipitation data are reasonably available; therefore, we expect uncertainty introduced by this variable to be minimal.

TABLE B-2-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
I	Average annual irrigation	cm/yr	**0 to 100** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual irrigation information is not available, default values (generally based on the closest comparable location) may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures, such as those based on the U.S. Soil Conservation Service CNE. U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
E_v	Average annual evapotranspiration	cm/yr	**35 to 100** This variable is site-specific. This range is based on information presented in U. S. EPA (1998), representing data from 69 selected cities. The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual evapotranspiration information is not available, default values may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.

TABLE B-2-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	(ml water/cm^3 soil)	**0.2** This variable is site-specific, and depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksl* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.

TABLE B-2-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended *BD* value may not accurately represent site-specific soil conditions.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen and R.W. Shor. 1984. “A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture.” Prepared for the U.S. Department of Energy under Contract No. DEAC05-840R21400.

For the continental United States, as cited in U.S. EPA (1998), this document is the source of a series of maps showing: (1) average annual precipitation (*P*), (2) average annual irrigation (*I*), and (3) average annual evapotranspiration isolines.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. “Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils.” *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, *BD*, of 1.5 g/cm^3 for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by NC DEHNR (1997), U.S. EPA (1994b), and U.S. EPA (1998) as a reference for calculating average annual runoff, *RO*. This document provides maps with isolines of annual average surface runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these volumes are total contributions and not only surface runoff, U.S. EPA (1994b) recommends that the volumes be reduced by 50 percent in order to estimate average annual surface runoff.

TABLE B-2-5

COPC LOSS CONSTANT DUE TO LEACHING
(CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 5 of 5)

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the source documents that cites the use of the equation in Table B-2-5. However, the document is not the original source of this equation. This document also recommends the following:

- Estimation of average annual surface runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service CNE; U.S. EPA 1985 is cited as an example of such a procedure.
- A default value of 0.2 (ml water/cm^3 soil) for soil volumetric water content, θ_{sw}

U.S. Bureau of the Census. 1987. *Statistical Abstract of the United States:* 1987. 107th edition. Washington, D.C.

This document is a source of average annual precipitation (*P*) information for 69 selected cites, as cited in U.S. EPA (1990); these 69 cities are not identified.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater*. Part I (Revised 1985). Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate *RO*.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc June..

This document presents values for soil mixing depth, Z_s, for tilled and untilled soil, as cited in U.S. EPA (1998).

U.S. EPA. 1994b. *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.

This document recommends (1) a default soil volumetric water content, θ_{sw}, value of 0.2 (ml water/cm^3 soil), and (2) a default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference source documents for the equation in Table B-2-5. The original source of this equation is not identified. This document also presents a range of values for soil mixing depth, *Z*, for tilled and untilled soil; the original source of these values is not identified.

TABLE B-2-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (U.S. EPA 1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986).

Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

Equation

$$ksv = \left[\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right] \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \left(\frac{BD}{\rho_{soil}}\right) - \theta_{sw}\right]$$

Variable	Definition	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	
3.1536 x 10^{+07}	Units conversion factor	s/yr	
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail, and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Values for this variable, estimated by using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, *ksv* may be under- or overestimated.

TABLE B-2-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

Variable	Definition	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
R	Universal gas constant	atm-m^3/mol-K	**8.205×10^{-5}** There are no uncertainties associated with this parameter.
T_a	Ambient air temperature	K	**298** This variable is site-specific. U.S. EPA (1990) also recommends an ambient air temperature of 298 K. The following uncertainty is associated with this variable: To the extent that site-specific or local values for the variable are not available, default values may not accurately represent site-specific conditions. We expect the uncertainty associated with the selection of a single value from within the temperature range at a single location to be more significant than the uncertainty associated with choosing a single ambient temperature to represent all localities.

TABLE B-2-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 4)

Variable	Definition	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.
ρ_{soil}	Solids particle density	g/cm^3	**2.7** We recommend using this value, based on Blake and Hartage (1996) and Hillel (1980). The solids particle density will vary with location and soil type.
D_a	Diffusivity of COPC in air	cm^2/s	**Varies** This value is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The default D_a values may not accurately represent the behavior of COPCs under site-specific conditions. However, we expect the degree of uncertainty to be minimal.
θ_{sw}	Soil volumetric water content	ml/cm^3 soil	**0.2** This variable depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: (1) Default θ_{sw} values may not accurately reflect site-specific or local conditions; therefore, *ksv* may be under- or overestimated to a small extent, based on the limited range of values.

TABLE B-2-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 4)

REFERENCES AND DISCUSSION

Blake, GR. and K.H. Hartge. 1996. *Particle Density. Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods.* Second Edition. Arnold Klute, Ed. American Society of Agronomy, Inc. Madison, WI., p. 381.

Carsel, R.F., R.S, Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. “Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils.” *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density value, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

Hwang S. T. and Falco, J. W. 1986. “Estimation of multimedia exposures related to hazardous waste facilities”, In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

Miller, R.W. and D.T. Gardiner. 1998. In: *Soils in Our Environment.* J.U. Miller, Ed. Prentice Hall. Upper Saddle River, NJ. pp. 80-123.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents value for soil, mixing depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends a default soil density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988).

U.S. EPA. 1998. “*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*.” Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil; however, the source or basis for these values is not identified
- A default ambient air temperature of 298 K
- A range of soil volumetric water content, θ_{sw}

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 12)

Description

This equation calculates the COPC concentration in aboveground vegetation, due to wet and dry deposition of COPCs onto plant surfaces. The limitations and uncertainty in calculating this value include the following:

(1) Uncertainties associated with the variables *Q, Dydp*, and *Dywp* are site-specific.
(2) The recommended equation for calculating *kp* values does not consider chemical degradation processes. Including chemical degradation would decrease the amount of time that a chemical remains on plant surfaces (half-life) and thereby increase *kp* values. *Pd* decreases with increased *kp* values. Reduction of half-life from the assumed 14 days to 2.8 days, for example, would decrease *Pd* about 5-fold.
(3) Calculating other parameter values (for example, *Fw* and *Rp*) is based directly or indirectly on studies of vegetation other than aboveground produce (primarily grasses). To the extent that the calculated parameter values don't accurately represent aboveground produce-specific values, uncertainty is introduced.
(4) The uncertainties associated with the variables F_v, *Tp*, and *Yp* are not expected to be significant.

As highlighted above, *Pd* is most significantly affected by the values assumed for *kp* and the extent to which parameter values (assumed based on studies of pasture grass) accurately reflect aboveground produce-specific values.

Equation

$$Pd = \frac{1000 \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp}$$

For mercury modeling

$$Pd_{(Mercury)} = \frac{1000 \cdot 0.48 Q_{(total)} \cdot \left(1 - F_{v_{(Hg^{2+})}}\right) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp}$$

Use *0.48Q* for total mercury and $F_v = 0.85$ in the mercury modeling equation above to calculate *Pd.* Apportion the calculated *Pd* value into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the 78% Hg^{2+} and 22% MHg speciation split in aboveground produce (see Chapter 2).

$Pd_{(Hg2+)} = 0.78\ Pd_{(Mercury)}$
$Pd_{(MHG)} = 0.22\ Pd_{(Mercury)}$

Evaluate divalent and methyl mercury as individual COPCs. Calculate *Pd* for divalent and methyl mercury using the corresponding equations above.

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 12)

Variable	Description	Units	Value
Pd	Concentration of COPC in aboveground produce due to direct (wet and dry) deposition	mg COPC/kg DW	
1000	Units conversion factor	mg/g	
Q	COPC-specific emission rate	g/s	**Varies** This value is COPC- and site-specific and is determined by air dispersion modeling. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are also COPC- and site-specific.
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. U.S. EPA (1994b) and NC DEHNR (1997) also present values. F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) The F_v calculation uses a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the F_v equation assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v. (3) Based on U.S. EPA (1994a), the F_v value for dioxins (PCDD/PCDF) is intended to represent 2, 3, 7, 8-TCDD TEQs by weighting data for all dioxin and furan congeners with nonzero TEFs. Uncertainty is introduced, because the Agency has been unable to verify the recommended F_v value for dioxins.
$Dydp$	Unitized yearly average dry deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

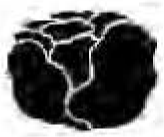
TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

Variable	Description	Units	Value
Rp	Interception fraction of the edible portion of plant	unitless	**0.39** We recommend using this default Rp value because it represents the most current information available; specifically, productivity and relative ingestion rates. As summarized in Baes et al. (1984), experimental studies of pasture grasses identified a correlation between initial Rp values and productivity (standing crop biomass [Yp]) (Chamberlain 1970): $Rp = 1 - e^{-\gamma \cdot Yp}$ where Rp = Interception fraction of the edible portion of plant (unitless) γ = Empirical constant. Chamberlain (1970) presented a range of 2.3 to 3.3; Baes et al. (1984) used 2.88, the midpoint for pasture grasses. Yp = Yield or standing crop biomass (productivity) (kg WW/m^2); the use of Yp value on a wet weight basis is in contrast to the equation presented in this table, which presents Yp on a dry weight basis. Baes et al. (1984) proposed using the same empirical relationship developed by Chamberlain (1970) for other vegetation classes. Class-specific estimates of the empirical constant, γ, were developed by forcing an exponential regression equation through several points, including average and theoretical maximum estimates of Rp and Yp (Baes et al. 1984) . The class-specific Rp estimates were then weighted, by relative ingestion of each class, to arrive at the weighted average Rp value of 0.39. U.S. EPA (1994b) and U.S. EPA (1995) recommended a weighted average Rp value of 0.05. However, the relative ingestion rates used in U.S. EPA (1994b) and U.S. EPA (1995) to weight the average Rp value were derived from U.S. EPA (1992) and U.S. EPA (1994b). The most current guidance available for ingestion rates of homegrown produce is the 1997 *Exposure Factors Handbook* (U.S. EPA 1997). The default Rp value of 0.39 was weighted by relative ingestion rates of homegrown exposed fruit and exposed vegetables found in U.S. EPA (1997). Uncertainties associated with this variable include the following: (1) The empirical relationship developed by Chamberlain (1970) on the basis of a study of pasture grass may not accurately represent aboveground produce. (2) The empirical constants developed by Baes et al. (1984) for use in the empirical relationship developed by Chamberlain (1970) may not accurately represent site-specific mixes of aboveground produce.

TABLE B-2-7

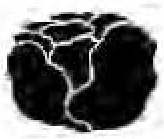
ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)



Variable	Description	Units	Value
Fw	Fraction of COPC wet deposition that adheres to plant surfaces	unitless	**0.2 for anions** **0.6 for cations and most organics** We recommend using the chemical class-specific values of 0.2 for anions and 0.6 for cations and most organics, as estimated by U.S. EPA (1994b) and U.S. EPA (1995). These values are the best available information, based on a review of the current scientific literature, with the following exception: We recommend using an *Fw* value of 0.2 for the three organic COPCs that ionize to anionic forms. These include (1) 4-chloroaniline, (2) n-nitrosodiphenylamine, and (3) n-nitrosodi-n-proplyamine (see Appendix A-2). The values estimated by U.S. EPA (1994b) and U.S. EPA (1995) were based on information presented in Hoffman, et al. (1992), which presented values for a parameter (*r*) termed the "interception fraction." These values were based on a study in which soluble radionuclides and insoluble particles labeled with radionuclides were deposited onto pasture grass via simulated rain. The parameter (*r*) is defined as "the fraction of material in rain intercepted by vegetation and initially retained" or, essentially, the product of *Rp* and *Fw,* as defined: $r = Rp \cdot Fw$ The *r* values developed by Hoffman, et al. (1992) were divided by an *Rp* value of 0.5 for forage (U.S. EPA 1994b). The *Fw* values developed by U.S. EPA (1994b) are 0.2 for anions and 0.6 for cations and insoluble particles. U.S. EPA (1994b) and U.S. EPA (1995) recommended using the *Fw* value calculated by using the *r* value for insoluble particles to represent organic compounds; however, no rationale for this recommendation was provided. Interception values (*r*)—as defined by Hoffman, et al. (1992)—have not been experimentally determined for aboveground produce. Therefore, U.S. EPA (1994b) and U.S. EPA (1995) apparently defaulted and assumed that the *Fw* values calculated for pasture grass (similar to forage) also apply to aboveground produce. The rationale for this recommendation was not provided. Uncertainties associated with this variable include the following: (1) Values of *r* developed experimentally for pasture grass may not accurately represent aboveground produce-specific *r* values. (2) Values of *r* assumed for most organic compounds, based on the behavior of insoluble polystyrene microspheres tagged with radionuclides, may not accurately represent the behavior of organic compounds under site-specific conditions.
Dywp	Unitized yearly wet deposition in particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)



(Page 5 of 12)

Variable	Description	Units	Value
kp	Plant surface loss coefficient	yr^{-1}	**18**

We recommend the *kp* value of 18 recommended by U.S. EPA (1998) and U.S. EPA (1994b). The recommended value is the midpoint of a possible range of values (7.44 to 90.36). U.S. EPA (1998) identified several processes—including wind removal, water removal, and growth dilution—that reduce the amount of COPC that has been deposited on a plant surface. The term *kp* is a measure of the amount of contaminant lost to these physical processes over time. U.S. EPA (1998) cites Miller and Hoffman (1983) for the following equation used to estimate *kp*:

$$kp \ = \ (ln\ 2 \ / \ t_{1/2}) \ \cdot \ 365 \text{ days/yr}$$

where

$t_{1/2}$ = half-life (days)

Miller and Hoffman (1983) report half-life values ranging from 2.8 to 34 days for a variety of COPCs on herbaceous vegetation. These half-life values result in *kp* values of 7.44 to 90.36 (yr^{-1}). U.S. EPA (1998) and U.S. EPA (1994b) recommend a *kp* value of 18, based on a generic 14-day half-life, corresponding to physical processes only. You can also calculate site- and compound-specific *kp* values using the equation from Miller and Hoffman (1983).

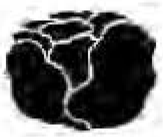
Uncertainties associated with this variable include the following:

(1) The recommended equation for calculating *kp* does not consider chemical degradation processes. Adding chemical degradation processes would decrease half-lifes and thereby increase *kp* values; plant concentration decreases as *kp* increases. Using a *kp* value that does not consider chemical degradation processes is protective.

(2) The half-life values reported by Miller and Hoffman (1983) may not accurately represent the behavior of compounds on aboveground produce.

(3) Based on this range (7.44 to 90.36), plant concentrations could range from about 1.8 times higher to about 5 times lower than the plant concentrations, based on a *kp* value of 18.

TABLE B-2-7

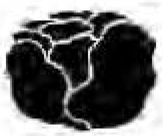


ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 6 of 12)

Variable	Description	Units	Value
Tp	Length of plant exposure to deposition per harvest of edible portion of plant	yr	**0.16** We recommend using a Tp value of 0.16 years; this is consistent with U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), which recommended treating Tp as a constant, based on the average period between successive hay harvests. Belcher and Travis (1989) estimated this period at 60 days. Tp is calculated as follows: 60 days ÷ 365 days/year = 0.16 years The following uncertainty is associated with this variable: The average period between successive hay harvests (60 days) may not reflect the length of the growing season or the length between successive harvests for site-specific aboveground produce crops. Pd will be (1) underestimated if the site-specific value of Tp is less than 60 days, or (2) overestimated if the site-specific value of Tp is more than 60 days.

TABLE B-2-7



ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 7 of 12)

Variable	Description	Units	Value
Yp	Yield or standing crop biomass of the edible portion of the plant (productivity)	kg DW/m^2	**Aboveground Produce: 2.24** We recommend using the Yp value of 2.24. Based on a review of the available literature, this value appears to be representative of the most complete and thorough information. U.S. EPA (1998) states that the best estimate of Yp is productivity. Baes et al. (1984) and Shor et al. (1982) define Yp as follows as: $Yp = Yh_i / Ah_i$ where Yh_i = Harvest yield of ith crop (kg DW) Ah_i = Area planted to ith crop (m^2) U.S. EPA (1994a) and NC DEHNR (1997) recommended using this equation. Class-specific Yp values were estimated by using average U.S. values for Yh and Ah for a variety of fruits and vegetables for 1993 (USDA 1994a and USDA 1994b). Yh values were converted to dry weight by using average conversion factors for fruits, fruiting vegetables, legumes, and leafy vegetables (Baes et al. 1984). Class-specific Yp values were grouped to reflect exposed fruits or exposed vegetables. Exposed fruit and exposed vegetable Yp values were then weighted by relative ingestion rates derived from the homegrown produce tables in U.S. EPA (1997). The average ingestion-weighted Yp value was 2.24. U.S. EPA (1994b) and U.S. EPA (1995) recommend a Yp value of 1.6; however, the produce classes and relative ingestion rates used to derive this Yp value are inconsistent with U.S. EPA (1997). The following uncertainty is associated with this variable: The harvest yield (Yh) and area planted (Ah) may not reflect site-specific conditions. This may under- or overestimate Yp.

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

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REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture*. ORNL-5786. Oak Ridge National Laboratory. Oak Ridge, Tennessee. September.

This document proposed using the same empirical relationship developed by Chamberlain (1970) for other vegetation classes. Class-specific estimates of the empirical constant, γ, were developed by forcing an exponential regression equation through several points, including average and theoretical maximum estimates of *Rp* and *Yp*.

The class-specific empirical constants developed are as follows:

Exposed produce — 0.0324
Leafy vegetables — 0.0846
Silage — 0.769

Beecher, G.D., and C.C. Travis. 1989. "Modeling Support for the RURA and Municipal Waste Combustion Projects: Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model." Interagency Agreement No. 1824-A020-A1, Office of Risk Analysis, Health and Safety Research Division, Oak Ridge National Laboratory. Oak Ridge, Tennessee. October.

This document recommends *Tp* values based on the average period between successive hay harvests and successive grazing.

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Pages 361-367. November 4.

This document is cited by U.S. EPA (1994a) and NC DEHNR (1997) as the source of the equation for calculating F_v. For discussion, see References and Discussion, Table B-1-1.

Chamberlain, A.C. 1970. "Interception and Retention of Radioactive Aerosols by Vegetation." *Atmospheric Environment.* 4:57 to 78.

Experimental studies of pasture grasses identified a correlation between initial *Rp* values and productivity (standing crop biomass [*Yp*]):

$$Rp = 1 - e^{-\gamma \cdot Yp}$$

where

γ = Empirical constant; range provided as 2.3 to 3.3
Yp = Yield or standing crop biomass (productivity) (kg DW/m^2)

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

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Hoffman, F.O., K.M. Thiessen, M.L. Frank, and B.G. Blaylock. 1992. "Quantification of the Interception and Initial Retention of Radioactive Contaminants Deposited on Pasture Grass by Simulated Rain." *Atmospheric Environment.* Vol. 26A. 18:3313 to 3321.

This document developed values for a parameter (*r*) that it termed "interception fraction," based on a study in which soluble gamma-emitting radionuclides and insoluble particles tagged with gamma-emitting radionuclides were deposited onto pasture grass (specifically, a combination of fescues, clover, and old field vegetation, including fescue) via simulated rain. The parameter, *r*, is defined as "the fraction of material in rain intercepted by vegetation and initially retained" or, essentially, the product of *Rp* and *Fw,* as defined for the HHRAP:

$$r = Rp \cdot Fw$$

Experimental *r* values obtained include the following:

- A range of 0.006 to 0.3 for anions (based on the soluble radionuclide iodide-131 [^{131}I]); when calculating *Rp* values for anions, U.S. EPA (1994a) used the highest geometric mean *r* value (0.08) observed in the study.
- A range of 0.1 to 0.6 for cations (based on the soluble radionuclide beryllium-7 [7Be]; when calculating *Rp* values for cations, U.S. EPA (1994a) used the highest geometric mean *r* value (0.28) observed in the study.
- A geometric range of values from 0.30 to 0.37 for insoluble polystyrene micro spheres (IPM) ranging in diameter from 3 to 25 micrometers, labeled with cerium-141 [^{141}Ce], [^{95}N]b, and strontium-85 ^{85}Sr; when calculating *Rp* values for organics (other than three organics that ionize to anionic forms: 4-chloroaniline, n-nitrosodiphenylamine, and n-nitrosodi-n-propylamine [see Appendix A-2]), U.S. EPA (1994a) used the geometric mean *r* value for IPM with a diameter of 3 micrometers; however, no rationale for this selection was provided.

The authors concluded that, for the soluble ^{131}I anion, interception fraction *r* is an inverse function of rain amount, whereas for the soluble cation ^{7}Be and the IPMs, *r* depends more on biomass than on amount of rainfall. The authors also concluded that (1) the anionic ^{131}I is essentially removed with the water after the vegetation surface has become saturated, and (2) the cationic ^{7}Be and the IPMs are adsorbed to or settle out onto the plant surface. This discrepancy between the behavior of the anionic and cationic species is consistent with a negative charge on the plant surface.

As summarized in U.S. EPA (1994a), this document is the source of the recommended F_v value of 0.27 for dioxins (polychlorinated dibenzodioxins/polychlorinated dibenzofurans [*PCDD/PCDF*]). This value is intended to represent 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) equivalents (TEQ) by weighting all dioxin and furan congeners with nonzero toxicity equivalency factors (TEF). U.S. EPA is investigating the appropriateness of the use of recommended F_v value for *PCDD/PCDFs.*

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

Miller, C.W. and F.O. Hoffman. 1983. "An Examination of the Environmental Half-Time for Radionuclides Deposited on Vegetation." *Health Physics*. 45 (3): 731 to 744.

This document is the source of the equation used to calculate *kp*:

$$kp = (\ln 2 / t_{1/2}) \cdot 365 \text{ days/year}$$

where

$t_{1/2}$ = half-life (days)

The study reports half-life values ranging from 2.8 to 34 days for a variety of COPCs on herbaceous vegetation.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This is one of the source documents for the equation in Table B-2-7.

Shor, R.W., C.F. Baes, and R.D. Sharp. 1982. *Agricultural Production in the United States by County: A Compilation of Information from the 1974 Census of Agriculture for Use in Terrestrial Food-Chain Transport and Assessment Models.* Oak Ridge National Laboratory Publication. ORNL-5786.

This document is the source of the equation used to calculate *Yp*:

$$Yp \approx P_i = Yh / Ah_i$$

where

P_i = productivity of *i*th crop (kilogram dry weight [kg DW]/square meter [m^2])
Yh = harvest yield of *i*th crop (kg DW)
Ah_i = area planted to crop *I* (m_2)

using the following information:

Produce Category	Empirical Constant (unitless)	*Rp* (unitless)	*Yp* (kg DW/m^2)	*Yp* (kg WW/m^2)	Intake (g/kg-day)
Exposed Fruits	0.0324	0.053	0.252	1.68	0.19
Exposed Vegetables	--	0.982	5.660	89.4	0.11
Leafy Vegetables	0.0846	0.215	0.246	2.86	--
Fruiting Vegetables	0.0324	0.996	10.52	167	--

Using the empirical relationship developed by Baes et al. (1984) to estimate *Rp* based on *Yp* requires that *Yp* term to be in whole-weight units. However, in Equation B-2-7, the *Yp* term should be in dry-weight units.

For exposed vegetables, *Rp* was derived from a weighted average of leafy vegetable and fruiting vegetable *Rp* values. This weighted average was based on whole-weight *Yp* values for leafy and fruiting vegetables. In addition, the exposed vegetable *Yp* value, both whole- and dry-weight, was derived by the following:

The following produce items were included in each category:

TABLE B-2-7

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Exposed Fruits—apple, apricot, berry, cherry, cranberry, grape, peach, pear, plum/prune, strawberry
Exposed Vegetables—asparagus, cucumber, eggplant, sweet pepper, tomato, snap beans, broccoli, brussel sprouts, cauliflower, celery, lettuce, and spinach

The ingestion rates for exposed fruits and exposed vegetables were based on U.S. EPA (1997), homegrown intake rates.

However, U.S. EPA has reviewed Baes et al. (1984), which also presents and discusses this equation.

U.S. Department of Agriculture (USDA). 1994a. *Vegetables 1993 Summary*. National Agricultural Statistics Service, Agricultural Statistics Board. Washington, D.C. Vg 1-2 (94).

USDA. 1994b. *Noncitrus Fruits and Nuts 1993 Summary*. National Agricultural Statistics Service, Agricultural Statistics Board, Washington, D.C. Fr Nt 1-3 (94).

One of the sources of Yh (harvest yield) and Ah (area planted for harvest) values for fruits, fruiting vegetables, legumes, and leafy vegetables used to calculate Yp (yield or standing crop biomass). Yh values were converted (for use in the equations) to dry weight by using average conversion factors for these same aboveground produce classes, as presented in Baes et al. (1984). The fruits and vegetables considered in each category are as follows:
Exposed fruits—apple, apricot, berry, cherry, cranberry, grape, peach, pear, plum/prune, and strawberry
Exposed vegetables—asparagus, cucumber, eggplant, sweet pepper, tomato, snap beans, broccoli, brussel sprouts, cauliflower, celery, lettuce, and spinach

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge*, Volumes I and II. Office of Water. Washington, D.C. EPA 822/R-93-001a.

This document is the source of ingestion rates (g DW/day) for aboveground produce classes—fruiting vegetables (4.2), leafy vegetables (2.0), and legumes (8.8)— that U.S. EPA (1994b) used to calculate Rp and Yp.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This is the source of ingestion rate for fruits, based on whole weight (88 g/day) and converted to dry weight by using an average whole-weight to dry-weight conversion factor for fruits (excluding plums/prunes, which had an extreme value) of 0.15 taken from Baes et al. (1984), used to calculate Rp and Yp.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This is one of the source documents for the equation in Table B-2-7.

This document also recommended weighted average Rp and Yp values of 0.05 and 1.6, respectively, based on the empirical relationships identified by Chamberlain (1970) and Shor et al. (1982).

$$Rp = 1 - e^{-\gamma \cdot Yp}$$

where
γ = Empirical constant; range provided as 2.3 to 3.3

TABLE B-2-7

ABOVEGROUND PRODUCE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 12 of 12)

Yp = Standing crop biomass (productivity) (kg DW/m^2)

and Shor et al. (1982):

$Yp = {}_{Yh} / Ah_i$

where

${}_{Yh}$ = Harvest yield of *i*th crop (kg DW)

Ah_i = Area planted to crop *I* (m^2)

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project*. Volumes I and II. Office of Solid Waste. March 3.

This is one of the source documents for the equation in Table B-2-7.

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source of relative ingestion rates.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This is one of the source documents for the equation in Table B-2-7. This document also states that the best estimate of *Yp* (yield or standing crop biomass) is productivity, as defined under Shor et al. (1982).

TABLE B-2-8

ABOVEGROUND PRODUCE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC concentration in aboveground produce resulting from direct uptake of vapor phase COPCs onto plant surfaces.

The limitations and uncertainty introduced in calculating this value include the following:

(1) The range of values for the variable *Bv* (air-to-plant biotransfer factor) is about 19 orders of magnitude for organic COPCs (this range may change on the basis of the tables in Appendix A-2). COPC-specific *Bv* values for nondioxin-like compounds may be overestimated by up to one order of magnitude, based on experimental conditions used to develop the algorithm used to estimate *Bv* values.

(2) The algorithm used to calculate values for the variable F_v assumes a default value for the parameter S_T (Whitby's average surface area of particulates [aerosols]) of background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. The S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower F_v value; however, the F_v value is likely to be only a few percent lower.

As highlighted by uncertainties described above, *Pv* is most affected by the value calculated for *Bv*.

Equation

For mercury modeling

Use *0.48Q* for total mercury and $F_v = 0.85$ in the mercury modeling equation to calculate *Pv*. Apportion the calculated *Pv* value into the divalent mercury (Hg^{2+}) and methyl mercury (MHG) forms based on the 78% Hg^{2+} and 22% MHG speciation split in aboveground produce.

$$Pv_{(Hg2+)} = 0.78\ Pv_{(Mercury)}$$

$$Pv_{(MHg)} = 0.22\ Pv_{(Mercury)}$$

Evaluate divalent and methyl mercury as individual COPCs. Calculate *Pv* for divalent and methyl mercury using the equations above.

TABLE B-2-8

ABOVEGROUND PRODUCE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
Pv	Concentration of COPC in aboveground produce due to air-to-plant transfer	µg COPC/g DW (equivalent to mg COPC/kg DW)	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling. See Chapters 2 and 3 of the HHRAP for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. U.S. EPA (1994b) and NC DEHNR (1997) also present values. F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) stated that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It is based on the assumption of a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Cyv	Unitized yearly average air concentration from vapor phase	µg-s/g-m^3	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-2-8

ABOVEGROUND PRODUCE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
Bv_{ag}	COPC air-to-plant biotransfer factor for aboveground produce	unitless ([mg COPC/g DW plant]/[(mg COPC/g air])	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Uncertainty associated with this variable include the following: (1) The studies that formed the basis of the algorithm used to estimate *Bv* values were conducted on azalea leaves and grasses, and may not accurately represent *Bv* for aboveground produce other than leafy vegetables.
VG_{ag}	Empirical correction factor for aboveground produce	unitless	**0.01 or 1.0** We recommend using a VG_{ag} value of 0.01 for COPCs with a log K_{ow} greater than 4 and a value of 1.0 for COPCs with a log K_{ow} less than 4. This variable is an empirical correction factor that reduces aboveground produce concentration. The equation in this table was developed to estimate the transfer of COPCs into leafy vegetation rather than into bulkier aboveground produce, such as apples. Because of the protective outer skin, size, and shape of bulky produce, transfer of lipophilic COPCs (log K_{ow} greater than 4) to the center of the produce is not likely. In addition, typical preparation techniques, such as washing, peeling, and cooking, will further reduce residues. U.S. EPA (1994b) recommended a value of 0.01, based on U.S. EPA (1994a), but made no distinction between fruits, vegetables, and leafy vegetation. NC DEHNR (1997), also citing U.S. EPA (1994a), recommended values of (1) 0.01 for fruits and fruiting vegetables, and (2) 1.0 for leafy vegetables. The values cited from U.S. EPA (1994a) are also based on information from Riederer (1990) and Wipf, et al. (1982). Uncertainties associated with this variable include the following: (1) U.S. EPA (1994a) assumed that translocation of compounds deposited on the surface of aboveground vegetation to inner parts of aboveground produce would be insignificant. This may underestimate *P*v. (2) U.S. EPA (1994a) assumed that the density of the skin and the whole vegetable are equal. This may overestimate *Pv*. (3) U.S. EPA (1994a) assumed that the thickness of vegetable skin and broadleaf tree skin are equal. The effect of this assumption on *Pv* is unknown.
ρ_a	Density of air	g/m^3	**1200.0** We recommend using this value based on Weast (1986). This reference indicates that air density varies with temperature. The density of air at both 20°C and 25°C (rounded to two significant figures) is $1.2 \times 10^{+3}$. U.S. EPA (1994b) and NC DEHNR (1997) recommended this same value but stated that it was calculated at standard conditions (20°C and 1 atmosphere).

TABLE B-2-8

ABOVEGROUND PRODUCE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion in Table B-1-1.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-2-8. This document also recommends that (1) F_v values be based on the work of Bidleman (1988), and (2) an empirical correction factor (VG_{ag}) be used to reduce concentrations of COPCs in specific vegetation types—specifically, a VG_{ag} value of 0.5 is recommended for silage. However, no rationale is provided for this value. This factor is used to reduce estimated COPC concentrations in specific vegetation types, because (1) Bv was developed for azalea leaves, and (2) it is assumed that there is insignificant translocation of compounds deposited on the surface of some vegetation types to the inner parts of this vegetation because of the lipophilicity of the COPC.

Riederer, M. 1990. "Estimating Partitioning and Transport of Organic Chemicals in the Foliage/Atmosphere System: Discussion of a Fugacity-Based Model." *Environmental Science and Technology*. 24: 829 to 837.

This is the source of the leaf thickness estimate used to estimate the empirical correction factor (VG_{ag}).

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume II: Properties, Sources, Occurrence, and Background Exposures.* External Review Draft. Office of Research and Development. Washington, DC. EPA/600/6-88/005Cc. June.

This document recommends an empirical correction factor of 0.01 to reduce estimated vegetable concentrations on the basis of the assumption that there is insignificant translocation of compounds deposited on the surface of aboveground vegetation to inner parts for aboveground produce. No reference or discussion regarding the validity of this assumption was given.

The factor of 0.01 is based on a similar correction factor for belowground produce (VG_{bg}), which is estimated on the basis of a ratio of the vegetable skin mass to vegetable total mass. The document assumes that the densities of the skin and vegetable are equal. The document also assumes an average vegetable skin leaf that is based on Rierderer (1990). Based on these assumptions, U.S. EPA (1994a) calculated VG_{bg} for carrots and potatoes of 0.09 and 0.03, respectively. By comparing these values to contamination reduction research completed by Wipf, et al. (1982), U.S. EPA (1994a) arrived at the recommended VG_{ag} value of 0.01.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This is one of the source documents for the equation in Table B-2-8. This document also presents a range (0.27 to 1) of F_v values for organic COPCs, based on the work of Bidleman (1988); F_v for all inorganics is set equal to zero.

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project*. Volumes I and II. Office of Solid Waste. March 3.

TABLE B-2-8

ABOVEGROUND PRODUCE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1997. Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment. Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

Based on attempts to model background concentrations of dioxin-like compounds in beef on the basis of known air concentrations, this document recommends reducing, by a factor of 10, *Bv* values calculated by using the Bacci,et al. (1992) algorithm The use of this factor "made predictions [of beef concentrations] come in line with observations."

Weast, R.C. 1986. *Handbook of Chemistry and Physics*. 66th Edition. Cleveland, Ohio. CRC Press.

This document is a reference for air density values, and is an update of Weast (1981).

Wipf, H.K., E. Homberger, N. Neuner, U.B. Ranalder, W. Vetter, and J.P. Vuilleumier. 1982. "TCDD Levels in Soil and Plant Samples from the Seveso Area." *In: Chlorinated Dioxins and Related Compounds: Impact on the Environment.* Eds. Hutzinger, O. et al. Pergamon, NY.

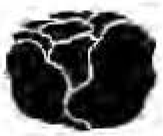


TABLE B-2-9

ABOVEGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the COPC concentration in aboveground produce due to direct uptake of COPCs from soil through plant roots. The limitations and uncertainty introduced in calculating this value include the following:

(1) The availability of site-specific information, such as meteorological data, will affect the accuracy of *Cs* estimates.

(2) Estimated COPC-specific soil-to-plant bioconcentration factors (*Br*) don't reflect site-specific conditions. This may be especially true for inorganic COPCs for which you could more accurately estimate *Br* by using site-specific *BCFs* rather than *BCF*s presented in Baes et al. (1984). We therefore recommend using plant uptake response slope factors derived in U.S. EPA (1992) for arsenic, cadmium, selenium, nickel, and zinc.

Equation

For mercury modeling, calculate aboveground produce concentration due to root uptake using the respective *Cs* and *Br* values for divalent mercury (Hg^{2+}) and methyl mercury (MHg).

Variable	Description	Units	Value
Pr_{ag}	Concentration of COPC in aboveground produce due to root uptake	mg COPC/kg DW	
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This value is COPC-and site-specific and calculated using the equation in Table B-2-1. Uncertainties associated with this variable are site-specific.

TABLE B-2-9

ABOVEGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
Br_{ag}	Plant-soil bioconcentration factor for aboveground produce	unitless ([mg COPC/kg DW plant]/[mg COPC/ kg soil])	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Uncertainties associated with this variable include the following: (1) Estimates of *Br* for some inorganic COPCs, based on plant uptake response slope factors, may be more accurate than those based on *BCF*s from Baes et al. (1984). (2) We recommend that uptake of organic COPCs from soil and transport of the COPCs to aboveground plant parts be calculated on the basis of a regression equation developed in a study of the uptake of 29 organic compounds. This regression equation, developed by Travis and Arms (1988), may not accurately represent the behavior of all organic COPCs under site-specific conditions.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture.* ORNL-5786. Oak Ridge National Laboratory. Oak Ridge, Tennessee. September.

Element-specific bioconcentration factors (*BCF*) were developed by Baes et al. (1984)—for both vegetative (stems and leaves) portions of food crops *(Bv)* and nonvegetative (reproductive—fruits, seeds, and tubers) portions of food crops (*Br*)—on the basis of a review and compilation of a wide variety of measured, empirical, and comparative data. Inorganic-specific *Br* values were calculated as a weighted average of vegetative (*Bv*) and reproductive (*Br*) *BCF*s. We recommend calculating inorganic-specific *Br* values as a weighted average of vegetative and reproductive BCFs. Relative ingestion rates determined from U.S. EPA (1997a) are 75 percent reproductive and 25 percent vegetative for homegrown produce. However, for exposed fruits only the reproductive BCFs should be used.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-2-9.

Travis, C.C. and A.D. Arms. 1988. "Bioconcentration of Organics in Beef, Milk, and Vegetation." *Environmental Science and Technology.* 22:271 to 274.

Based on paired soil and plant concentration data for 29 organic compounds, this document developed a regression equation relating soil-to-plant *BCF* (*Br*) to K_{ow};

$$\log Br = 1.588 - 0.578 \log K_{ow}$$

TABLE B-2-9

ABOVEGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ABOVEGROUND PRODUCE EQUATIONS)

(Page 3 of 3)

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge*, Volumes I and II. Office of Water. Washington, D.C. EPA 822/R-93-001a.

Source of plant uptake response factors for arsenic, cadmium, nickel, selenium, and zinc. Plant uptake response factors are converted to BCFs by multiplying the plant uptake response factor by 2.

U.S. EPA. 1994. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This is the source for ingestion rate for fruits, based on whole weight (88 g/day), and converted to dry weight by using an average whole-weight to dry-weight conversion factor for fruits (excluding plums/prunes, which had an extreme value) of 0.15 from Baes et al. (1984)—used to calculate *Br*.

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project*. Volumes I and II. Office of Solid Waste. March 3.

This document recommends using the *BCFs*, *Bv*, and *Br* from Baes et al. (1984) for calculating the uptake of inorganics into vegetative growth (stems and leaves) and nonvegetative growth (fruits, seeds, and tubers), respectively.

Although most *BCFs* used in this document come from Baes et al. (1984), values for some inorganics were apparently obtained from plant uptake response slope factors. These uptake response slope factors derived from U.S. EPA (1992).

U.S. EPA. 1997a. *Exposure Factors Handbook*. Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for relative intake rate split of 75 percent reproductive and 25 percent vegetative for homegrown produce.

U.S. EPA. 1997b. Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment. Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This is one of the source documents for the equation in Table B-2-9.

TABLE B-2-10

BELOWGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF BELOWGROUND PRODUCE EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC concentration in belowground vegetation due to direct uptake of COPCs from soil. The limitations and uncertainty introduced in calculating this value include the following:

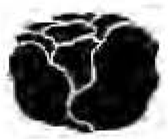
(1) The availability of site-specific information, such as meteorological data, will affect the accuracy of *Cs* estimates.

(2) Estimated COPC-specific soil-to-plant biotransfer factors (*Br*) don't necessarily reflect site-specific conditions. This may be especially true for inorganic COPCs for which estimates of *Br* would be more accurately estimated by using site-specific BCFs from Baes et al. (1984). Hence, for arsenic, cadmium, selenium, nickel, and zinc, we recommend using plant uptake response slope factors derived from U.S. EPA (1992).

For mercury modeling, belowground produce concentration due to root uptake is calculated using the respective *Cs* and *Br* values for divalent mercury (Hg^{2+}) and methyl mercury (MHg).

Variable	Description	Units	Value
Pr_{bg}	Concentration of COPC in belowground produce due to root uptake	mg COPC/kg DW	
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This value is COPC-and site-specific and calculated using the equation in Table B-2-1. Uncertainties associated with this variable are site-specific.

TABLE B-2-10



BELOWGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF BELOWGROUND PRODUCE EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
$Br_{rootveg}$	Plant-soil bioconcentration factor for belowground produce	unitless ([mg COPC/kg plant DW]/[mg COPC/ kg soil])	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Uncertainties associated with this variable include the following: (1) Estimates of *Br* for some inorganic COPCs, based on plant uptake response slope factors, may be more accurate than those based on BCFs from Baes et al. (1984). (2) We recommend that you calculate uptake of organic COPCs from soil and the transport of COPCs to belowground produce on the basis of a regression equation developed by Briggs et al (1982). This regression equation may not accurately represent the behavior of all classes of organic COPCs under site-specific conditions.

TABLE B-2-10

BELOWGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF BELOWGROUND PRODUCE EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
$VG_{rootveg}$	Empirical correction factor for belowground produce	unitless	**0.01 or 1.0** We recommend that you use a $VG_{rootveg}$ value of 0.01 for COPCs with a log K_{ow} greater than 4 and use a $VG_{rootveg}$ value of 1.0 for COPCS with a log K_{ow} less than 4. This variable is an empirical correction factor that reduces produce concentration. Because of the protective outer skin, size, and shape of bulky produce, transfer of lipophilic COPCs (log K_{ow} greater than 4) to the center of the produce isn't likely. In addition, typical preparation techniques, such as washing, peeling, and cooking, will further reduce residues. U.S. EPA (1994) recommended a $VG_{rootveg}$ value of 0.01 for lipophilic COPCs (log K_{ow} greater than 4) to reduce estimated belowground produce concentrations. This estimate for unspecified vegetables is based on: where M_{skin} = Mass of thin (skin) layer of an below ground vegetable (g) $M_{vegetable}$ = Mass of entire vegetable (g) If you assume that the density of the skin and the whole vegetable are the same, this equation can become a ratio of the volume of the skin to that of the whole vegetable. With this assumption, U.S. EPA (1994) calculated $VG_{rootveg}$ values of 0.09 and 0.03 for carrots and potatoes, respectively. U.S. EPA (1994) identified other processes, such as peeling, cooking, and cleaning, that will further reduce the vegetable concentration. Because of these other processes, U.S. EPA (1994) recommended a $VG_{rootveg}$ value of 0.01 for lipophilic COPCs. The following uncertainty is associated with this variable: U.S. EPA (1994) assumed that the density of the skin and the whole vegetable are equal. This may overestimate *Pr*. However, based on the limited range of $VG_{rootveg}$ (compared to *Br*), it appears that in most cases, these uncertainties will have a limited impact on the calculation of *Pr* and, ultimately, risk.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

TABLE B-2-10

BELOWGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF BELOWGROUND PRODUCE EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture.* ORNL-5786. Oak Ridge National Laboratory. Oak Ridge, Tennessee. September.

For discussion, see References and Discussion in Table B-2-10.

Briggs, G.G., R.H. Bromilow, and A.A. Evans. 1982. Relationships between lipophilicity and root uptake and translocation of non-ionized chemicals by barley. *Pesticide Science* 13:495-504.

This document presents the relationship between RCF and K_{ow} presented in the equation in Table B-2-10..

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is a source document for the equation in Table B-2-10.

Travis, C.C. and A.D. Arms. 1988. "Bioconcentration of Organics in Beef, Milk, and Vegetation." *Environmental Science and Technology.* 22:271 to 274.

Based on paired soil and plant concentration data for 29 organic compounds, this document developed a regression equation relating soil-to-plant BCF (Br) to K_{ow}

$$\log Br = 1.588 - 0.578 \log K_{ow}$$

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge*, Volumes I and II. Office of Water. Washington, D.C. EPA 822/R-93-001a.

Source of plant uptake response factors for arsenic, cadmium, nickel, selenium, and zinc. Plant uptake response factors are converted to BCFs by multiplying the plant uptake response factor by 2.

U.S. EPA. 1993. *Review Draft Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Office of Health and Environmental Assessment. Office of Research and Development. EPA-600-AP-93-003. November 10.

This document is a source of COPC-specific Kd_s values.

U.S. EPA. 1994. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This is a source document for $Vg_{rootveg}$ values.

TABLE B-2-10

BELOWGROUND PRODUCE CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF BELOWGROUND PRODUCE EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project*. Volumes I and II. Office of Solid Waste. March 3.

This document recommends using the *BCFs*, *Bv*, and *Br* from Baes et al. (1984) for calculating the uptake of inorganics into vegetative growth (stems and leaves) and nonvegetative growth (fruits, seeds, and tubers), respectively.

Although most *BCFs* used in this document come from Baes et al. (1984), values for some inorganics were apparently obtained from plant uptake response slope factors. These uptake response slope factors were calculated from field data, such as metal methodologies. References used to calculate the uptake response slope factors are not clearly identified.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 8)

Description

Use the equations in this table to calculate an average COPC soil concentration resulting from wet and dry deposition of particles and vapors to soil over the exposure duration. We recommend assuming that COPCs are incorporated only to a finite depth (the soil mixing zone depth, Z_s). Use the COPC soil concentration averaged over the exposure duration, represented by Cs, for carcinogenic COPCs, where risk is averaged over the lifetime of an individual. Because the hazard quotient associated with noncarcinogenic COPCs is based on a reference dose rather than a lifetime exposure, we recommend using the highest annual average COPC soil concentration occurring during the exposure duration period for noncarcinogenic COPCs. The highest annual average COPC soil concentration would most likely occur at the end of the time period of combustion and is represented by Cs_{tD}.

The following uncertainties are associated with this variable:

(1) We assume that the time period for deposition of COPCs resulting from hazardous waste combustion is a conservative, long-term value. This assumption may overestimate Cs and Cs_{tD}.

(2) Exposure duration values (T_2) are based on historical mobility studies and won't necessarily remain constant. Specifically, mobility studies indicate that most receptors that move remain in the vicinity of the combustion unit; however, it is impossible to accurately predict the probability that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants.

(3) Using a value of zero for T_1 doesn't account for exposure that may have occurred from historic operations and emissions from hazardous waste combustion. This may underestimate Cs and Cs_{tD}.

(4) For soluble COPCs, leaching might lead to movement to below the mixing depth, resulting in lower concentrations within the mixing depth. This uncertainty may overestimate Cs and Cs_{tD}.

(5) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This may underestimate Cs and Cs_{tD}.

Equation for Carcinogens

Soil Concentration Averaged Over Exposure Duration

$$Cs = \frac{\left(\dfrac{Ds \cdot tD - Cs_{tD}}{ks}\right) + \left(\dfrac{Cs_{tD}}{ks} \cdot [1 - \exp(-ks\,(T_2 - tD))]\right)}{(T_2 - T_1)} \quad for\ T_1 < tD < T_2$$

$$Cs = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left(\left[tD + \frac{\exp(-ks \cdot tD)}{ks}\right] - \left[T_1 + \frac{\exp(-ks \cdot T_1)}{ks}\right]\right) \quad for\ T_2 \le tD$$

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 8)

Equation for Noncarcinogens

Highest Annual Average Soil Concentration

$$Cs_{tD} = \frac{Ds \cdot [1 - \exp(-ks \cdot tD)]}{ks}$$

where

$$Ds = \frac{100 \cdot Q}{Z_s \cdot BD} \cdot [F_v \cdot (Dydv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)]$$

For mercury modeling

$$Ds_{(Mercury)} = \frac{100 \cdot [0.48Q_{(Total)}]}{Z_s \cdot BD} \cdot [F_{v_{(Hg^{2+})}} (Dydv + Dywv) + (Dydp + Dywp) \cdot [1 - F_{v_{(Hg^{2+})}}]$$

Use *0.48Q* for total mercury and $F_v = 0.85$ in the mercury modeling equation to calculate *Ds*. Apportion the calculated *Ds* value into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the assumed 98% Hg^{2+} and 2% MHg speciation split in soils (see Chapter 2). Elemental mercury (Hg^0) occurs in very small amounts in the vapor phase and does not exist in the particle or particle-bound phase. Therefore, assume elemental mercury deposition onto soils is negligible or zero. Evaluate elemental mercury for the direct inhalation pathway only (Table B-5-1).

$Ds_{(Hg2+)} = 0.98\ Ds_{(Mercury)}$
$Ds_{(Mhg)} = 0.02\ Ds_{(Mercury)}$
$Ds_{(Hg0)} = 0.0$

Evaluate divalent and methyl mercury as individual COPCs. Calculate *Cs* for divalent and methyl mercury using the corresponding (1) fate and transport parameters for mercuric chloride (divalent mercury, Hg^{2+}) and methyl mercury provided in Appendix A-2, and (2) *Ds* (Hg^{2+}) and *Ds* (MHg) as calculated above.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 8)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	
Cs_{tD}	Soil concentration at time *tD*	mg COPC/kg soil	
Ds	Deposition term	mg COPC/kg soil-yr	**Varies** U.S. EPA (1994a) and NC DEHNR (1997) recommended incorporating a deposition term into the *Cs* equation. Uncertainties associated with this variable include the following: (1) Five of the variables in the equation for *Ds (Q, Cyv, Dywv*, *Dywp*, and *Dydp*) are COPC- and site-specific. Values for these variables are estimated through modeling. The direction and magnitude of any uncertainties shouldn't be generalized. (2) Based on the narrow recommended ranges, we expect uncertainties associated with *Vdv*, F_v, and *BD* to be low. (3) Values for Z_s vary by about one order of magnitude. Uncertainty is greatly reduced if you know whether soils are tilled or untilled.
tD	Time period over which deposition occurs (time period of combustion)	yr	**30** U.S. EPA (1998) suggests that this period of time can be ≥ 30 years. We recommend using 30 years unless site-specific information is available indicating that this assumption is unreasonable (see Chapter 6 of the HHRAP).
ks	COPC soil loss constant due to all processes	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-3-2. The COPC soil loss constant is the sum of all COPC removal processes. Uncertainty associated with this variable includes the following: COPC-specific values for *ksg* (one of the variables in the equation in Table B-3-2) are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 8)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td>T_2</td><td>Length of exposure duration</td><td>yr</td><td>

6, 30, or 40

We recommend reasonable maximum exposure (RME) values for T_2:

<table>
<tr><th>Exposure Duration</th><th>RME</th><th>Reference</th></tr>
<tr><td>Child Resident
Farmer Child
Fisher Child</td><td>6 years</td><td>U.S. EPA (1997b)</td></tr>
<tr><td>Adult Resident and Fisher</td><td>30 years</td><td>U.S. EPA (1997b)</td></tr>
<tr><td>Farmer</td><td>40 years</td><td>U.S. EPA (1994b)</td></tr>
</table>

U.S. EPA (1994c) recommended the following unreferenced values:

<table>
<tr><th>Exposure Duration</th><th>Years</th></tr>
<tr><td>Subsistence Farmer</td><td>40</td></tr>
<tr><td>Adult Resident</td><td>30</td></tr>
<tr><td>Subsistence Fisher</td><td>30</td></tr>
<tr><td>Child Resident</td><td>9</td></tr>
</table>

Uncertainties associated with this variable include the following:

(1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate Cs and Cs_{tD}.

(2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources; however, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate Cs and Cs_{tD}.

</td></tr>
<tr><td>T_1</td><td>Time period at the beginning of combustion</td><td>yr</td><td>

0

Consistent with U.S. EPA (1994c), we recommend a value of 0 for T_1.

The following uncertainty is associated with this variable:

A T_1 of zero doesn't account for exposure that may have occurred from historical operation or emissions from burning hazardous waste. This may underestimate Cs and Cs_{tD}.

</td></tr>
<tr><td>100</td><td>Units conversion factor</td><td>mg-cm^2/kg-cm^2</td><td></td></tr>
</table>

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 8)

Variable	Description	Units	Value
Q	COPC emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 of the HHRAP for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended *BD* value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 6 of 8)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It assumes a default S_T value or background plus local sources, rather than an S_T value for urban sources. If your site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the F_v equation assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Dydv	Unitized yearly average dry deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dywv	Unitized yearly average wet deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dydp	Unitized yearly average dry deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dywp	Unitized yearly average wet deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 7 of 8)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Brzuzy, L.P. and R.A. Hites. 1995. "Estimating the Atmospheric Deposition of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans from Soils." *Environmental Science and Technology.* Volume 29. Pages 2090-2098.

This reference presents soil profiles for dioxin measurements.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This reference is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York.

Cited by U.S. EPA (1998) for the statement that *BD* is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-3-1. This document also recommends using (1) a deposition term, *Ds*, and (2) COPC-specific F_v values.

Research Triangle Institute (RTI). 1992. *Preliminary Soil Action Level for Superfund Sites.* Draft Interim Report. Prepared for U.S. EPA Hazardous Site Control Division, Remedial Operations Guidance Branch. Arlington, Virginia. EPA Contract 68-W1-0021. Work Assignment No. B-03, Work Assignment Manager Loren Henning. December.

This document is a reference source for COPC-specific F_v values.

U.S. EPA. 1992. *Estimating Exposure to Dioxin-Like Compounds.* Draft Report. Office of Research and Development. Washington, D.C. EPA/600/6-88/005b.

The External Review Draft of the MPE document (the final is U.S. EPA 1998) cites this document as the source of values for soil mixing zone depth, Z_s, for tilled and untilled soils.

TABLE B-3-1

SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 8 of 8)

U.S. EPA. 1993b. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. Working Group Recommendations. Office of Solid Waste. Office of Research and Development. Washington, D.C. September.

This document is a reference for the equation in Table B-3-1. It recommends using a deposition term, *Ds*, and COPC-specific F_v values in the *Cs* equation.

U.S. EPA 1994a. *Draft Guidance for Performing Screening Level Risk Analysis 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. April 15.

This document is a reference for the equation in Table B-3-1; it recommends that the following be used in the *Cs* equation: (1) a deposition term, *Ds*, and (2) a default soil bulk density value of 1.5 g/cm^3, based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document recommends values for length of exposure duration, T_2, for the farmer.

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes*. Office of Emergency and Remedial Response. Office of Solid Waste. December 14.

This document recommends the following:

- Values for the length of exposure duration, T_2
- Value of 0 for the time period of the beginning of combustion, T_1
- F_v values that range from 0.27 to 1 for organic COPCs
- Default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean for loam soil from Carsel et al. (1988)

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1997b. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002Fc. August.

This document is a reference source for values for length of exposure duration, T_2.

U.S. EPA. 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (MPE)*. Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

TABLE B-3-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC soil loss constant, which accounts for the loss of COPCs from soil by several mechanisms. Uncertainties associated with this equation include the following:

(1) COPC-specific values for *ksg* are empirically determined from field studies; no information is available regarding the application of these values to the site-specific conditions associated with affected facilities.
(2) The source of the equations in Tables B-3-3 through B-3-5 has not been identified.

Equation

$$ks = ksg + kse + ksr + ksl + ksv$$

Variable	Description	Units	Value
ks	COPC soil loss constant due to all processes	yr^{-1}	
ksg	COPC soil loss constant due to biotic and abiotic degradation	yr^{-1}	**Varies** This variable is COPC-specific and should be determined from the COPC tables in Appendix A-2. "Degradation rate" values are also presented in NC DEHNR (1997); however, no reference or source is provided for the values. U.S. EPA (1994a) and U.S. EPA (1994b) state that *ksg* values are COPC-specific; however, all *ksg* values are presented as zero (U.S. EPA 1994a) or as "NA" (U.S. EPA 1994b); the basis of these assumptions is not addressed. The following uncertainty is associated with this variable: COPC-specific values for *ksg* are empirically determined from field studies; no information is available on applying these values to the site-specific conditions associated with affected facilities.

TABLE B-3-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** This variable is COPC- and site-specific, and is further discussed in Table B-3-3. Consistent with U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), we recommend a default value of zero for *kse* because contaminated soil erodes both onto the site and away from the site. Uncertainties associated with this variable include the following: (1) The source of the equation in Table B-3-3 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *kse*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *kse*.
ksr	COPC loss constant due to surface runoff	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-3-4. No reference document is cited for this equation; using this equation is consistent with U.S. EPA (1998). U.S. EPA (1994a) assumes that all *ksr* values are zero but does not explain the basis of this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-3-4) include the following: (1) The source of the equation in Table B-3-4 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksr*.
ksl	COPC loss constant due to leaching	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-3-5. Using this equation is consistent with U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997). U.S. EPA (1994a) assumes that *ksl* is zero but does not explain the basis of this assumption. Uncertainties associated with this variable (calculated by using the equation in Table B-3-5) include the following: (1) The source of the equation in Table B-3-5 has not been identified. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

TABLE B-3-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-3-6. This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from U.S. EPA (1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986). Uncertainties associated with this equation include the following: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

REFERENCES AND DISCUSSION

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the reference documents for the equations in Tables B-3-4 and B-3-5. This document is also cited as (1) the source for a range of COPC-specific degradation rates (*ksg*), and (2) one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because of contaminated soil eroding both onto the site and away from the site.

U.S. EPA. 1994a. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

This document is cited as a source for the assumptions that losses resulting from erosion (*kse*), surface runoff (*ksr*), degradation (*ksg*), leaching (*ksl*), and volatilization (*ksv*) are all zero.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is one of the reference documents for the equations in Tables B-3-4 and B-3-5. This document is also cited as one of the sources that recommend using the assumption that the loss resulting from erosion (*kse*) is zero and the loss resulting from degradation (*ksg*) is "NA" or zero for all compounds.

TABLE B-3-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December. Environmental Criteria and Assessment Office. ORD. Cincinnati, Ohio.

This document is one of the reference documents for the equations for *ksr*, *ksl*, and *ksv*.

TABLE B-3-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the constant for COPC loss resulting from erosion of soil. Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend a default value of zero for *kse* because of contaminated soil eroding onto the site and away from the site. In site-specific cases where the permitting authority considers it appropriate to calculate a *kse*, we recommend using the equation presented in this table along with associated uncertainties. You can find additional discussion on determining *kse* in U.S. EPA (1998). Uncertainties associated with this equation include:

(1) For soluble COPCs, leaching might lead to movement below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *kse*.

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) in comparison to that of other residues. This uncertainty may underestimate *kse*.

Equation

$$kse = \frac{0.1 \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s} \cdot \left(\frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)} \right)$$

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend that the default value assumed for *kse* is zero because contaminated soil erodes onto the site and away from the site. Uncertainty may overestimate *kse*.
0.1	Units conversion factor	g-kg/cm^2-m^2	
X_e	Unit soil loss	kg/m^2-yr	**Varies** This variable is site-specific and is calculated by using the equation in Table B-4-13. The following uncertainty is associated with this variable: All of the equation variables are site-specific. Using default values rather than site-specific values for any or all of these variables will result in unit soil loss (X_e) estimates that are under- or overestimated to some degree. Based on default values, X_e estimates can vary over a range of less than two orders of magnitude.

TABLE B-3-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
SD	Sediment delivery ratio	unitless	**Varies** This value is site-specific and is calculated by using the equation in Table B-4-14. Uncertainties associated with this variable include the following: (1) The recommended default values for the empirical intercept coefficient, *a*, are average values that are based on studies of sediment yields from various watersheds. Therefore, those default values may not accurately represent site-specific watershed conditions. As a result, using these default values may under- or overestimate *SD*. (2) The recommended default value for the empirical slope coefficient, *b*, is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, using this default value may under- or overestimate *SD*.
ER	Soil enrichment ratio	unitless	**Inorganics: 1** **Organics: 3** COPC enrichment occurs because (1) lighter soil particles erode more than heavier soil particles, and (2) concentration of organic COPCs—which is a function of organic carbon content of sorbing media—is expected to be higher in eroded material than in *in-situ* soil (U.S. EPA 1998). In the absence of site-specific data, we recommend a default value of 3 for organic COPCs and 1 for inorganic COPCs. This is consistent with other U.S. EPA guidance (1998), which recommends a range of 1 to 5 and a value of 3 as a “reasonable first estimate.” This range has been used for organic matter, phosphorus, and other soil-bound COPCs (U.S. EPA 1998); however, no sources or references were provided for this range. *ER* is generally higher in sandy soils than in silty or loamy soils (U.S. EPA 1998). The following uncertainty is associated with this variable: The default *ER* value may not accurately reflect site-specific conditions; therefore, *kse* may be over- or underestimated to an unknown extent. Using county-specific *ER* values will reduce the extent of any uncertainties.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

TABLE B-3-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil / Depth (cm) / Reference Untilled / 2 / Brzuzy et al. (1995) Tilled / 20 / U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.
Kd_s	Soil-water partition coefficient	mL water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *kse* may be under- or overestimated to a small extent, based on the limited range of values.

TABLE B-3-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because contaminated soil erodes both onto the site and away from the site.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document recommends (1) a default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988), and (2) a default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil).

TABLE B-3-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the source of a range of COPC enrichment ratio, *ER*, values. The recommended range, 1 to 5, was used for organic matter, phosphorous, and other soul-bound COPCs. This document recommends a value of 3 as a "reasonable first estimate," and states that COPC enrichment occurs because lighter soil particles erode more quickly than heavier soil particles. Lighter soil particles have higher ratios of surface area to volume and are higher in organic matter content. Therefore, concentration of organic COPCs, which is a function of the organic carbon content of sorbing media, is expected to be higher in eroded material than in *in situ* soil.

This document is also a source of the following:

- A range of soil volumetric water content (θ_{sw}) values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil
- The equations in Tables B-3-3 and B-3-5.

TABLE B-3-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant due to runoff of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution in comparison to that of other residues. This uncertainty may underestimate *ksr*.

Equation

$$ksr = \frac{RO}{\theta_{sw} \cdot Z_s} \cdot \left(\frac{1}{1 + \left(Kd_s \cdot BD / \theta_{sw} \right)} \right)$$

Variable	Description	Units	Value
ksr	COPC loss constant due to runoff	yr^{-1}	
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures for estimating the amount of surface runoff, such as those based on the U.S. Soil Conservation Service curve number equation (CNE). U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksr* may be under- or overestimated to an unknown degree.

TABLE B-3-4

COPC LOSS CONSTANT DUE TO RUNOFF
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is depends on the available water and soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 (mL water/cm^3 soil) as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils), which is recommended by U.S. EPA (1998) (no source or reference is provided for this range), and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksr* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.
Kd_s	Soil-water partition coefficient	mL water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

TABLE B-3-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994), and NC DEHNR (1997) as a reference to calculate average annual runoff, *RO*. This reference provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these values are total contributions and not only surface runoff, U.S. EPA (1994) recommends that the volumes be reduced by 50 percent in order to estimate surface runoff.

Hillel, D. 1980. Fundamentals of Soil Physics. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the source documents that cites the use of Table B-3-4; however, this document is not the original source of this equation (this source is unknown). This document also recommends the following:

- Estimation of annual current runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service curve number equation (CNE); U.S. EPA (1985) is cited as an example of such a procedure.
- Default value of 0.2 (mL water/cm^3 soil) for soil volumetric water content (θ_{sw})

TABLE B-3-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water—Part I (Revised. 1985)*. Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate site-specific surface runoff.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents a range of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes*. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities. Offices of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- Estimation of average annual runoff, *R*O, by using the *Water Atlas of the United States* (Geraghty et al. 1973)
- Default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on the mean for loam soil that is taken from Carsel et al. (1988)
- Default soil volumetric water content, θ_{sw}, value of (0.2 mL water/cm^3 soil)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of soil volumetric water content, θ_{sw}, values of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) (the original source of, or reference for, these values is not identified)
- A range of values for soil mixing depth, Z_s, for tilled and untilled soil (the original source of, or reference for, these values is not identified)
- Using the *Water Atlas of the United States* (Geraghty et al. 1973) to calculate average annual runoff, *RO*

TABLE B-3-5

COPC LOSS CONSTANT DUE TO LEACHING
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC loss constant due to leaching of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksl.*

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl.*

(3) The original source of this equation has not been identified. U.S. EPA (1998) presents the equation as shown here. U.S. EPA (1994b) and NC DEHNR (1997) replaced the numerator as shown with "*q*", defined as average annual recharge (cm/yr).

Equation

$$ksl = \frac{P + I - OR - E_v}{\theta_{sw} \cdot Z_s \cdot \left[1.0 + \left(BD \cdot Kd_s / \theta_{sw}\right)\right]}$$

Variable	Description	Units	Value
ksl	Constant for COPC loss due to soil leaching	yr^{-1}	
P	Average annual precipitation	cm/yr	**18.06 to 164.19** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (U.S. Bureau of Census 1987; Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. We recommend using site-specific data. The following uncertainty is associated with this variable: To the extent that a site is not located near an established meteorological data station, and site-specific data are not available, default average annual precipitation data may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated. However, average annual precipitation data are reasonably available; therefore, we expect uncertainty introduced by this variable to be minimal.

TABLE B-3-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
I	Average annual irrigation	cm/yr	**0 to 100** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual irrigation information is not available, default values (generally based on the closest comparable location) may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures, such as those based on the U.S. Soil Conservation Service CNE. U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
E_v	Average annual evapotranspiration	cm/yr	**35 to 100** This variable is site-specific. This range is based on information presented in U. S. EPA (1998), representing data from 69 selected cities. The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual evapotranspiration information is not available, default values may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.

TABLE B-3-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksl* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate *Cs* and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *Cs* and Cs_{tD}.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

TABLE B-3-5

COPC LOSS CONSTANT DUE TO LEACHING
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen and R.W. Shor. 1984. "A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture." Prepared for the U.S. Department of Energy under Contract No. DEAC05-840R21400.

For the continental United States, as cited in U.S. EPA (1998), this document is the source of a series of maps showing: (1) average annual precipitation (*P*), (2) average annual irrigation (*I*), and (3) average annual evapotranspiration isolines.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, *BD*, of 1.5 g soil/cm^3 soil for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997) as a reference for calculating average annual runoff, *RO*. This document provides maps with isolines of annual average surface runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these volumes are total contributions and not only surface runoff, U.S. EPA (1994b) recommends that the volumes be reduced by 50 percent in order to estimate average annual surface runoff.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

TABLE B-3-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 5)

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the source documents that cites the use of the equation in Table B-1-5. However, the document is not the original source of this equation. This document also recommends the following:

- Estimation of average annual surface runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service CNE; U.S. EPA 1985 is cited as an example of such a procedure.
- A default value of 0.2 (mL water/cm^3 soil) for soil volumetric water content, θ_{sw}

U.S. Bureau of the Census. 1987. *Statistical Abstract of the United States:* 1987. 107th edition. Washington, D.C.

This document is a source of average annual precipitation (*P*) information for 69 selected cites, as cited in U.S. EPA (1998); these 69 cities are not identified.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater*. Part I (Revised 1985). Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate *RO*.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *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.

This document recommends (1) a default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil), and (2) a default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference source documents for the equation in Table B-3-5. The original source of this equation is not identified. This document also presents a range of values for soil mixing depth, *Z*, for tilled and untilled soil; the original source of these values is not identified.

TABLE B-3-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (U.S. EPA 1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986).

Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

Equation

$$ksv = \left[\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right] \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \left(\frac{BD}{\rho_{soil}}\right) - \theta_{sw}\right]$$

Variable	Definition	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	
3.1536 x 10^{+7}	Units conversion factor	s/yr	
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Values for this variable, estimated using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, *ksv* may be under- or overestimated.

TABLE B-3-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 4)

Variable	Definition	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil / Depth (cm) / Reference Untilled / 2 / Brzuzy et al. (1995) Tilled / 20 / U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
R	Universal gas constant	atm-m^3/mol-K	**8.205×10^{-5}** There are no uncertainties associated with this parameter.
T_a	Ambient air temperature	K	**298** This variable is site-specific. U.S. EPA (1998) recommends an ambient air temperature of 298 K. The following uncertainty is associated with this variable: To the extent that site-specific or local values for T_a are not available, default values may not accurately represent site-specific conditions. We expect the uncertainty associated with the selection of a single value from within the temperature range at a single location to be more significant than the uncertainty associated with choosing a single ambient temperature to represent all localities.

TABLE B-3-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 4)

Variable	Definition	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.
ρ_{soil}	Solids particle density	g/cm^3	**2.7** We recommend the use of this value, based on Blake and Hartage (1996) and Hillel (1980). The solids particle density will vary with location and soil type.
D_a	Diffusivity of COPC in air	cm^2/s	**Varies** This value is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The default D_a values may not accurately represent the behavior of COPCs under site-specific conditions. However, we expect the degree of uncertainty to be minimal.
θ_{sw}	Soil volumetric water content	mL/cm^3 soil	**0.2** This variable depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: (1) The default θ_{sw} values may not accurately reflect site-specific or local conditions; therefore, *ksv* may be under- or overestimated to a small extent, based on the limited range of values.

TABLE B-3-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 4)

REFERENCES AND DISCUSSION

Blake, GR. and K.H. Hartge. 1996. *Particle Density. Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods.* Second Edition. Arnold Klute, Ed. American Society of Agronomy, Inc. Madison, WI., p. 381.

Carsel, R.F., R.S, Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density value, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

Miller, R.W. and D.T. Gardiner. 1998. In: *Soils in Our Environment.* J.U. Miller, Ed. Prentice Hall. Upper Saddle River, NJ. pp. 80-123.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends a default soil density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil; however, the source or basis for these values is not identified
- A default ambient air temperature of 298 K
- A range of soil volumetric water content, θ_{sw}

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 10)

Description

This equation calculates the COPC concentration in forage and silage (aboveground vegetation) due to wet and dry deposition of COPCs onto plant surfaces. The limitations and uncertainty introduced in calculating this variable include the following:

(1) Uncertainties associated with the variables *Q, Dydp*, and *Dywp* are *COPC*- are site-specific.

(2) In calculating the variable *Fw*, values of *r* assumed for most organic compounds—based on the behavior of insoluble polystyrene micro spheres tagged with radionuclides— may accurately represent the behavior of organic compounds under site-specific conditions.

(3) The empirical relationship used to calculate the variable *Rp*, and the empirical constant for use in the relationship, may not accurately represent site-specific silage types.

(4) The recommended equation for calculating *kp* does not consider chemical degradation processes. This conservative approach contributes to the possible overestimation of plant concentrations.

(5) The harvest yield (*Yh*) and area planted (*Ah*) values used to estimate the variable *Yp* may not reflect site-specific conditions.

Equation

$$Pd = \frac{1000 \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp}$$

$$Pd_{(Mercury)} = \frac{1000 \cdot 0.48Q_{(total)} \cdot \left(1 - F_{v_{(Hg^{2+})}}\right) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - e^{(-kp \cdot Tp)}]}{Yp \cdot kp}$$

For mercury modeling

Forage and silage concentrations due to direct deposition are calculated using *0.48Q* for total mercury and $F_v = 0.85$ in the mercury modeling equation. Apportion the calculated *Pd* values into the divalent (Hg^{2+}) and methyl mercury (MHg) forms based on the 78% divalent mercury (Hg^{2+}) and 22% methyl mercury (MHg) speciation split in aboveground produce and forage.

$$Pd_{(Hg2+)} = 0.78\ Pd_{(Mercury)}$$
$$Pd_{(Mhg)} = 0.22\ Pd_{(Mercury)}$$

Evaluate divalent and methyl mercury as individual COPCs. Calculate *Pd* for divalent and methyl mercury using the corresponding equations above.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 10)

Variable	Description	Units	Value
Pd	Concentration of COPC in forage and silage due to direct deposition	mg COPC/kg DW	
1000	Units conversion factor	mg/g	
Q	COPC-specific emission rate	g/s	**Varies** This value is COPC- and site-specific, and is determined by air dispersion modeling. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. U.S. EPA (1994b) and NC DEHNR (1997) also present values. F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the F_v equation assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v. (3) Based on U.S. EPA (1994a), the F_v value for dioxins (PCDD/PCDF) is intended to represent 2, 3, 7, 8-TCDD TEQs by weighting data for all dioxin and furan congeners with nonzero TEFs. Uncertainty is introduced, because the Agency has been unable to verify the recommended F_v value for dioxins.
$Dydp$	Unitized yearly average dry deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 10)

Variable	Description	Units	Value
Fw	Fraction of COPC wet deposition that adheres to plant surfaces	unitless	**0.2 for anions** **0.6 for cations and most organics** We recommend using the chemical class-specific values of 0.2 for anions and 0.6 for cations and most organics, as estimated by U.S. EPA (1994b) and U.S. EPA (1995). These values are the best available information, based on a review of the current scientific literature, with the following exception: We recommend using an *Fw* value of 0.2 for the three organic COPCs that ionize to anionic forms. These include (1) 4-chloroaniline, (2) n-nitrosodiphenylamine, and (3) n-nitrosodi-n-proplyamine (see Appendix A-2). The values estimated by U.S. EPA (1994b) and U.S. EPA (1995) are based on information presented in Hoffman, Thiessen, Frank, and Blaylock (1992), which presented values for a parameter (*r*) termed the "interception fraction." These values were based on a study in which soluble radionuclides and insoluble particles labeled with radionuclides were deposited onto pasture grass via simulated rain. The parameter (*r*) is defined as "the fraction of material in rain intercepted by vegetation and initially retained" or, essentially, the product of *Rp* and *Fw,* as defined: $r = Rp \cdot Fw$ The *r* values developed by Hoffman, Thiessen, Frank, and Blaylock (1992) were divided by an *Rp* value of 0.5 for forage (U.S. EPA 1994b). The *Fw* values developed by U.S. EPA (1994b) are 0.2 for anions and 0.6 for cations and insoluble particles. U.S. EPA (1994b) and U.S. EPA (1995) recommended using the *Fw* value calculated by using the *r* value for insoluble particles to represent organic compounds; however, no rationale for this recommendation was provided. Uncertainties associated with this variable include the following: Values of *r* assumed for most organic compounds, based on the behavior of insoluble polystyrene micro spheres tagged with radionuclides, may not accurately represent the behavior of organic compounds under site-specific conditions.
Dywp	Unitized yearly average wet deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 10)

Variable	Description	Units	Value
Rp	Interception fraction of the edible portion of plant	unitless	**Forage: 0.5** **Silage: 0.46** We recommend using these default *Rp* values because they represent the most current information available; specifically, productivity and relative ingestion rates. As summarized in Baes et al. (1984), experimental studies of pasture grasses identified a correlation between initial *Rp* values and productivity (standing crop biomass [*Yp*]) (Chamberlain 1970): $Rp = 1 - e^{-\gamma \cdot Yp}$ where Rp = Interception fraction of the edible portion of plant (unitless) γ = Empirical constant. Chamberlain (1970) presents a range of 2.3 to 3.3; Baes et al. (1984) uses 2.88, the midpoint for pasture grasses. Yp = Yield or standing crop biomass (productivity) (kg DW/m^2) Baes et al. (1984) proposed using the same empirical relationship developed by Chamberlain (1970) for other vegetation classes. Class-specific estimates of the empirical constant, γ, were developed by forcing an exponential regression equation through several points, including average and theoretical maximum estimates of *Rp* and *Yp* (Baes et al. 1984) . The class-specific *Rp* estimates were then weighted, by relative ingestion of each class, to arrive at the weighted average *Rp* value of 0.5 for forage and 0.46 for silage. U.S. EPA (1994b) and U.S. EPA (1995) recommend a weighted average *Rp* value of 0.05. However, the relative ingestion rates used in U.S. EPA (1994b) and U.S. EPA (1995) to weight the average *Rp* value were derived from U.S. EPA (1992) and U.S. EPA (1994b). The most current guidance available for ingestion rates of homegrown produce is the 1997 *Exposure Factors Handbook* (U.S. EPA 1997). The default *Rp* values of 0.5 for forage and 0.46 for silage were weighted by relative ingestion rates of homegrown exposed fruit and exposed vegetables found in U.S. EPA (1997). Uncertainties associated with this variable include the following: The empirical constants developed by Baes et al. (1984) to use in the empirical relationship developed by Chamberlain (1970) may not accurately represent site-specific mixes of forage or silage.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 10)

Variable	Description	Units	Value
kp	Plant surface loss coefficient	yr^{-1}	**18** This value is site-specific. We recommend the *kp* value of 18 recommended by U.S. EPA (1998) and U.S. EPA (1994b). The *kp* value selected is the midpoint of a possible range of values (7.44 to 90.36). U.S. EPA (1998) identified several processes—including wind removal, water removal, and growth dilution—that reduce the amount of COPC that has been deposited on a plant surface. The term *kp* is a measure of the amount of contaminant lost to these physical processes over time. U.S. EPA (1998) cites Miller and Hoffman (1983) for the following equation used to estimate *kp*: $kp = (\ln 2 / t_{1/2}) \cdot 365$ days/year where $t_{1/2}$ = half-life (days) Miller and Hoffman (1983) report half-life values ranging from 2.8 to 34 days for a variety of COPCs on herbaceous vegetation. These half-life values convert to *kp* values of 7.44 to 90.36 yr^{-1}. U.S. EPA (1998) and U.S. EPA (1994b) recommend a *kp* value of 18, based on a generic 14-day half-life, corresponding to physical processes only. The 14-day half-life is approximately the midpoint of the range (2.8 to 34 days) estimated by Miller and Hoffman (1983). Uncertainties associated with this variable include the following: (1) the recommended equation for calculating *kp* does not consider chemical degradation processes. Adding chemical degradation processes would decrease half-lives and thereby increase *kp* values; plant concentration decreases as *kp* increases. Therefore, using a *kp* value that does not consider chemical degradation processes is conservative. (2) Based on this range (7.44 to 90.36), plant concentrations could range from about 1.8 times higher to about 5 times lower than the plant concentrations, based on a *kp* value of 18.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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Variable	Description	Units	Value
Tp	Length of plant exposure to deposition per harvest of edible portion of plant	yr	**Forage: 0.12** **Silage: 0.16** This variable is site-specific. We recommend the using these default values in the absence of site-specific information. U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997) recommended treating Tp as a constant, based on the average periods between successive hay harvests and successive grazing. For forage, the average of the average period between successive hay harvests (60 days) and the average period between successive grazing (30 days) is used (that is, 45 days). Tp is calculated as follows: $Tp = (60 \text{ days} + 30 \text{ days})/\ 2 \div 365 \text{ days/yr} = 0.12 \text{ yr}$ Use these average periods from Beecher and Travis (1989) when calculating the COPC concentration in cattle forage. When calculating the COPC concentration in silage fed to cattle, the average period between successive hay harvests (60 days) is used (Beecher and Travis 1989). Tp is calculated as follows: $Tp = 60 \text{ days} \div 365 \text{ days/year} = 0.16 \text{ year}$ The following uncertainty is associated with this variable: Using hay harvest cycles to estimate silage Tp values may underestimate COPC uptakes if silage types differ significantly from hay, and have longer actual harvest cycles (for example, if grains or other feeds with longer harvest cycles are used as silage). This underestimation will increase as actual harvest cycles increase, up to about 3 months. Beyond that time frame, if the kp value remains unchanged at 18. Higher Tp values will have little effect on predicted COPC concentrations in plants.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 7 of 10)

Variable	Description	Units	Value
Yp	Yield or standing crop biomass of the edible portion of the plant	kg DW/m^2	**Forage: 0.24** **Silage: 0.8** This variable is site-specific. We recommend the use of these default values in the absence of site-specific information. U.S. EPA (1990) states that the best estimate of *Yp* is productivity, which Baes et al. (1984) and Shor et al. (1982) define as follows: $Yp \approx {}_{Yh} / Ah_i$ where ${}_{Yh}$ = Harvest yield of *i*th crop (kg DW) Ah_i = Area planted to crop *i* (m^2) U.S. EPA (1994b) and NC DEHNR (1997) recommended using either previously calculated *Yp* values or the equation presented above to calculate a *Yp* value. We recommend that the forage *Yp* value be calculated as a weighted average of pasture grass and hay *Yp* values. Weights (0.75 for forage and 0.25 for hay) are based on (1) the fraction of a year during which cattle are assumed to be pastured and eating grass (9 mo/yr), and (2) the fraction of a year during which cattle are assumed to not be pastured and to be fed hay (3 mo/yr). An unweighted *Yp* value for pasture grass of 0.15 kg DW/m^2 is assumed (U.S. EPA 1994b). An unweighted *Yp* value for hay of 0.5 kg DW is calculated by the above equation, using the following dry harvest yield (*Yh*) and area harvested (*Ah*) values: Yh = 1.22 x 10^{+11} kg DW; from 1993 U.S. average wet weight *Yh* of 1.35 x 10^{11} kg (USDA 1994) and conversion factor of 0.9 (Agricultural Research Service 1994) Ah = 2.45 x 10^{+11} m^2; from 1993 U.S. average for hay (USDA 1994). The unweighted pasture grass and hay *Yp* values are multiplied by 3/4 and 1/4, respectively. They are then added to calculate the weighted forage *Yp* of 0.24 kg DW. We recommend that a production weighted U.S. average *Yp* of 0.8 be assumed for silage (Shor et al. 1982). The following uncertainty is associated with this variable: The harvest yield (*Yh*) and area planted (*Ah*) may not reflect site-specific conditions. This may under- or overestimate *Yp*.

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 8 of 10)

REFERENCES AND DISCUSSION

Agricultural Research Service. 1994. Personal communication regarding the dry weight fraction value for hay between G.F. Fries, and Glenn Rice and Jennifer Windholz, U.S. EPA Office of Research and Development. March 22.

This communication is cited by NC DEHNR (1997) for the fraction of 0.9 used to convert wet weight to dry weight for hay.

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture*. ORNL-5786. Oak Ridge National Laboratory. Oak Ridge, Tennessee. September.

This document proposes using the empirical relationship developed by Chamberlain (1970) (see further discussion in reference section of Table B-2-7) that identifies a correlation between initial Rp values and productivity (standing crop biomass [*Yp*]). It uses this relationship to calculate *Rp values* for forage and silage.

Beecher, G.D., and C.C. Travis. 1989. *Modeling Support for the RURA and Municipal Waste Combustion Projects: Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model*. Interagency Agreement No. 1824-A020-A1, Office of Risk Analysis, Health and Safety Research Division, Oak Ridge National Laboratory. Oak Ridge, Tennessee. October.

This document recommends *Tp* values based on the average period between successive hay harvests and successive grazing.

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Chamberlain, A.C. 1970. "Interception and Retention of Radioactive Aerosols by Vegetation." *Atmospheric Environment.* 4:57 to 78.

Cited by Baes et al. (1984) as a source for an empirical correlation between initial *Rp* values and productivity (standing crop biomass [*Yp*])

Hoffman, F.O., K.M. Thiessen, M.L. Frank, and B.G. Blaylock. 1992. "Quantification of the Interception and Initial Retention of Radioactive Contaminants Deposited on Pasture Grass by Simulated Rain." *Atmospheric Environment.* Vol. 26A, 18:3313 to 3321.

For discussion, see References and Discussion, Table B-2-7.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

Miller, C.W., and F.O. Hoffman. 1983. "An Examination of the Environmental Half-Time for Radionuclides Deposited on Vegetation." *Health Physics*. 45 (3): 731 to 744.

For discussion, see References and Discussion, Table B-2-7.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This a source document for the equation in Table B-3-7.

This document also recommends the following:
- *Rp* values of 0.5 (forage) and 0.46 (silage), based on the correlation from Chamberlain (1970)
- Treating *Tp* as a constant, based on the average periods between successive hay harvests and successive grazing
- Bidleman (1988) as source of equation for calculating F_v

Shor, R.W., C.F. Baes, and R.D. Sharp. 1982. *Agricultural Production in the United States by County: A Compilation of Information from the 1974 Census of Agriculture for Use in Terrestrial Food-Chain Transport and Assessment Models.* Oak Ridge National Laboratory Publication. ORNL-5786.

For discussion, see References and Discussion in Table B-2-7.

U.S. Department of Agriculture (USDA). 1994. *Vegetables 1993 Summary*. National Agricultural Statistics Service, Agricultural Statistics Board. Washington, D.C. Vg 1-2 (94).

This document is cited by NC DEHNR (1997) as the source for the average wet weight harvest yield (*Yh*) for hay.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document recommends an unweighted estimate of yield or standing crop biomass of 0.15 kg DW/m^2 for pasture grass.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This is one of the source documents for the equation in Table B-3-7. This document also
(1) developed and recommends *Fw* values of 0.2 for anions and 0.6 for cations and insoluble particles, based on dividing "*r*" values developed by Hoffman, Thiessen, Frank, and Blaylock (1992) and an *Rp* value of 0.5 for forage;
(2) recommends *Rp* values of 0.5 (forage) and 0.46 (silage);
(3) recommends a *kp* value of 18, based on a generic 14-day half-time, corresponding to physical processes only;
(4) recommends treating *Tp* as a constant ,based on the average periods between successive hay harvests and successive grazing; and
(5) cites Bidleman (1988) as the source of the equation for calculating F_v.

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project*. Volumes I and II. Office of Solid Waste. March 3.

This is one of the source documents for the equation in Table B-2-6. This document also recommends (1) using the *Fw* value calculated by using the *r* value for insoluble particles (see Hoffman, Thiessen, Frank, and Blaylock 1992) to represent organic compounds; however, no rationale for this recommendation is provided, and (2) *Rp* values of 0.5 (forage) and 0.46 (silage), based on the correlation from Chamberlain (1970).

TABLE B-3-7

FORAGE AND SILAGE CONCENTRATION DUE TO DIRECT DEPOSITION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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U.S. EPA. 1997. *Exposure Factors Handbook.* “Food Ingestion Factors”. Volume II. SAB Review Draft. EPA/600/P-95/002F. August.

This document is the source of relative ingestion rates.

U.S. EPA. 1998. “*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*.” Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This is one of the source documents for the equation in Table B-3-7. This document also states that the best estimate of *Yp* (yield or standing crop biomass) is productivity, as defined under Shor et al. (1982).

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC concentration in forage and silage (aboveground vegetation) resulting from direct uptake of vapor phase COPCs onto plant surfaces.

Uncertainties associated with the use of this equation include the following:

(1) The range of values for the variable *Bv* (air-to-plant biotransfer factor) is about 19 orders of magnitude for organic COPCs. COPC-specific *Bv* values for nondioxin-like compounds may be overestimated by up to one order of magnitude, based on experimental conditions used to develop the algorithm used to estimate *Bv* values.

(2) The algorithm used to calculate values for the variable F_v assumes a default value for the parameter S_T (Whitby's average surface area of particulates [aerosols]) of background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. The S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower *Fv* value; however, the F_v value is likely to be only a few percent lower.

Equation

$$Pv = Q \cdot F_v \cdot \frac{Cyv \cdot Bv_{forage} \cdot VG_{ag}}{\rho_a}$$

For mercury modeling

$$Pv_{(Mercury)} = (0.48Q_{(Total)}) \cdot F_{v_{(Hg^{2+})}} \cdot \frac{Cyv \cdot Bv_{forage} \cdot VG_{ag}}{\rho_a}$$

Use *0.48Q* for total mercury and $F_v = 0.85$ in the mercury modeling equation to calculate *Pv*. Apportion the calculated *Pv* value into the divalent (Hg^{2+}) and methyl mercury (MHg) forms based on the 78% Hg^{2+} and 22% MHg speciation split in aboveground produce and forage.

$Pv_{(Hg2+)} = 0.78\ Pv_{(Mercury)}$

$Pv_{(Mhg)} = 0.22\ Pv_{(Mercury)}$

Variable	Description	Units	Value
Pv	Forage and silage concentration due to air-to-plant transfer	µg COPC/g DW plant tissue (equivalent to mg/kg DW)	

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are also COPC- and site-specific.
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. U.S. EPA (1994b) and NC DEHNR (1997) also present values. F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Cyv	Unitized yearly average air concentration from vapor phase	μg-s/g-m^3	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
Bv_{forage}	Air-to-plant biotransfer factor for forage and silage	(mg COPC/g plant tissue DW)/ (mg COPC/g air)	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Uncertainty associated with this variable include the following: The studies that formed the basis of the algorithm used to estimate Bv values were conducted on azalea leaves and grasses, and may not accurately represent Bv for aboveground produce other than leafy vegetables.
VG_{ag}	Empirical correction factor for forage and silage	unitless	**Forage: 1.0** **Silage: 0.5** This variable is site-specific. In the absence of site-specific information, we recommend using VG_{ag} values of 1.0 for forage and 0.5 for silage. U.S. EPA (1994a), U.S. EPA (1994b), and NC DEHNR (1997) recommended an empirical correction factor to reduce estimated concentrations of constituents in specific vegetation types. This factor is used to reduce estimated bulky silage concentrations, because (1) Bv was developed for azalea leaves, and (2) it was assumed that there is insignificant translocation of compounds deposited on the surface of specific vegetation types (such as bulky silage) to the inner parts of this vegetation. U.S. EPA (1994a) and U.S. EPA (1994b) recommended a VG_{ag} of 1.0 for pasture grass and other leafy vegetation because of a direct analogy to exposed azalea and grass leaves. Pasture grass is described as "leafy vegetation." U.S. EPA (1994a) and U.S. EPA (1994b) didn't recommend a VG_{ag} value for silage. NC DEHNR (1997) recommended a VG_{ag} factor of 0.5 for bulky silage but didn't present a specific rationale for this recommendation. U.S. EPA (1995) noted that a volume ratio of outer surface area volume to whole vegetation volume could be used to assign a value to VG_{ag} for silage, if specific assumptions concerning the proportions of each type of vegetation of which silage may consist of were known (for example, corn and other grains). In the absence of specific assumptions concerning hay/silage/grain intake, however, U.S. EPA (1995) recommended assuming a VG_{ag} of 0.5 for silage without rigorous justification. The following uncertainty is associated with this variable: The VG_{ag} value of 0.5 for silage is recommended without vigorous justification. Depending on the composition of site-specific silage, the recommended value may under- or overestimate the actual value.

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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Variable	Description	Units	Value
ρ_a	Density of air	g/m^3	**1200.0** We recommend using this value based on Weast (1986). This reference indicates that air density varies with temperature. The density of air at both 20°C and 25°C (rounded to two significant figures) is $1.2 \times 10^{+3}$. U.S. EPA (1998) also recommends this value, but states that is was based on a temperature of 25°C. U.S. EPA (1994b) and NC DEHNR (1997) recommended this same value but stated that it was calculated at standard conditions (20°C and 1 atmosphere). Both documents cited Weast (1981).

REFERENCES AND DISCUSSION

Bidelman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367

For discussion, see References and Discussion in Table B-1-1.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is a source document for the equation in Table B-3-8. This document also recommended (1) that F_v values be based on the work of Bidleman (1988) , and (2) the use of an empirical correction factor (VG_{ag}) to reduce concentrations of COPCs in some vegetation types- (specifically, a VG_{ag} value of 0.5 is recommended for silage; however, no rationale is provided for this value). This factor is used to reduce estimated COPC concentrations in specific vegetation types, because (1) *Bv* was developed for azalea leaves, and (2) it is assumed that there is significant translocation of compounds deposited on the surface of specific vegetation types to the inner parts of this vegetation.

Riederer, M. 1990. "Estimating Partitioning and Transport of Organic Chemicals in the Foliage/Atmosphere: Discussion of a Fugacity-Based Model." *Environmental Science and Technology.* 24: 829 to 837.

This is the source of the leaf thickness used to estimate the empirical correction factor (VG_{ag}).

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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U. S. EPA 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume II: Properties, Sources, Occurrence, and Background Exposures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cb. June.

This document recommends an empirical correction factor of 0.01 to reduce estimated vegetable concentrations, based on the assumption that there is insignificant translocation of compounds deposited on the surface of aboveground vegetation to inner parts for aboveground produce. The document provides no reference or discussion regarding the validity of this assumption.

The factor of 0.01 is based on a similar correction factor for below ground produce (VG_{ag}), which is estimated based on a ratio of the vegetable skin mass to vegetable total mass. The document assumes that the density of the skin and vegetable are equal. The document also assumes an average vegetable skin leaf based on Rierderer (1990). Based on these assumptions, U.S. EPA (1994a) calculated VG_{ag} for carrots and potatoes of 0.09 and 0.03, respectively. By comparing these values to contamination reduction research completed by Wipf, et al. (1982), U.S. EPA (1994a) arrived at the recommended VG_{ag} of 0.01.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This is one of the source documents for the equation in Table B-3-8. This document also presents a range (0.27 to 1) of F_v values for organic COPCs, calculated on the basis of Bidleman (1988); F_v for all inorganics is set equal to zero.

U.S. EPA. 1995. *Review Draft Development of Human-Health Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project.* Volumes I and II. Office of Solid Waste. March 3.

This document presents estimated VG_{ag} values. U.S. EPA (1995) notes that a volume ratio of outer surface area volume to whole vegetation volume could be used to assign a value to VG_{ag} for silage, if specific assumptions (concerning the proportions of each type of vegetation of which silage may consist of) were known (for example, corn and other grains). In the absence of specific assumptions concerning hay/silage/grain intake, however, U.S. EPA (1995) recommends assuming a VG_{ag} value of 0.5 for silage (for COPCs with a log K_{ow} greater than 4) without rigorous justification.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions.*" Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

Based on attempts to model background concentrations of dioxin-like compounds in beef on the basis of known air concentrations, this document recommends reducing, by a factor of 10, *Bv* values calculated by using the Bacci, et al. (1992) algorithm The use of this factor "made predictions [of beef concentrations] come in line with observations."

Weast, R.C. 1981. *Handbook of Chemistry and Physics*. 62nd Edition. Cleveland, Ohio. CRC Press.

This document is a reference for air density values.

Weast, R.C. 1986. *Handbook of Chemistry and Physics*. 66th Edition. Cleveland, Ohio. CRC Press.

TABLE B-3-8

FORAGE AND SILAGE CONCENTRATION DUE TO AIR-TO-PLANT TRANSFER (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 6 of 5)

This document is a reference for air density values, and is an update of Weast (1981).

Wipf, H.K., E. Hamberger, N. Neuner, U.B. Ranalder, W. Vetter, and J.P. Vuilleumier. 1982 "TCDD Levels in Soil and Plant Samples from the Seveso Area." In: *Chlorinated Dioxins and Related Compounds: Impact on the Environment*. Eds. Hutzinger, O. et al. Perganon. New York.

TABLE B-3-9

FORAGE/SILAGE/GRAIN CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the COPC concentration in forage/silage/grain (aboveground produce), due to direct uptake of COPCs from soil through plant roots. Uncertainties associated with the use of this equation include the following:

(1) The availability of site-specific information, such as meteorological data, will affect the accuracy of *Cs* estimates.

(2) Estimated COPC-specific soil-to-plant bioconcentration factors (*Br*) don't reflect site-specific conditions. This may especially be true for inorganic COPCs for which *Br* would be more accurately estimated by using site-specific bioconcentration factors rather than bioconcentration factors from Baes et al. (1984). We therefore recommend using plant uptake response slope factors derived from U.S. EPA (1992) for arsenic, cadmium, selenium, nickel, and zinc.

Equation

$$Pr = Cs \cdot Br_{forage}$$

For mercury modeling, forage/silage/grain concentration due to root uptake is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective *Cs* and *Br* values.

$$Pr_{Hg^{2+}} = Cs_{Hg^{2+}} \cdot Br_{forage(Hg^{2+})}$$

$$Pr_{MHg} = Cs_{MHg} \cdot Br_{forage(MHg)}$$

Variable	Description	Units	Value
Pr	Concentration of COPC in forage/silage/grain due to root uptake	mgCOPC/kg DW plant tissue	
Cs	Average soil concentration over exposure duration	mg/kg	**Varies** This value is COPC and site-specific, and calculated using the equation in Table B-3-1. Uncertainties associated with this variable are site-specific.

TABLE B-3-9

FORAGE/SILAGE/GRAIN CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
Br_{forage}/ Br_{grain}	Plant-soil bioconcentration factor for forage/silage, or grain	unitless [(mg COPC/kg plant DW)/ (mg COPC/kg soil)]	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Uncertainties associated with this variable include the following: (1) Estimates of *Br* for some inorganic COPCs, based on plant uptake response slope factors, may be more accurate than those based on *BCF*s from Baes et al. (1984). (2) We recommend that you calculate uptake of organic COPCs from soil, and transport of the COPCs to aboveground plant parts, by using a regression equation developed in a study of the uptake of 29 organic compounds. This regression equation, developed by Travis and Arms (1988), may not accurately represent the behavior of all classes of organic COPCs under site-specific conditions.

REFERENCES AND DISCUSSION

Baes, C.F. R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides Through Agriculture*. ORNL-5786. Oak Ridge National Laboratory, Oak Ridge, Tennessee. September.

This document presents inorganic-specific transfer factors (*Br*) for both vegetative (*Bv*) portions of food crops and nonvegetative (reproductive—fruits, seeds, and tubers) portions (Br) of food crops. These bioconcentration factors were developed based on review and compilation of a wide variety of measured, empirical, and comparative data.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This is a source document for the equation in Table B-3-9.

Travis, C.C., and A.O. Arms. 1988. "Bioconcentration of Organics in Beef, Milk, and Vegetation." *Environmental Science and Technology.* 22:271 to 274.

This document developed the following regression equation relating soil-to-plant bioconcentration factor (*Br*) to K_{ow}, based on paired soil and plant concentration data:

$$\log Br = 1.588 - 0.578 \cdot \log K_{ow}$$

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge.* Volumes I and II. Office of Water. Washington, D.C. EPA 822/R-93-001a.

Source of plant uptake response factors for arsenic, cadmium, nickel, selenium, and zinc. Plant uptake response factors can be converted to BCFs by multiplying the plant uptake response factor by a factor of 2.

TABLE B-3-9

FORAGE/SILAGE/GRAIN CONCENTRATION DUE TO ROOT UPTAKE (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 3)

U.S. EPA. 1995. *Review Draft Development of Human Health Based and Ecologically Based Exit Criteria for the Hazardous Waste Identification Project. Volumes I and II.* Office of Solid Waste. March 3.

This document recommends using the bioconcentration factors *Bv* and *Br* from Baes et al. (1984) for calculating the uptake of inorganics into vegetative and nonvegetative growth, respectively.

Although most *BCFs* used in this document came from Baes et al. (1984), values for some inorganics were apparently obtained from plant uptake response slope factors. These uptake response slope factors were calculated from field data, such as metal loading rates and soil metal concentrations. However, the methodologies and references used to calculate the uptake response slope factors were not clearly identified.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This is one of the source documents for the equation in Table B-3-9.

TABLE B-3-10

BEEF CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 7)

Description

This equation first estimates the daily amount of COPCs cattle are exposed to through ingesting contaminated plant and soil material. The equation then recommends the use of biotransfer factors to transform the daily animal intake of a COPC (mg COPC/day) into an animal COPC tissue concentration (mg COPC/kg FW tissue).

The limitations and uncertainty introduced in calculating this variable include the following:

(1) Variables P_i and Cs are COPC- and site-specific. Uncertainties associated with these variables are site-specific.
(2) Uncertainties associated with the variables F_i, Qs, and Qp_i are expected to be minimal.
(3) Using a single Ba_{beef} value for each COPC may not accurately reflect site-specific conditions. It is not clear whether the default values are likely to under - or overestimate A_{beef}.

Based on the information below, A_{beef} is dependent on the concentrations of COPCs estimated in plant feeds and soil, and the biotransfer factor estimated for each constituent.

Equation

$$A_{beef} = \left(\sum \left(F_i \cdot Qp_i \cdot P_i \right) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{beef} \cdot MF$$

For mercury modeling, beef concentration due to plant and soil ingestion is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective P_i, Cs, and Ba_{beef} values.

Variable	Description	Units	Value
A_{beef}	Concentration of COPC in beef	mg COPC/kg FW tissue	
F_i	Fraction of plant type (i) grown on contaminated soil and ingested by the animal	unitless	**1** This variable is site- and plant type-specific. Plant types for cattle are typically identified as grain, forage, and silage. We recommend using a default value of 1.0 for all plant types when site-specific information is not available. This is consistent with U.S. EPA (1998), U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), which recommend assuming that 100 percent of the plant materials ingested by cattle were grown on soil contaminated by emissions. The following uncertainty is associated with this variable: (1) Assuming 100 percent of the plant materials eaten by cattle were grown on soil contaminated by emissions may overestimate A_{beef}.

TABLE B-3-10

BEEF CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 7)

Variable	Description	Units	Value
Qp_i	Quantity of plant type (i) ingested by the animal per day	kg DW plant/day	**Forage: 8.8** **Silage: 2.5** **Grain: 0.47** This variable is site- and plant type-specific. Plant types for cattle are typically identified as grain, forage, and silage. We recommend that you use the following *Qp* values when evaluating cattle raised by beef farmers: forage (8.8), silage (2.5), and grain (0.47). These values are consistent with U.S. EPA (1994c), and NC DEHNR (1997). The reference documents cite Boone et al. (1981), NAS (1987), McKone and Ryan (1989), and Rice (1994) as primary references for plant ingestion rates. Uncertainties introduced by this variable include the following: (1) The recommended daily grain ingestion rate of 0.47 kg dry weight (DW)/day is calculated indirectly from (1) a recommended total daily dry matter intake of 11.8 kg DW plant/day, based on NAS (1987) and McKone and Ryan (1989), and (2) daily ingestion rates of forage (8.8 kg/day) and silage (2.5 kg DW/day), recommended by Boone et al. (1981). However, Boone et al. (1981) recommended an alternative daily grain ingestion rate of 1.9 kg DW/day, about four times higher than the rate we recommend. As shown in Equations in Tables B-3-7 through B-3-9, the concentrations of COPCs in forage, silage, and grain are calculated similarly. Therefore, the relative amounts of forage, silage, and grain ingested daily have a limited effect on the intake of COPCs, if the total daily intake of dry matter is held constant. Therefore, limited uncertainty is introduced. (2) The recommended daily ingestion rates (total and plant type-specific) may not accurately represent site-specific or local conditions. Therefore, A_{beef} may be under- or overestimated, but to a limited degree.

TABLE B-3-10

BEEF CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 7)

Variable	Description	Units	Value
P_i	Concentration of COPC in plant type *(i)* ingested by the animal	mg/kg DW	**Varies** This variable is COPC-, site-, and plant type-specific; plant types for cattle are typically identified as grain, forage, and silage. Values for *Pd*, *Pv*, and *Pr* are calculated by using the equations in Tables B-3-7, B-3-8, and B-3-9; and then summed for each plant type to determine P_i. Uncertainties introduced by this variable include the following: (1) Some of the variables in the equations in Tables B-3-7, B-3-8, and B-3-9—including *Cs, Cyv, Q, Dydp*, and *Dywp*—are COPC- and site-specific. Uncertainties associated with these variables are site-specific. (2) In the equation in Table B-3-7, uncertainties associated with other variables include the following: F_w (values for organic compounds estimated on the basis of the behavior of polystyrene micro spheres), *Rp* (estimated on the basis of a generalized empirical relationship), *kp* (estimation process does not consider chemical degradation), and *Yp* (estimated on the basis of national harvest yield and area planted values). All of these uncertainties contribute to the overall uncertainty associated with P_i. (3) In the equation in Table B-3-8, COPC-specific *Bv* values for nondioxin-like compounds may be overestimated by up to one order of magnitude, based on experimental conditions used to develop the algorithm to estimate *Bv* values. (4) In the equation in Table B-3-9, COPC-specific plant-soil biotransfer factors (*Br*) may not reflect site-specific conditions. This may be especially true for inorganic COPCs for which estimates of *Br* would be more accurately estimated by using plant uptake response slope factors.
Qs	Quantity of soil ingested by the animal	kg/day	**0.5** This variable is site-specific. We recommend using a soil ingestion rate of 0.5 kg/day. This is consistent with NC DEHNR (1997) and U.S. EPA (1994c), which cite USDA (1994), Rice (1994), and NAS (1987). Uncertainties introduced by this variable include the following: (1) The recommended soil ingestion rate may not accurately represent site-specific or local conditions. However, we expect any differences between the recommended value and site-specific or local soil ingestion rates to be small. Therefore, we likewise expect any uncertainty introduced to be limited.
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-3-1. Uncertainties are site-specific.

TABLE B-3-10

BEEF CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 7)

Variable	Description	Units	Value
Bs	Soil bioavailability factor	unitless	**1.0** The soil bioavailability factor, *Bs,* can be thought of as the ratio between bioconcentration (or biotransfer) factors for soil and vegetation for a given contaminant. The efficiency of transfer from soil may differ from efficiency or transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, then this ratio would be less than 1.0. If it is equal or greater than that of vegetation, the *Bs* would be equal to or greater than 1.0. Since there is not enough data regarding bioavailability from soil, we recommend a default value of 1.0 for *Bs,* until more COPC data becomes available for this parameter. There is a fair amount of uncertainty associated with the use of this default value, because some COPCs may be much less bioavailable from soil than from plant tissues.
Ba_{beef}	Biotransfer factor for beef	day/kg FW tissue	**Varies** This variable is COPC-specific. We discuss this variable and COPC-specific values in Appendix A-2. Ba_{beef} is defined as the ratio of the COPC concentration in animal tissue (mg COPC/kg animal tissue) to the daily intake of the COPC (mg COPC/day) by the animal. Uncertainties introduced by this variable include the following: (1) We recommend using the regression equation developed by RTI (2005) to calculate Ba_{beef} values for organic COPCs. Uncertainties listed in RTI (2005) in deriving the regression equation include: 1) the necesssity to extrapolate data to steady state conditions, 2) metabolism may not be accounted for equally for all data points, and 3) there is a +/- 0.5 variability in measured Log Kow values available in the literature. In addition, values calculated by using this regression equation may not accurately represent the behavior of organic COPCs under site-specific conditions. Ba_{beef} and subsequent A_{beef} values may therefore be under- or overestimated to some degree. (2) We recommend calculating Ba_{beef} values for metals be using single COPC-specific uptake factors developed by Baes et al. (1984). These uptake factors may not accurately represent the behavior of inorganic COPCs under site-specific conditions. Ba_{beef} and subsequent A_{beef} values may therefore be under- or overestimated to some degree.
MF	Metabolism factor	unitless	**0.01 and 1.0** This variable is COPC-specific. Based on a study by Ikeda et al. (1980), U.S. EPA (1995a) recommended using a metabolism factor to account for metabolism in animals to offset the amount of bioaccumulation suggested by biotransfer factors. MF applies only to beef, milk, and pork. It does not apply to direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish. U.S. EPA (1995b) recommended an MF of 0.01 for bis(2-ethylhexyl)phthalate (BEHP) and 1.0 for all other contaminants.

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REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. "Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides Through Agriculture." Oak Ridge National Laboratory, Oak Ridge, Tennessee.

U.S. EPA (1994c) recommends Baes et al. (1984) as a source of Ba_{beef} values for inorganics.

Boone, F.W., Yook C. Ng, and John M. Palms. 1981. "Terrestrial Pathways of Radionuclide Particulates." *Health Physics*, Vol. 41, No. 5, pp. 735-747. November.

This document is identified as a source of plant ingestion rates. Boone et al. (1981) reports forage, grain, and silage ingestion rates of 8.8, 1.9, and 2.5 kg DW/day, respectively, for beef cattle.

Ikeda, G.J., P.P. Sapenza, and J.L. Couvillion. 1980. "Comparative distribution, excretion, and metabolism of di(2-ethylhexyl)phthalate in rats, dogs, and pigs." *Food Cosmet. Toxicology.* 18:637-642.

McKone, T.E., and P.B. Ryan. 1989. *Human Exposures to Chemicals Through Food Chains: An Uncertainty Analysis.* Livermore, California: Lawrence Livermore National Laboratory Report. UCRL-99290.

This document is cited as a source of plant ingestion rates. McKone and Ryan (1989) report an average total ingestion rate of 12 kg DW/day for the three plant feeds, which is consistent with the total recommended by other guidance documents for cattle (that is, forage, grain, and silage total of 11.8 kg DW/day).

National Academy of Sciences (NAS). 1987. *Predicting Feed Intake of Food-Producing Animals*. National Research Council, Committee on Animal Nutrition, Washington, D.C.

This document is identified as a source of food ingestion rates. NC DEHNR (1997) and U.S. EPA (1994c) note that NAS (1987) reports a daily dry matter intake that is 2 percent of an average beef cattle body weight of 590 kilograms. This results in a daily total intake rate of 11.8 kg DW/day, and the daily soil ingestion rate of approximately 0.5 kg soil/day (based on USDA [1994]).

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is a reference source for the equation in Table B-3-10.

NC DEHNR (1997) recommends forage, grain, and silage ingestion rates of 3.8, 3.8, and 1.0 kg dry weight/day, respectively, for typical farmer beef cattle. NC DEHNR (1997) reports Rice (1994) as a references for these variable.

Research Triangle Institute (RTI). 2005. *Methodology for Predicting Cattle Biotransfer Factors.* Prepared for U.S. Environmental Protection Agency (EPA) Office fo Solid Waste. EPA Contract No. 68-W-03-042. August.

U.S. Department of Agriculture (USDA). 1994. Personal Communication Between G.F. Fries, and Glenn Rice and Jennifer Windholtz, U.S. Environmental Protection Agency, Office of Research and Development. Agricultural Research Service. March 22.

NC DEHNR (1997) and U.S. EPA (1994c) note that this reference reports soil ingestion for cattle to be 4 percent of the total daily dry matter intake.

U.S. EPA. 1993. *Technical Support Document for Land Application of Sewage Sludge.* Volumes I and II. EPA 822/R-93-001a. Office of Water. Washington, D.C.

U.S. EPA (1995) recommended that bioconcentration factors for the metals cadmium, mercury, selenium, and zinc presented in this document be used to derive Ba_{beef} values. Following the method recommended by U.S. EPA (1992) for dioxins, the bioconcentration factors—with units of (kilograms feed DW/kilogram tissue DW—are divided by feed ingestion rates (kilogram feed DW/day]) to calculate Ba_{beef} values (day/kilogram tissue DW). U.S. EPA (1993) recommended a feed ingestion rate of 20 kg DW/day.

U.S. EPA. 1994a. *Estimating Exposures to Dioxin-like Compounds. Volume III: Site-specific Assessment Procedures.* Office of Research and Development. EPA/600/6-88/005Cc. External Review Draft. June.

This document recommends an F_i value of 1; this value assumes that 100 percent of the plant materials ingested by cattle have been grown on soil contaminated by emissions.

U.S. EPA. 1994b. *Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Solid Waste and Emergency Response. EPA-530-R-94-021. April.

This document recommends an F_i value of 1; this value assumes that 100 percent of the plant materials ingested by cattle have been grown on soil contaminated by emissions.

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is one of the reference source documents for the equation in Table B-3-10. This document also recommends the following:

- An F_i value of 100 percent
- Qp_i values for forage, silage, and grain of 8.8, 2.5 and 0.47 kg dry weight/day, respectively, based on Boone et al. (1981), NAS (1987), McKone and Ryan (1989), and Rice (1994)
- A soil ingestion rate for cattle (θ_{sw}) of 0.5 kg/day, based on USDA (1994), Rice (1994), and NAS (1987)

U.S. EPA. 1995a. *Further Issues for Modeling the Indirect Exposure Impacts from Combustor Emissions.* Office of Research and Development. Washington, D.C. January.

This document recommends using *BCF* for the metals cadmium, mercury, selenium, and zinc, presented in U.S. EPA (1993), to calculate Ba_{beef} values for these metals. Specifically, the *BCFs* from U.S. EPA (1993)—which are in units of kilogram feed DW/kilogram tissue DW are divided by a feed ingestion rate of 20 kilograms DW/day to arrive at Ba_{beef} values in units of day/kilogram tissue DW, according to the methodology developed for dioxins (U.S. EPA 1992).

U.S. EPA. 1995b. “Waste Technologies Industries Screening Human Health Risk Assessment (SHHRA): Evaluation of Potential Risk from Exposure to Routine Operating Emissions.” Volume V. External Review Draft. U.S. EPA Region 5, Chicago, Illinois.

U.S. EPA. 1997a. *Exposure Factors Handbook.* “Food Ingestion Factors”. Volume II. SAB Review Draft. EPA/600/P-95/002F. August.

TABLE B-3-10

BEEF CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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U.S. EPA. 1997b. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends an F_i value of 1; this value assumes that 100 percent of the plant materials ingested by cattle have been grown on soil contaminated by emissions.

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 7)

Description

This equation first estimates the daily amount of COPCs taken in by cattle through the ingestion of contaminated plant and soil material. The equation then recommends the use of biotransfer factors to transform the daily animal intake of a COPC (mg COPC/day) into an animal (dairy cattle) milk COPC concentration (mg COPC/kg FW tissue).

The limitations and uncertainty introduced in calculating this variable include the following:

(1) Variables P_i and Cs are COPC- and site-specific. Uncertainties associated with these variables are site-specific.
(2) Uncertainties associated with the variables F_i, Qs, and Qp_i are expected to be minimal.
(3) Ba_{milk} values may not reflect site-specific conditions—Ba_{milk} values for nondioxin-like organics are based on a generalized regression equation; Ba_{milk} values for dioxins and furans are estimated on the basis of experimental values from a single lactating cow; and Ba_{milk} values for inorganics are based on integration of a wide variety of empirical and experimental result which can mean that site-specific difference are ignored.

Based on the information below, A_{milk} is dependent on the concentrations of COPCs estimated in plant feeds and soil, and the biotransfer factor estimated for each compound.

Equation

$$A_{milk} = \left(\sum \left(F_i \cdot Qp_i \cdot P_i \right) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{milk} \cdot MF$$

For mercury modeling, calculate milk concentrations due to plant and soil ingestion for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective P_i, Cs, and Ba_{milk} values.

Variable	Description	Units	Value
A_{milk}	Concentration of COPC in milk	mg COPC/kg FW tissue	
F_i	Fraction of plant type (*i*) grown on contaminated soil and ingested by the animal	unitless	**1.0** This variable is site- and plant type-specific. Plant types for cattle are identified as grain, forage, and silage. We recommend using a default value of 1.0 for all plant types. This is consistent with U.S. EPA (1998), U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), which recommend assuming that 100 percent of the plant materials ingested by cattle were grown on soil contaminated by emissions. The following uncertainty is associated with this variable: (1) Assuming 100 percent of the plant materials eaten by cattle were grown on soil contaminated by facility emissions may overestimate A_{milk}.

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 7)

Variable	Description	Units	Value
Qp_i	Quantity of plant type (i) ingested by the animal per day	kg DW plant/day	**Forage: 13.2** **Silage: 4.1** **Grain: 3.0** This variable is site- and plant type-specific. Plant types for cattle are identified as grain, forage, and silage. We recommend that you use the following Qp values when evaluating cattle raised by milk farmers: forage (13.2), silage (4.1), and grain (3.0). The recommended plant type-specific Qp_i values were calculated as follows. ***First***, total dry matter intake (DMI) was estimated as 20 kg DW/day, based on information presented in NAS (1987). ***Second***, data from Boone et al. (1981) were used to separate the total DMI into plant type-specific fractions. ***Finally***, the recommended plant type-specific Qp_i values were calculated by multiplying the estimated total DMI (20 kg DW/day) by the plant type-specific fractions. For example, the Qp_i for forage was calculated as 20 kg DW/day · 0.65 = 13.2 kg DW/day. These values are consistent with U.S. EPA (1993; 1994b; 1995), and NC DEHNR (1997). These reference documents cite Boone et al. (1981), NAS (1987), McKone and Ryan (1989), and Rice (1994) as primary references for plant ingestion rates. Uncertainties introduced by this variable include the following: (1) The plant type-specific Qp_i values were calculated based on a total DMI of 20 kg DW/day (NAS 1987) rather than the total DMI of 17 kg DW/day presented in Boone et al. (1981) and McKone and Ryan (1989). Site-specific total DMI values may vary. (2) The plant type-specific fractions calculated from Boone et al. (1981) may not accurately represent site-specific or local plant type-specific fractions.

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 7)

Variable	Description	Units	Value
P_i	Concentration of COPC in plant type *(i)* ingested by the animal	mg/kg DW	**Varies** This variable is COPC-, site-, and plant type-specific; plant types for cattle are identified as grain, forage, and silage. Values for *Pd*, *Pv*, and *Pr* are calculated by using the equations in Tables B-3-7, B-3-8, and B-3-9, and then summed for each plant type to determine P_i. Uncertainties introduced by this variable include the following: (1) Some of the variables in the equations in Tables B-3-7, B-3-8, and B-3-9—including *Cs, Cyv, Q, Dydp*, and *Dywp*—are COPC- and site-specific. Uncertainties associated with these variables are site-specific. (2) In the equation in Table B-3-7, uncertainties associated with other variables include the following: F_w (values for organic compounds estimated on the basis of the behavior of polystyrene micro spheres), *Rp* (estimated on the basis of a generalized empirical relationship), *kp* (estimation process does not consider chemical degradation), and *Yp* (estimated on the basis of national harvest yield and area planted values). All of these uncertainties contribute to the overall uncertainty associated with P_i. (3) In the equation in Table B-3-8, COPC-specific *Bv* values for nondioxin-like compounds may be overestimated by up to one order of magnitude, based on experimental conditions used to develop the algorithm to estimate *Bv* values. (4) In the equation in Table B-3-9, COPC-specific plant-soil biotransfer factors (*Br*) may not reflect site-specific conditions. This may be especially true for inorganic COPCs for which estimates of *Br* would be more accurately estimated by using plant uptake response slope factors.
Qs	Quantity of soil ingested by the animal	kg/day	**0.4** This variable is site-specific. We recommend using a soil ingestion rate of 0.4 kg/day. This is consistent with NC DEHNR (1997) and U.S. EPA (1994b), which cite USDA (1994), Rice (1994), and NAS (1987). Briefly, the recommended *Qs* value was calculated as follows. ***First***, a total DMI was estimated as 20 kg DW/day based on information presented in NAS (1987). ***Second***, USDA (1994) estimates that *Qs* equals 2 percent of the total DMI. ***Finally***, the recommended *Qs* value was calculated as 20 kg DW/day · 0.02 = 0.4 kg DW /day. Uncertainties introduced by this variable include: (1) The recommended *Qs* value was based on a total DMI of 20 kg DW/day NAS (1987) rather than the total DMI of 17 kg DW/day presented in Boone et al. (1981) and McKone and Ryan (1989). To the extent that site-specific or local total DMI values may vary, A_{milk} may be under- or overestimated to a limited degree. (2) USDA (1994) states that *Qs* equals 2 percent of the total DMI for dairy cattle on a farm. Although the basis of the estimate of 2 percent is not known, it is apparent that to the extent that site-specific or local *Qs* values are different than 2 percent, A_{milk} may be under- or overestimated to some degree.

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 7)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-3-1. Uncertainties are site-specific.
Bs	Soil bioavailability factor	unitless	**1.0** The soil bioavailability factor, *Bs,* can be thought of as the ratio between bioconcentration (or biotransfer) factors for soil and vegetation for a given COPC. The efficiency of transfer from soil may differ from efficiency or transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, then this ratio would be less than 1.0. If it is equal or greater than that of vegetation, the *Bs* would be equal to or greater than 1.0. Due to limited data regarding bioavailability from soil, we recommend a default value of 1.0 for *Bs,* until more COPC-specific data is available for this parameter. Some COPCs may be much less bioavailable from soil than from plant tissues. This uncertainty may overestimate *Bs*.
Ba_{milk}	Biotransfer factor for milk	day/kg FW tissue	**Varies** This variable is COPC-specific. A detailed discussion of this variable and COPC-specific values are presented in Appendix A-2. Ba_{milk} is defined as the ratio of the COPC concentration in milk (mg COPC/kg tissue) to the daily intake of the COPC (mg COPC/day) by the animal. Uncertainties introduced by this variable include the following: (1) We recommend using the regression equation developed by RTI (2005) to calculate Ba_{milk} values for organic COPCs. Uncertainties listed in RTI (2005) in deriving the regression equation include: 1) the necesssity to extrapolate data to steady state conditions, 2) metabolism may not be accounted for equally for all data points, and 3) there is a +/- 0.5 variability in measured Log Kow values available in the literature. In addition, values calculated by using this regression equation may not accurately represent the behavior of organic COPCs under site-specific conditions. Ba_{milk} and subsequent A_{milk} values may therefore be under- or overestimated to some degree. (2) We recommend calculating Ba_{milk} values for metals be using single COPC-specific uptake factors developed by Baes et al. (1984). These uptake factors may not accurately represent the behavior of inorganic COPCs under site-specific conditions. Ba_{milk} and subsequent A_{milk} values may therefore be under- or overestimated to some degree.

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 7)

Variable	Description	Units	Value
MF	Metabolism factor	unitless	**0.01 and 1.0** This variable is COPC-specific. Based on a study by Ikeda et al. (1980), U.S. EPA (1995a) recommended using a metabolism factor to account for metabolism in animals to offset the amount of bioaccumulation suggested by biotransfer factors. MF applies only to beef, milk, and pork. It does not apply to direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish. U.S. EPA (1995b) recommended an MF of 0.01 for bis(2-ethylhexyl)phthalate (BEHP) and 1.0 for all other COPCs.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture.* Oak Ridge National Laboratory, Oak Ridge, Tennessee.

U.S. EPA (1994c) recommends Baes et al. (1984) as a source of (1) Ba_{milk} values for inorganics, and (2) water content of 0.9 for cow's milk, which can be used to convert Ba_{milk} values in dry weight to wet weight.

Beecher, G.D., and C.C. Travis. 1989. *Modeling Support for the RURA and Municipal Waste Combustion Project Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model*. Prepared under IAG-1824-A020-A1 by Oak Ridge National Laboratory for U.S. EPA Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office. Cincinnati, Ohio.

This document was cited by U.S. EPA (1990) as the source of Ba_{milk} values for cadmium.

Boone, F.W., Yook C. Ng, and John M. Palms. 1981. "Terrestrial Pathways of Radionuclide Particulates." *Health Physics*. Vol. 41, No. 5, pages 735-747. November.

This document is identified as a source of plant ingestion rates. Boone et al. (1981) reports a total forage, grain, and silage ingestion rate of 17 kg DW/day for dairy cattle. Also, this document states that this total DMI of 17 kg DW/day is made up of the following plant type-specific fractions: forage (65 percent), grain (15 percent), and silage (20 percent).

Ikeda, G.J., P.P. Sapenza, and J.L. Couvillion. 1980. "Comparative distribution, excretion, and metabolism of di(2-ethylhexyl)phthalate in rats, dogs, and pigs." *Food Cosmet. Toxicology.* 18:637-642.

McKone, T.E., and P.B. Ryan. 1989. *Human Exposures to Chemicals Through Food Chains: An Uncertainty Analysis.* Livermore, California: Lawrence Livermore National Laboratory Report. UCRL-99290.

This document is cited as a source of plant ingestion rates. It reports an average total ingestion rate of 17 kg dry weight/day for the three plant feeds, which is consistent with the total recommended by Boone et al. (1981) for cattle.

NAS. 1987. *Predicting Feed Intake of Food-Producing Animals.* National Research Council, Committee on Animal Nutrition. Washington, D.C.

NC DEHNR (1997) and U.S. EPA (1994c) note that this document reports a daily DMI equal to 3.2 percent of an average dairy cattle body weight of 630 kilograms; this results in a daily DMI of 630 kg DW · 0.032 = 20.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

Grains such as corn may be grown specifically as cattle feed. COPC uptake into these feed materials may occur through root uptake, wet and dry deposition of particulate-bound COPCs on plants, and vapor-phase uptake of COPCs through plant foliage. Plants are classified as "protected" if they have an outer covering that acts as a barrier to direct deposition and vapor uptake of air contaminants. NC DEHNR (1997) classifies grains as protected, and recommends that only root uptake of COPCs be evaluated for grains. Because silage may consist of forage materials that have been stored and fermented, it should be treated as forage (that is, as unprotected).

This document is a reference source for the equation in Table B-3-11. This document also recommends the following:

(1) An F_i value of 1
(2) Forage, silage, and grain ingestion rates (Qp_i) of 13.2, 4.1, and 3.0 kg DW/day for dairy farmer cattle, respectively, based on a total DMI of 20 kg DW/day calculated from NAS (1987) and plant type-specific fractions from Boone et al. (1981)
(3) Forage, silage, and grain ingestion rates (Qp_i) of 6.2, 1.9, and 12.2 kg DW/day, respectively for typical dairy farmer cattle based on USDA (1994)
(4) A Qs value of 0.4 kg/day, based on NAS (1987) and USDA (1994)
(5) Ba_{milk} values ranging from 3.5 x 10^{-10} to 4.8, based on Baes et al. (1984).

Research Triangle Institute (RTI). 2005. *Methodology for Predicting Cattle Biotransfer Factors.* Prepared for U.S. Environmental Protection Agency (EPA) Office fo Solid Waste. EPA Contract No. 68-W-03-042. August.

USDA. 1994. Personal Communication Regarding Soil Ingestion Rate for Dairy Cattle. Between G.F. Fries, Agricultural Research Service, and Glenn Rice and Jennifer Windholtz, U.S. EPA, Office of Research and Development. March 22.

NC DEHNR (1997) and EPA (1994c) note that USDA (1994) reports soil ingestion to be 2 percent of the total DMI for dairy cattle on farms.

U.S. EPA. 1994a. *Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Solid Waste and Emergency Response. EPA-530-R-94-021. April.

This document recommends a F_i value of 1, assuming that 100 percent of the plant materials ingested by cattle have been grown on soil contaminated by combustion unit emissions.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is a reference source for the equation in Table B-3-11. This document also recommends the following:

(1) An F_i value of 1
(2) A forage ingestion rate (Qp_i) value of 13.2 kg DW/day, from NAS (1987) and Boone et al. (1981)
(3) A quantity of soil ingested (Qs) value of 0.4 kg/day, based on NAS (1987) and USDA (1994)
(4) Ba_{milk} values ranging from 3.5 x 10^{-10} to 4.8, based on Baes et al. (1984)

TABLE B-3-11

MILK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

U.S. EPA. 1994c. *Estimating Exposures to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures*. External Review Draft. Office of Research and Development. EPA/600/6-88/005Cc. June.

This document reported bioconcentration factors for dioxin-like compounds (dioxin and furan congeners) calculated on the basis of experimental data derived by McLachlan et al. (1990).

U.S. EPA. 1995a. *Further Issues for Modeling the Indirect Exposure Impacts from Combustor Emissions.* Office of Research and Development. Washington, D.C. January.

U.S. EPA. 1995b. "Waste Technologies Industries Screening Human Health Risk Assessment (SHHRA): Evaluation of Potential Risk from Exposure to Routine Operating Emissions." Volume V. External Review Draft. U.S. EPA Region 5, Chicago, Illinois.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends an F_i value of 1; this value assumes that 100 percent of the plant materials ingested by cattle have been grown on soil contaminated by emissions.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 7)

Description

This equation first estimates the daily intake of COPCs through the ingestion of contaminated plant and soil material. The equation uses biotransfer factors to transform the daily animal intake of a COPC (mg COPC/day) into an animal COPC tissue concentration (mg COPC/kg tissue).

The limitations and uncertainty introduced in calculating this variable include the following:

(1) Uncertainties associated with the variables P_i and Cs are COPC- and site-specific.
(2) Uncertainties associated with the variables F_i, Q, and Qp_i are expected to be minimal.
(3) Uncertainties associated with Ba_{pork} values may be significant for two primary reasons: (a) Ba_{pork} for dioxins are calculated from Ba_{milk} values that are based on cattle metabolism of dioxins rather than a sow metabolism, and (b) the source or methodology used to calculate the Ba_{pork} values for organics other than dioxins and inorganics other than cadmium, mercury, selenium, and zinc as reported in NC DEHNR (1997) is not known. Therefore, the magnitude and direction of the associated uncertainties cannot be specified.

Based on the information below, A_{pork} is dependent on the concentrations of COPCs estimated in plant feeds and soil, and the biotransfer factor estimated for each COPC.

Equation

$$A_{pork} = \left(\sum \left(F_i \cdot Qp_i \cdot P_i\right) + Qs \cdot Cs \cdot Bs\right) \cdot Ba_{pork} \cdot MF$$

For mercury modeling, pork concentration due to plant and soil ingestion is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective P_i, Cs, and Ba_{pork} values.

Variable	Description	Units	Value
A_{pork}	Concentration of COPC in pork	mg COPC/kg FW tissue	
F_i	Fraction of plant type *(i)* grown on contaminated soil and ingested by the animal	unitless	**1.0** This variable is site- and plant type-specific; plant types for swine are typically identified as grain and silage. We recommend using a default value of 1.0 for all plant types. This is consistent with U.S. EPA (1998; 1994a; 1994c), and NC DEHNR (1996), which recommend assuming that 100 percent of the plant materials ingested by swine were grown on soil contaminated by emissions. The following uncertainty is associated with this variable: Assuming 100 percent of the plant materials ingested by swine were grown on soil contaminated by facility emissions may overestimate A_{pork}.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 7)

Variable	Description	Units	Value
Qp_i	Quantity of plant type (*i*) ingested by the animal each day	kg DW plant/day	**Silage: 1.4** **Grain: 3.3** This variable is site- and plant type-specific; plant types for swine are typically identified as grain and silage. We recommend that you use the following *Qp* values when evaluating swine raised by farmers: silage (1.4) and grain (3.3). These Qp_i values are based on a total DMI value of 4.7 kg DW/day, and plant type-specific diet fractions (70 percent grain and 30 percent silage) are based on U.S. EPA (1982). NC DEHNR (1997) and U.S. EPA (1990) recommend silage and grain ingestion rates of 1.3 and 3.0 kg dry/day, respectively, for swine. NC DEHNR (1997) references U.S. EPA (1990) as the source of these ingestion rates. The difference between the default Qp_i values and values recommended by NC DEHNR (1997) and U.S. EPA (1990) is the total DMI upon which they are based. Specifically, we recommend the use of a total DMI for swine of 4.7 kg DW/day, based on U.S. EPA (1995), whereas NC DEHNR (1997) and U.S. EPA (1990) recommend a total DMI of 4.3 kg dry weight/day. NC DEHNR (1997) and U.S. EPA (1990) don't differentiate between subsistence and typical hog farmers as they do for cattle, because it is assumed that forage is not a significant portion of a hog's diet. Uncertainties introduced by this variable include the following: The recommended grain and silage ingestion rates may not accurately represent site-specific or local conditions. Therefore, Qp_i and A_{pork} values may be under- or overestimated to some degree.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 7)

Variable	Description	Units	Value
P_i	Concentration of COPC in plant type (*i*) ingested by the animal	mg/kg DW	**Varies** This variable is COPC-, site-, and plant type-specific; plant types for swine are identified as grain and silage. Values for *Pd*, *Pv*, and *Pr* are calculated by using the equations in Tables B-3-7, B-3-8, and B-3-9; and then summed for each plant type to determine P_i. Uncertainties introduced by this variable include the following: (1) Some of the variables in the equations in Tables B-3-7, B-3-8, and B-3-9—including *Cs, Cyv, Q, Dydp*, and *Dywp*—are COPC- and site-specific. Uncertainties associated with these variables are site-specific. (2) In the equation in Table B-3-7, uncertainties associated with other variables include: F_w (values for organic compounds based on behavior of polystyrene micro spheres), *Rp* (estimated on the basis of a generalized empirical relationship), *kp* (estimation process does not consider chemical degradation) and *Yp* (estimated based on national harvest yield and area planted values). All of these uncertainties contribute to the overall uncertainty associated with P_i. (3) In the equation in Table B-3-8, COPC-specific *Bv* values for nondioxin-like compounds may be overestimated by up to one order of magnitude, based on experimental conditions used to develop the algorithm to estimate *Bv* values. (4) In the equation in Table B-3-9, COPC-specific soil-to-plant biotransfer factors (*Br*) may not reflect site-specific conditions. This may be especially true for inorganic COPCs for which estimates of *Br* would be accurately estimated by using plant uptake response slope factors.
Qs	Quantity of soil ingested by the animal	kg/day	**0.37** This variable is site-specific. We recommend using the soil ingestion rate of 0.37 kg/day. NC DEHNR (1997) recommended a soil ingestion rate for swine of 0.37 kg/day. This is estimated by assuming a soil intake of 8 percent of the total DMI. NC DEHNR (1997) does not specify the total DMI used to estimate *Qs*. However, mathematically, *Qs* appears to be based on a total DMI of 4.7 kg DW/day ($4.7 \cdot 0.08 = 0.37$), which is consistent with U.S. EPA (1995). The following uncertainty is associated with this variable: The recommended soil ingestion rate may not accurately represent site-specific or local conditions. Therefore, *Qs* and A_{pork} values, may be under- or overestimated to some degree.
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-3-1. Uncertainties are site-specific.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 7)

Variable	Description	Units	Value
Bs	Soil bioavailability factor	unitless	**1.0** The soil bioavailability factor, *Bs,* can be thought of as the ratio between bioconcentration (or biotransfer) factors for soil and vegetation for a given COPC. The efficiency of transfer from soil may differ from efficiency or transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, then this ratio would be less than 1.0. If it is equal or greater than that of vegetation, the *Bs* would be equal to or greater than 1.0. Due to limited data regarding bioavailability from soil, we recommend a default value of 1.0 for *Bs,* until more COPC-specific data is available for this parameter. Some COPCs may be much less bioavailable from soil than from plant tissues. This uncertainty may overestimate *Bs*.
Ba_{pork}	Biotransfer factor for pork	day/kg FW tissue	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Ba_{pork} is defined as the ratio of the COPC concentration in animal tissue (mg COPC/kg FW tissue) to the daily intake of the COPC (mg COPC/day) by the animal. Uncertainties introduced by this variable include the following: (1) We recommend calculating Ba_{pork} values for organic COPCs from Ba_{beef} values, assuming that pork is 23 percent fat and beef is 19 percent fat. Values derived this way might not accurately represent the behavior of organic COPCs under site-specific conditions. Ba_{pork} and consequent A_{pork} estimates may be under- or overestimated to some degree. (2) The sources or method used to support or estimate Ba_{pork} values presented in NC DEHNR (1997) are not known. Therefore, the degree to which these values represent the behavior of COPCs under site-specific conditions cannot be determined.
MF	Metabolism factor	unitless	**0.01 and 1.0** This variable is COPC-specific. Based on a study by Ikeda et al. (1980), U.S. EPA (1995a) recommended using a metabolism factor to account for metabolism in animals to offset the amount of bioaccumulation suggested by biotransfer factors. MF applies only to beef, milk, and pork. It does not apply to direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish. U.S. EPA (1995b) recommends an MF of 0.01 for bis(2-ethylhexyl)phthalate (BEHP) and 1.0 for all other COPCs.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 7)

REFERENCES AND DISCUSSION

Boone, F.W., Yook C. Ng, and John M. Palms. 1981. "Terrestrial Pathways of Radionuclide Particulates." *Health Physics*, Vol. 41, No. 5, pp. 735-747. November.

This document is cited as the source of a total DMI for hogs of 3.4 kg DW/day.

Ikeda, G.J., P.P. Sapenza, and J.L. Couvillion. 1980. "Comparative distribution, excretion, and metabolism of di(2-ethylhexyl)phthalate in rats, dogs, and pigs." *Food Cosmet. Toxicology.* 18:637-642.

NAS. 1987. *Predicting Feed Intake of Food-Producing Animals*. National Research Council, Committee on Animal Nutrition, Washington, D.C.

This document presents a total DMI for lactating sows of 5.2 kg DW/day. This document is also cited by U.S. EPA (1995) as the source of a total DMI for swine of 4.7 kg DW/day. As presented in this document, the value of 4.7 kg DW/day represents the average of specific total DMI values for gilts (young sows) and for lactating sows.

Ng, Y.C., C.S. Colsher, and S.E. Thomson. 1982. *Transfer Coefficients for Assessing the Dose from Radionuclides in Meat and Eggs*. U.S. Nuclear Regulatory Commission. Final Report. NUREG/CR-2976.

This document is cited as the source of biotransfer factors (Ba_{pork}) for several inorganic COPCs. However, U.S. EPA (1995) notes that "a large degree of uncertainty" exists in many of the experiments used in this document to develop the biotransfer factors. The biotransfer factors developed by Ng, Colsher, and Thompson (1982) are not recommended for use by U.S. EPA .

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

Grains such as corn may be grown specifically as swine feed. COPC uptake into these feed materials may occur through root uptake, wet and dry deposition of particulate-bound constituents on plants, and vapor-phase uptake of COPCs through plant foliage. Plants are classified as "protected" if they have an outer covering that acts as a barrier to direct deposition and vapor uptake of air contaminants. NC DEHNR (1997) classifies grains as protected, and recommends that only root uptake of COPCs be evaluated for grains; because silage may consist of forage materials that have been stored and fermented, it should be treated as forage (that is, as unprotected).

This document also recommends the following:

- An F_i value of 1, assuming that 100 percent of the plant material eaten by swine have been grown on soil contaminated by combustion unit emissions.
- Plant type-specific Qp_i values for hogs of 3.0 and 1.3 kg DW/day for grain and silage, respectively. This document cites U.S. EPA (1990) as the source of these ingestion rates.
- A quantity of soil ingested (Qs) value of 0.37 kg DW/day. This value is calculated as 8 percent of the total DMI (U.S. EPA 1993a). The total DMI of 4.3 kg DW/day comes from U.S. EPA (1990).
- A range of Ba_{pork} values (1.3×10^{-09} to 5.8 day/kg wet tissue); however, the sources of or methodology used to estimate, these values are not identified.

Pennington, J.A.T. 1989. *Food Values of Portions Commonly Used*. 15th ed. Harper and Row. New York.

Cited by NC DEHNR (1997)—actually NC DEHNR (1997) cities "Pennington (1993)" but presents only this document (Pennington 1989) in the reference section—for the estimated fat content of pork, 23 percent.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

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U.S. EPA. 1982. "Pesticides Assessment Guidelines Subdivision O." Residue Chemistry. Office of Pesticides and Toxic Substances, Washington, D.C. EPA/540/9-82-023.

This document is cited by U.S. EPA (1990) as the source of the assumption that 70 percent of the total DMI for swine is grain and 30 percent is silage.

U.S. EPA. 1990. *Interim Final Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Environmental Criteria and Assessment Office. Office of Research and Development. EPA-600-90-003. January.

This document represents total dry matter intake (DMI) rates for hogs and lactating sows of 3.4 and 5.2 kg DW/day, respectively, and recommends the average of these two rates (4.3 kg DW/day) as the total DMI. U.S. EPA (1990) cites Boone et al. (1981) as the source of the hog ingestion rate and NAS (1987) as the source of the lactating sow ingestion rate.

This document then assumes that 70 percent of the total DMI for swine is grain and 30 percent is silage; fractions then are used to arrive at the recommended grain ingestion rate of 3.0 kg DW/day (4.3 kg DW/day · 0.70) and the recommended silage ingestion rate of 1.3 kg DW/day (4.3 kg DW/day · 0.30). U.S. EPA (1990) cites U.S. EPA (1982) as the source of the grain and silage fractions.

This document also recommends an F_i value of 1. This assumes that 100 percent of the plant material eaten by swine is grown on soil contaminated by combustion unit emissions.

U.S. EPA. 1992. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* External Review Draft. Office of Research and Development. Washington, D.C. November.

This document recommends that the quantity of soil (*Qs*) eaten by swine be estimated as 8 percent of the total DMI. This document states "Fries of USDA notes pigs exhibit 'rooting' behavior and assumes a maximum soil ingestion intake of 8 percent of dry matter based on a 2 to 8 percent range noted in his earlier PCB work." However, this document provides no citations of work performed by Fries or personal communications with Fries.

U.S. EPA. 1994a. *Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Solid Waste and Emergency Response. EPA-530-R-94-021. April.

This document recommends an F_i value of 1. This assumes that 100 percent of the plant material ingested by swine has been grown on soil contaminated by combustion unit emissions.

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document states that milk is 3.5 percent fat. This document also uses experimental data derived by McLachlon, et al. (1990) to calculate biotransfer factors with units of (kg feed/kg tissue).

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document recommends an F_i value of 1. This assumes that 100 percent of the plant material eaten by swine has been grown on soil contaminated by combustion unit emissions.

TABLE B-3-12

PORK CONCENTRATION DUE TO PLANT AND SOIL INGESTION (CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 7 of 7)

U.S. EPA. 1995a. *Further Issues for Modeling the Indirect Exposure Impacts from Combustor Emissions.* Office of Research and Development. Washington, D.C. January 20.

This document calculates Ba_{pork} values for cadmium, mercury, selenium, and zinc by dividing uptake slope factors ([mg COPC/kg tissue DW]/[mg COPC/kg feed DW]) from U.S. EPA (1993b) - 0.003 (cadmium), 0.0234 (mercury), 2.94 (selenium), and 0.002 (zinc)—by a daily feed ingestion rate for pork of 4.7 kg DW/day (NAS 1987). This approach is similar to that recommended by U.S. EPA (1994b) for dioxins. The calculated biotransfer factors are 6.0×10^{-04} (cadmium); 0.0051 (mercury); 6.255×10^{-01} (selenium); and 4.0×10^{-04} (zinc).

U.S. EPA. 1995b. "Waste Technologies Industries Screening Human Health Risk Assessment (SHHRA): Evaluation of Potential Risk from Exposure to Routine Operating Emissions." Volume V. External Review Draft. U.S. EPA Region 5, Chicago, Illinois.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-3-13

COPC CONCENTRATION IN EGGS
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC concentration in eggs due to ingestion of contaminated soil and grain by home grown chickens that have access to soil.

Uncertainties associated with this equation include the following:

(1) This pathway has typically been applied only to PCDDs and PCDFs. However, concentrations in chicken eggs for other organics and metals can be calculated using biotransfer factors in an approach similar to that used to calculate concentrations in animal tissue.
(2) Assuming that 10 percent of a chicken's diet is soil may not represent site-specific conditions. Stephens et al. (1995) suggested that the percentage of soil in the diet of chickens raised under field conditions may be greater than 10 percent. Therefore, the concentration of COPCs in eggs, A_{egg}, may be underestimated.
(3) Estimated COPC-specific soil-to-plant biotransfer factors (Br) may not reflect site-specific or local conditions. Therefore, estimates of Pr and A_{egg} may be under- or overestimated to some degree.
(4) The recommended *BCFs* used in calculating Ba_{egg} may not accurately represent the behavior of COPCs under site-specific and local conditions. For example, Stephens et al. (1995) noted that chickens raised under field conditions showed larger apparent *BCFs*. Therefore, the recommended *BCFs* may underestimate the concentration of COPCs in eggs, A_{egg}.
(5) The recommended *BCFs* are based on incomplete experimental results. Stephens et al. (1995) presented complete experimental results. This study included results from a high-dose group and a low-dose group; results were based on the full exposure period. A brief comparison of the results from Stephens et al. (1992) with those from Stephens et al. (1995) indicates that *BCFs* from the high-dose group are generally higher than *BCFs* from the low-dose group. Therefore, using the currently recommended *BCFs* may underestimate the COPC concentration in eggs, A_{egg}.

Equation

$$A_{egg} = \left(\sum \left(F_i \cdot Qp_i \cdot P_i \right) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{egg}$$

For mercury modeling, the concentration of COPC in eggs is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective P_i, Cs, and Ba_{eggs} values.

Variable	Description	Units	Value
A_{egg}	Concentration of COPC in eggs	mg COPC/kg FW tissue	

TABLE B-3-13

COPC CONCENTRATION IN EGGS
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
F_i	Fraction of plant type *i* (grain) grown on contaminated soil and ingested by the animal	unitless	**1.0** This variable is site- and plant type-specific. F_i for chickens is estimated for grain feed only. We recommend using a default value of 1.0. This is consistent with U.S. EPA (1998), U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), which recommend assuming that 100 percent of the plant materials ingested were grown on soil contaminated by facility emissions. The following uncertainty is associated with this variable: Assuming that 100 percent of the plant material eaten by chickens was grown on soil contaminated by emissions may overestimate A_{egg}.
Qp_i	Quantity of plant type *i* (grain) ingested by the animal	kg DW plant/day	**0.2** Qp_i for chicken is estimated for grain feed only, as recommended by NC DEHNR (1997). Uncertainties associated with this variable include the following: Actual grain ingestion rates can vary from site to site; this can over- or underestimate Qp_i.
P_i	Concentration of COPC in plant type *I* (grain)	mg COPC/kg DW	**Varies** This variable is COPC-, site-, and plant type-specific. Calculate values for *Pi* for grain using the equation in Table B-3-9. Uncertainties introduced by this variable include the following: (1) Some of the variables in the equation in Table B-3-9—including *Cs, Cyv, Q, Dydp*, and *Dywp*—are COPC- and site-specific. Uncertainties associated with these variables are site-specific. (2) In the equation in Table B-3-9, COPC-specific plant-soil biotransfer factors (*Br*) may not reflect site-specific conditions. This may be especially true for inorganic COPCs for which *Br's* would be more accurately estimated by using plant uptake response slope factors.
Qs	Quantity of soil ingested by the animal	kg/day	**0.022** This variable is site-specific. We recommend that the soil ingestion rate of 0.022 kg/day be used. This is consistent with Stephens et al. (1995). Uncertainties introduced by this variable include the following: (1) The recommended soil ingestion rate may not accurately represent site-specific or local conditions. (2) Empirical data to support soil ingestion rates of chickens are limited.
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-3-1. Uncertainties are site-specific.

TABLE B-3-13

COPC CONCENTRATION IN EGGS
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
Bs	Soil bioavailability factor	unitless	**1.0** The soil bioavailability factor, Bs, can be thought of as the ratio between bioconcentration (or biotransfer) factors for soil and vegetation for a given COPC. The efficiency of transfer from soil may differ from efficiency or transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, than this ratio would be less than 1.0. If it is equal or greater than that of vegetation, the Bs would be equal to or greater than 1.0. Due to limited data regarding bioavailability from soil, we recommend a default value of 1.0 for Bs, until more COPC-specific data is available for this parameter. Some COPCs may be much less bioavailable from soil than from plant tissues. This uncertainty may overestimate Bs.
Ba_{egg}	Biotransfer factor for chicken eggs	day/kg FW tissue	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainties are associated with this variable: (1) We recommend calculating Ba_{egg} values for organic COPCs other than dioxins and furans by using the regression equation developed on the basis of a study of 29 organic compounds. Values calculated by using this regression equation may not accurately represent the behavior of organic COPCs under site-specific conditions. Therefore, estimates of Ba_{egg} and, therefore, A_{egg} may be under- or overestimated to some degree. (2) The recommended *BCFs* may not accurately represent the behavior of COPCs under site-specific or local conditions. For example, Stephens et al. (1995) noted that chickens raised under field conditions, and which probably had a more than 10 percent soil in their diet, showed larger apparent *BCFs*. Therefore, the recommended *BCFs* may underestimate the concentration of COPCs in eggs, A_{egg}. (3) The recommended *BCFs* are based on incomplete experimental results. Stephens et al. (1995) include results from a high-dose group and as a low-dose group; results are based on the full exposure period. A brief comparison of the results from Stephens et al. (1992) and those from Stephens et al. (1995) indicates that *BCFs* from the high-dose group are generally higher than *BCFs* from the low-dose group. Therefore, using the currently recommended *BCFs* may underestimate the COPC concentration in eggs, A_{egg}.

REFERENCES AND DISCUSSION

California Environmental Protection Agency (CEPA). 1993. " Parameter Values and Ranges for CALTOX." Draft. Office of Scientific Affairs. California Department of Toxics Substances Control. Sacramento, CA. July.

Chang, R.R., D. Hayward, L. Goldman, M. Harnly, J. Flattery, and R.D. Stephens. 1989. "Foraging Farm Animals as Biomonitors for Dioxin Contamination." *Chemosphere.* Volume 19: 481-486.

This document appears to be cited by Stephens et al. (1992) as support for the assumption that soil represents 10 percent of the diet of free-range chickens.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is a reference source for the equation in Table B-3-13. This document also cites Stephens et al. (1992) as the source of estimates of the fraction of diet that is soil (*Fd*), and BCF_{egg} for dioxins and furans.

Petreas, M.X., L.R. Goldman, D.G. Hayward, R. Chang, J. Flattery, T. Wiesmuller, R.D. Stephens, D.M. Fry, and C. Rappe. 1991. "Biotransfer and Bioaccumulation of PCDD/PCDFs from Soils: Controlled Exposure Studies of Chickens." *Chemosphere.* Volume 23: 1731-1741.

This document appears to be cited by Stephens et al. (1992) and Stephens et al. (1995) as support for the assumption that soil represents 10 percent of the diet of free-range chickens.

Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1992. "Biotransfer and Bioaccumulation of Dioxins and Dibenzofurans from Soil." Hazardous Materials Laboratory, California Department of Health Services. Berkeley, California.

This document is cited as the source of the assumption that free- range chickens ingest soil as 10 percent of their diet and as the source of the dioxin and furan congener-specific *BCFs*. However, this document does not clearly reference or document the assumption that soil represents 10 percent of a free-range chicken diet. The document appears to cite two other documents as supporting this assumption, Chang, Hayward, Goldman, Harnly, Flattery, and Stephens (1989) and Petreas, Goldman, Hayward, Chang, Flattery, Wiesmuller, Stephens, Fry, and Rappe (1992). Also, this document presents dioxin and furan congener-specific *BCFs* (egg yolk) for the low-exposure group after 80 days of a 178-day exposure period. The chickens in the low-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 42 parts per trillion (ppt) I-TEQ. Chickens in the high-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 458 ppt I-TEQ; *BCF* results were not presented for this group.

Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1995. "Biotransfer and Bioaccumulation of Dioxins and furans from Soil: Chickens as a Model for Foraging Animals." *The Science of the Total Environment.* Volume 175: 253-273.

This document is an expansion of the results originally presented in Stephens et al. (1992). In particular, this document suggests that the percentage of soil in the diet of chickens raised under field conditions is likely to be greater than 10 percent, the value that was used in the experimental study presented in this document.

This document also presents dioxin and furan congener-specific *BCFs* (egg yolk) under two exposure schemes: low exposure and high exposure. The white leghorn (Babcock D 300) chickens in the low group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 42 ppt I-TEQ. Chickens in the high group were fed a diet consisting of 10 percent soil with a PCDD/PCDF concentration of 460 ppt I-TEQ (some congeners were fortified by spiking). The *BCFs* presented for low- and high-dose groups both represent averages of results from Day-80, Day-160, and Day-178 (the end of the exposure duration).

TABLE B-3-13

COPC CONCENTRATION IN EGGS
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1990. *Interim Final Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Environmental Criteria and Assessment Office. Office of Research and Development. EPA/600/6-90/003. January.

This document is a reference source for the equation in Table B-3-9; and an F_i value of 1.0.

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge.* Volumes I and II. EPA 822/R-93-001a. Office of Water. Washington, D.C.

U.S. EPA (1995) recommends that uptake slope factors for the metals cadmium, selenium, and zinc cited by this document be used to derive Ba_{egg} values.

U.S. EPA. 1995. *Further Issues for Modeling the Indirect Exposure Impacts from Combustor Emissions.* Office of Research and Development. Washington, D.C. January 20.

U.S. EPA. 1997a. *Exposure Factors Handbook.* “Food Ingestion Factors”. Volume II. EPA/600/P-95/002F. August.

U.S. EPA. 1997b. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-3-14

CONCENTRATION IN CHICKEN
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC concentration ($A_{chicken}$) in chicken meat due to ingestion of contaminated soil and grain by home grown chickens that have access to soil.

Uncertainties associated with this equation include the following:

(1) This pathway has typically been applied only to PCDDs and PCDFs. However, concentrations in chickens for other organics and metals can be calculated using biotransfer factors using a similar approach as was used to calculate concentrations in other animal tissue.

(2) The assumption that 10 percent of a chicken's diet is soil may not represent site-specific or local conditions of chickens raised on farms. Stephens et al. (1995) suggests that the percentage of soil in the diet of chickens raised under field conditions may be greater than 10 percent. Therefore, the concentration of COPCs in chicken, $A_{chicken}$ may be underestimated.

(3) The recommended *BCFs* are based on incomplete experimental results. Stephens et al. (1995) presents results for a high-dose group and low-dose group (results are based on the full 178-day exposure period). A comparison of the results from Stephens et al. (1992) with those from Stephens et al. (1995) shows that *BCF*s from the high dose group are generally higher than *BCFs* from the low dose group. Therefore, use of the currently recommended *BCFs* may underestimate the COPC concentration in chicken, $A_{chicken}$.

Equation

$$A_{chicken} = \left(\sum_i \left(F_i \cdot Qp_i \cdot P_i \right) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{chicken}$$

For mercury modeling, the concentration of COPC in chicken is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective P_i, Cs, and $Ba_{chicken}$ values.

Variable	Description	Units	Value
$A_{chicken}$	Concentration of COPC in chicken meat	mg COPC/kg FW tissue	
F_i	Fraction of plant type *i* (grain) grown on contaminated soil and ingested by the animal	unitless	**1.0** This variable is site- and plant type-specific. F_i for chickens is estimated for grain feed only. We recommend using a default value of 1.0. This is consistent with U.S. EPA (1998), U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), which recommend assuming that 100 percent of the plant materials ingested were grown on soil contaminated by facility emissions. The following uncertainty is associated with this variable: Assuming that 100 percent of the plant materials eaten by chickens were grown on soil contaminated by facility emissions may overestimate $A_{chicken}$.

TABLE B-3-14

CONCENTRATION IN CHICKEN
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
Qp_i	Quantity of plant type *i* (grain) ingested by the animal	kg DW plant/day	**0.2** Qp_i for chicken is estimated for grain feed only, as recommended by NC DEHNR (1997) and U.S. EPA (1990). Uncertainties associated with this variable include the following: Actual grain ingestion rates can vary from site to site; this can over- or underestimate Qp_i.
P_i	Concentration of COPC in plant type *I* (grain)	mg COPC/kg DW	**Varies** This variable is COPC-, site-, and plant type-specific. Values for *Pi* are calculated for grain using the equations in Table B-3-9. Uncertainties introduced by this variable include the following: (1) Some of the variables in the equation in Table B-3-9—including *Cs, Cyv, Q, Dydp*, and *Dywp*—are COPC- and site-specific. Uncertainties associated with these variables are site-specific. (2) In the equation in Table B-3-9, COPC-specific plant-soil biotransfer factors (*Br*) may not reflect site-specific conditions. This may be especially true for inorganic COPCs for which *Br's* would be more accurately estimated by using plant uptake response slope factors.
Qs	Quantity of soil ingested by the animal	kg/day	**0.022** This variable is site-specific. We recommend using the soil ingestion rate of 0.022 kg/day. This is consistent with Stephens et al. (1995). Uncertainties introduced by this variable include the following: (1) The recommended soil ingestion rate may not accurately represent site-specific or local conditions. (2) Empirical data to support soil ingestion rates of chickens are limited.
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-3-1. Uncertainties are site-specific.
Bs	Soil bioavailability factor	unitless	**1.0** The soil bioavailability factor, *Bs,* can be thought of as the ratio between bioconcentration (or biotransfer) factors for soil and vegetation for a given COPC. The efficiency of transfer from soil may differ from efficiency or transfer from plant material for some COPCs. If the transfer efficiency is lower for soils, then this ratio would be less than 1.0. If it is equal or greater than that of vegetation, the *Bs* would be equal to or greater than 1.0. Due to limited data regarding bioavailability from soil, we recommend a default value of 1.0 for *Bs,* until more COPC-specific data is available for this parameter. Some COPCs may be much less bioavailable from soil than from plant tissues. This uncertainty may overestimate *Bs*.

TABLE B-3-14

CONCENTRATION IN CHICKEN
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
$Ba_{chicken}$	Biotransfer factor for chicken	day/kg FW tissue	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. $Ba_{chicken}$ is defined as the ratio of the COPC concentration in fresh weight tissue (mg COPC/kg FW tissue) to the daily intake of the COPC (mg COPC/day) from chicken feed. Uncertainties associated with this variable include the following: (1) We recommend calculating $Ba_{chicken}$ values for organic COPCs other than dioxins and furans by from Ba_{beef} values by assuming that chicken is 15 percent fat and beef is 19 percent fat. Values calculated this way may not accurately represent the behavior of organic COPCs under site-specific conditions. Therefore, estimates of $Ba_{chicken}$ and, therefore $A_{chicken}$, may be under- or overestimated to some degree. (2) The beef-to-chicken fat content ratio method which is used to estimate $Ba_{chicken}$ values from Ba_{beef} values for organics (except PCDDs and PCDFs) assumes that (1) COPCs bioconcentrate in the fat tissues, and (2) there are no differences in metabolism or feeding characteristics between beef cattle and chicken. Due to uncertainties associated with these assumptions, $Ba_{chicken}$, and $A_{chicken}$ values may be under- or overestimated to some degree. (3) The recommended *BCFs* may not accurately represent the behavior of COPCs under site-specific or local conditions. For example, Stephens et al. (1995) noted that chickens raised under field conditions, and which probably had more than 10 percent soil in their diet, showed larger apparent *BCFs*. Therefore, using the recommended *BCFs* may underestimate $A_{chicken}$, to some extent. (4) The recommended *BCFs* are based on incomplete experimental results. Stephens et al. (1995) presented results that are based on the full 178-day exposure period. A comparison of the results from Stephens et al. (1992) to those from Stephens et al. (1995) shows that *BCFs* from the high-dose group are generally higher than *BCFs* from the low-dose group. Therefore, using the currently recommended *BCFs* may underestimate the COPC concentration in chicken, $A_{chicken}$.

TABLE B-3-14

CONCENTRATION IN CHICKEN
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Chang, R.R., D. Hayward, L. Goldman, M. Harnly, J. Flattery, and R.D. Stephens. 1989. "Foraging Farm Animals as Biomonitors for Dioxin Contamination." *Chemosphere*. Volume 19; 481-486.

This document appears to be cited by Stephens et al. (1992) as support for the assumption that soil represents 10 percent of the diet of free-range chickens.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is the reference source for the equation in Table B-3-14. This document also cites Stephens et al. (1992) as the source for the recommended fraction of diet that is soil (Fd) and $BCF_{Chicken}$ for dioxins and furan congeners.

Petreas, M.X., L. R. Goldman, D. G. Hayward, R. Chang, J. Flattery, T. Wiesmuller, R.D. Stephens, D.M. Fry, and C. Rappe. 1991. "Biotransfer and Bioaccumulation of PCDD/PCDFs from Soils: Controlled Exposure Studies of Chickens." *Chemosphere*. Volume 23: 1731-1741.

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Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1992. "Biotransfer and Bioaccumulation of Dioxins and Dibenzofurans from Soil." Hazardous Materials Laboratory, California Department of Health Services. Berkeley, California. Presented at the 12th International Symposium on Dioxins and Related Compounds. August 24 through 28. University of Tampere, Tampere, Finland.

This document is cited as the source of the assumption that free-range chickens ingest soil as 10 percent of their diet and as the source of the dioxin and furan congeners-specific *BCFs* recommended by NC DEHNR (1997). However this document does not clearly reference or document the assumption that soil represents 10 percent of a free-range chicken's diet. The document appears to cite two other documents as supporting its assumption, (1) Change, Hayward, Goldman, Harnly, Flattery and Stephens (1989) and (2) Petreas, Goldman, Hayward, Chang, Flattery, Wiesmuller, Stephens, Fry, and Rappe (1992).

This document also presents dioxin and furan congener-specific *BCFs* (thigh) for the low- exposure group after 80 days of a 178-day total exposure period. The chickens in the low-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 42 ppt I-TEQ. Chickens in the high-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 458 ppt I-TEQ; *BCF* results were not presented from the high-dose group.

Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1995. "Biotransfer and Bioaccumulation of Dioxins and Furans from Soil: Chickens as a Model for Foraging Animals." *The Science of the Total Environment.* Volume 175: 253-273.

This document is an expansion of the results originally presented in Stephens et al. (1992). In particular, this document suggests that the percentage of soil in the diet of chickens raised under field conditions is likely to be greater than 10 percent, the value that was used in the experimental study presented in this document.

This document also presents dioxin and furan congener-specific *BCFs* (thigh) under two exposure schemes—low exposure and high exposure. The white leghorn (Babcock D 300) chickens in the low group were fed a diet containing 10 percent soil with a PCDD/PCDF concentrations of 42 ppt I-TEQ. Chickens in the high group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 460 ppt I-TEQ (some congeners were fortified by spiking).

The *BCFs* presented for low- and high-dose groups both represent averages of results from Day-80 and Day-164 of a total 178-day exposure period.

TABLE B-3-14

CONCENTRATION IN CHICKEN
(CONSUMPTION OF ANIMAL PRODUCTS EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1990. *Interim Final Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Environmental Criteria and Assessment Office. Office of Research and Development. EPA/600/6-90/003. January.

This document is a reference source for the equation in Table B-3-9; and an F_i value of 1.0.

U.S. EPA. 1992. *Technical Support Document for Land Application of Sewage Sludge.* Volumes I and II. EPA 822/R-93-001a. Office of Water. Washington, D.C.

U.S. EPA (1995) recommends that uptake slope factors for the metals cadmium, selenium, and zinc cited by this document be used to derive $Ba_{chicken}$ values.

U.S. EPA. 1995. *Further Issues for Modeling the Indirect Exposure Impacts from Combustor Emissions.* Office of Research and Development. Washington, D.C. January 20.

U.S. EPA. 1997a. *Exposure Factors Handbook.* “Food Ingestion Factors”. Volume II. EPA/600/P-95/002F. August.

U.S. EPA. 1997b. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 8)

Description

Use the equations in this table to calculate an average COPC soil concentration resulting from wet and dry deposition of particles and vapors to soil over the exposure duration. We recommend assuming that COPCs are incorporated only to a finite depth (the soil mixing zone depth, Z_s). Use the COPC soil concentration averaged over the exposure duration, represented by Cs, for carcinogenic COPCs, where risk is averaged over the lifetime of an individual. Because the hazard quotient associated with noncarcinogenic COPCs is based on a reference dose rather than a lifetime exposure, we recommend using the highest annual average COPC soil concentration occurring during the exposure duration period for noncarcinogenic COPCs. The highest annual average COPC soil concentration would most likely occur at the end of the time period of combustion and is represented by Cs_{tD}.

The following uncertainties are associated with this variable:

(1) We assume that the time period for deposition of COPCs resulting from hazardous waste combustion is a conservative, long-term value. This assumption may overestimate Cs and Cs_{tD}.
(2) Exposure duration values (T_2) are based on historical mobility studies and won't necessarily remain constant. Specifically, mobility studies indicate that most receptors that move remain in the vicinity of the combustion unit; however, it is impossible to accurately predict the probability that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants.
(3) Using a value of zero for T_1 doesn't account for exposure that may have occurred from historic operations and emissions from hazardous waste combustion. This may underestimate Cs and Cs_{tD}.
(4) For soluble COPCs, leaching might lead to movement to below the mixing depth, resulting in lower concentrations within the mixing depth. This uncertainty may overestimate Cs and Cs_{tD}.
(5) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This may underestimate Cs and Cs_{tD}.

Equation for Carcinogens

Soil Concentration Averaged Over Exposure Duration

$$Cs = \frac{\left(\frac{Ds \cdot tD - Cs_{tD}}{ks}\right) + \left(\frac{Cs_{tD}}{ks} \cdot [1 - \exp(-ks\,(T_2 - tD))]\right)}{(T_2 - T_1)} \quad for\ T_1 < tD < T_2$$

$$Cs = \frac{Ds}{ks \cdot (tD - T_1)} \cdot \left(\left[tD + \frac{\exp(-ks \cdot tD)}{ks}\right] - \left[T_1 + \frac{\exp(-ks \cdot T_1)}{ks}\right]\right) \quad for\ T_2 \leq tD$$

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

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Equation for Noncarcinogens

Highest Annual Average Soil Concentration

$$Cs_{tD} = \frac{Ds \cdot [1 - \exp(-ks \cdot tD)]}{ks}$$

where

$$Ds = \frac{100 \cdot Q}{Z_s \cdot BD} \cdot [F_v \ (Dytwv) + Dytwp \ (1 - F_v)]$$

For mercury modeling

$$Ds_{(Mercury)} = \frac{100 \cdot [0.48 Q_{(Total)}]}{Z_s \cdot BD} \cdot [F_{v_{(Hg^{2+})}} \ (Dytwv) + Dytwp \cdot [1 - F_{v_{(Hg^{2+})}}]$$

Use *0.48Q* for total mercury and F_v = 0.85 in the mercury modeling equation to calculate *Ds*. Apportion the calculated *Ds* value into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the assumed 98% Hg^{2+} and 2% MHg speciation split in soils (see Chapter 2). Elemental mercury (Hg^0) occurs in very small amounts in the vapor phase and does not exist in the particle or particle-bound phase. Therefore, assume elemental mercury deposition onto soils is negligible or zero. Evaluate elemental mercury for the direct inhalation pathway only (Table B-5-1).

$Ds_{(Hg2+)} = 0.98 \ Ds_{(Mercury)}$
$Ds_{(Mhg)} = 0.02 \ Ds_{(Mercury)}$
$Ds_{(Hg0)} = 0.0$

Evaluate divalent and methyl mercury as individual COPCs. Calculate *Cs* for divalent and methyl mercury using the corresponding (1) fate and transport parameters for mercuric chloride (divalent mercury, Hg^{2+}) and methyl mercury provided in Appendix A-2, and (2) *Ds* (Hg^{2+}) and *Ds* (MHg) as calculated above.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 8)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	
Cs_{tD}	Soil concentration at time *tD*	mg COPC/kg soil	
Ds	Deposition term	mg COPC/kg soil-yr	**Varies** U.S. EPA (1994a) and NC DEHNR (1991) recommended incorporating a deposition term into the *Cs* equation. Uncertainties associated with this variable include the following: (1) Five of the variables in the equation for *Ds* (*Q, Cyv, Dywv*, *Dywp*, and *Dydp*) are COPC- and site-specific. Values of these variables are estimated through modeling. The direction and magnitude of any uncertainties shouldn't be generalized. (2) Based on the narrow recommended ranges, we expect uncertainties associated with *Vdv*, F_v, and *BD* to be low. (3) Values for Z_s vary by about one order of magnitude. Uncertainty is greatly reduced if you know whether soils are tilled or untilled.
tD	Time period over which deposition occurs (time period of combustion)	yr	**30** U.S. EPA (1998) suggests that this period of time can be ≥ 30 years. We recommend using 30 years unless site-specific information is available indicating that this assumption is unreasonable (see Chapter 6 of the HHRAP).
ks	COPC soil loss constant due to all processes	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-2. The COPC soil loss constant is the sum of all COPC removal processes. Uncertainty associated with this variable includes the following: COPC-specific values for *ksg* (one of the variables in the equation in Table B-4-2) are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 8)

Variable	Description	Units	Value
T_2	Length of exposure duration	yr	**6, 30, or 40** We recommend reasonable maximum exposure (RME) values for T_2: Exposure Duration — RME — Reference Child Resident, Farmer Child, Fisher Child — 6 years — U.S. EPA (1997b) Adult Resident and Fisher — 30 years — U.S. EPA (1997b) Farmer — 40 years — U.S. EPA (1994b) U.S. EPA (1994c) recommended the following unreferenced values: Exposure Duration — Years Subsistence Farmer — 40 Adult Resident — 30 Subsistence Fisher — 30 Child Resident — 9 Uncertainties associated with this variable include the following: (1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate Cs and Cs_{tD}. (2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources. However, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate Cs and Cs_{tD}.
T_1	Time period at the beginning of combustion	yr	**0** Consistent with U.S. EPA (1994c), we recommend a value of 0 for T_1. The following uncertainty is associated with this variable: A T_1 of zero doesn't account for exposure that may have occurred from historical operation or emissions from the combustion of hazardous waste. This may underestimate Cs and Cs_{tD}.
100	Units conversion factor	mg-cm^2/kg-cm^2	

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 8)

Variable	Description	Units	Value
Q	COPC emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 of the HHRAP for guidance calculating this variable. Uncertainties associated with this variable are site-specific.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil / Depth (cm) / Reference Untilled / 2 / Brzuzy et al. (1995) Tilled / 20 / U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below Z_s, resulting in lower concentrations within the Z_s. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default BD value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended BD value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 6 of 8)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) F_v calculations assume a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Dytwv	Unitized yearly (water body or watershed) average total deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dytwp	Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 7 of 8)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Brzuzy, L.P. and R.A. Hites. 1995. "Estimating the Atmospheric Deposition of Polychlorinated Dibenzo-p-dioxins and Dibenzofurans from Soils." *Environmental Science and Technology.* Volume 29. Pages 2090-2098.

This reference presents soil profiles for dioxin measurements.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This reference is cited by U.S. EPA (1994b) as the source for a mean soil bulk density value, *BD*, of 1.5 g soil/cm^3 soil for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Press, Inc. New York, New York.

Cited by U.S. EPA (1998) for the statement that *BD* is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This is one of the source documents for the equation in Table B-4-1. This document also recommends using (1) a deposition term, *Ds*, and (2) COPC-specific F_v values.

Research Triangle Institute (RTI). 1992. *Preliminary Soil Action Level for Superfund Sites.* Draft Interim Report. Prepared for U.S. EPA Hazardous Site Control Division, Remedial Operations Guidance Branch. Arlington, Virginia. EPA Contract 68-W1-0021. Work Assignment No. B-03, Work Assignment Manager Loren Henning. December.

This document is a reference source for COPC-specific F_v values.

U.S. EPA. 1992. *Estimating Exposure to Dioxin-Like Compounds.* Draft Report. Office of Research and Development. Washington, D.C. EPA/600/6-88/005b.

The External Review Draft of the MPE document (the final is U.S. EPA 1998) cites this document as the source of values for soil mixing zone depth, Z_s, for tilled and untilled soils.

U.S. EPA. 1993. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste. Office of Research and Development. Washington, D.C. September.

This document is a reference for the equation in Table B-4-1. It recommends using a deposition term, *Ds*, and COPC-specific F_v values in the *Cs* equation.

TABLE B-4-1

WATERSHED SOIL CONCENTRATION DUE TO DEPOSITION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 8 of 8)

U.S. EPA 1994a. *Draft Guidance for Performing Screening Level Risk Analysis 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. April.

This document is a reference for the equation in Table B-4-1; it recommends using the following in the *Cs* equation: (1) a deposition term, *Ds*, and (2) a default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-Specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. June. EPA/600/6-88/005Cc.

This document recommends T_2 values for the farmer.

U.S. EPA. 1994c. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes.* Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- Values for the length of exposure duration, T_2
- Value of 0 for the time period of the beginning of combustion, T_1
- F_v values that range from 0.27 to 1 for organic COPCs
- Default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean for loam soil from Carsel et al. (1988)

U.S. EPA. 1997a. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1997b. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002Fc. August.

This document is a reference source for values for length of exposure duration, T_2.

U.S. EPA. 1998. *Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (MPE).* Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is a reference for recommended values for soil mixing zone depth, Z_s, for tilled and untilled soils.

TABLE B-4-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC soil loss constant, which accounts for the loss of COPCs from soil by several mechanisms.

Uncertainties associated with this equation include the following:

(1) COPC-specific values for *ksg* are empirically determined from field studies. No information is available regarding the application of these values to the site-specific conditions associated with affected facilities.
(2) The source of the equations in Tables B-4-3 through B-4-5 have not been identified.

Equation

$$ks = ksg + kse + ksr + ksl + ksv$$

Variable	Description	Units	Value
ks	COPC soil loss constant due to all processes	yr^{-1}	
ksg	COPC loss constant due to biotic and abiotic degradation	yr^{-1}	**Varies** This variable is COPC-specific. Values are available in the COPC tables in Appendix A-2. "Degradation rate" values are also presented in NC DEHNR (1997), however, no reference or source is provided for the values. U.S. EPA (1994a) and U.S. EPA (1994b) state that *ksg* values are COPC-specific; however, all *ksg* values are presented as zero (U.S. EPA 1994a) or as "NA" (U.S. EPA 1994b); the basis of these assumptions is not addressed. The following uncertainty is associated with this variable: COPC-specific values for *ksg* are empirically determined from field studies; no information is available regarding the application of these values to the site-specific conditions associated with affected facilities.

TABLE B-4-2

COPC SOIL LOSS CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** This variable is COPC- and site-specific, and is further discussed in Table B-4-3. Consistent with U.S. EPA (1994a), U.S. EPA (1994b) and NC DEHNR (1997), we recommend a default value of 0 for *kse* because contaminated soil erodes both onto the site and away from the site. Uncertainties associated with this variable include the following: (1) The source of the equation in Table B-4-3 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *kse*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *kse*.
ksr	COPC loss constant due to surface runoff	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-4. No reference document is cited for this equation; using this equation is consistent with U.S. EPA (1994b) and NC DEHNR (1997). U.S. EPA (1994a) assumes that all *ksr* values are zero but does not explain the basis of this assumption. Uncertainties associated with this variable (calculated using Table B-4-4) include the following: (1) The source of Table B-4-4 has not been identified. (2) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *ksr*. (3) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksr*.
ksl	COPC loss constant due to leaching	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-5. Using this equation is consistent with U.S. EPA (1998), U.S. EPA(1994b), and NC DEHNR (1997). U.S. EPA (1994a) assumes that all *ksl* values are zero but does not explain the basis of this assumption. Uncertainties associated with this variable (calculated using Table B-4-5) include the following: (1) The source of the equation in Table B-4-5 has not been identified. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in-situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

TABLE B-4-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
ksv	COPC loss constant due to volatilization	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-6. This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from U.S. EPA (1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986). Uncertainties associated with this equation include the following: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in lower concentrations within the Z_s. This uncertainty may overestimate *ksv*. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

REFERENCES AND DISCUSSION

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the reference documents for Tables B-4-4 and B-4-5. This document is also cited as (1) the source for a range of COPC-specific degradation rates (*ksg*), and (2) one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because of contaminated soil eroding both onto the site and away from the site.

U.S. EPA. 1994a. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

This document is cited as a source for the assumptions that losses resulting from erosion (*kse*), surface runoff (*ksr*), degradation (*ksg*), leaching (*ksl*), and volatilization (*ksv*) are all zero.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is one of the reference documents for Tables B-4-4 and B-4-5. This document is also cited as one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero and the loss resulting from degradation (*ksg*) is "NA" or zero for all compounds.

TABLE B-4-2

COPC SOIL LOSS CONSTANT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1998. “*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*.” Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference documents for the equations for *ksr*, *ksl*, and *ksv*.

TABLE B-4-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the constant for COPC loss resulting from erosion of soil. Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend a default value of zero for *kse* is zero because of contaminated soil eroding both onto the site and away from the site. In site-specific cases where the permitting authority considers it appropriate to calculate a *kse*, we recommend using the equation presented in this table along with associated uncertainties. You can find additional discussion on determining *kse* in U.S. EPA (1998). Uncertainties associated with this equation include:

(1) For soluble COPCs, leaching might lead to movement below 1 centimeter in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *kse*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *kse*.

Equation

$$kse = \frac{0.1 \cdot X_e \cdot SD \cdot ER}{BD \cdot Z_s} \cdot \left(\frac{Kd_s \cdot BD}{\theta_{sw} + (Kd_s \cdot BD)} \right)$$

Variable	Description	Units	Value
kse	COPC loss constant due to soil erosion	yr^{-1}	**0** Consistent with U.S. EPA (1994), U.S. EPA (1994b), and NC DEHNR (1997), we recommend assuming a default value of zero for *kse* because contaminated soil erodes both onto the site and away from the site. Uncertainty may overestimate *kse*.
0.1	Units conversion factor	g-kg/cm^2-m^2	
X_e	Unit soil loss	kg/m^2-yr	**Varies** This variable is site-specific and is calculated by using the equation in Table B-4-13. The following uncertainty is associated with this variable: All of the equation variables are site-specific. Using default values rather than site-specific values for any or all of these variables will result in unit soil loss (X_e) estimates that are under- or overestimated to some degree. Based on default values, X_e estimates can vary over a range of less than two orders of magnitude.

TABLE B-4-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
SD	Sediment delivery ratio	unitless	**Varies** This value is site-specific and is calculated using the equation in Table B-4-14. Uncertainties associated with this variable include the following: (1) The recommended default values for the empirical intercept coefficient, *a*, are average values that are based on studies of sediment yields from various watersheds. Therefore, those default values may not accurately represent site-specific watershed conditions. As a result, using these default values may under- or overestimate *SD*. (2) The recommended default value for the empirical slope coefficient, *b*, is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, using the default value may under- or overestimate *SD*.
ER	Soil enrichment ratio	unitless	**Inorganics: 1** **Organics: 3** COPC enrichment occurs because (1) lighter soil particles erode more quickly than heavier soil particles, and (2) concentration of organic COPCs—which is a function of organic carbon content of sorbing media—is expected to be higher in eroded material than in *in situ* soil (U.S. EPA 1998). In the absence of site-specific data, we recommend a default value of 3 for organic COPCs and 1 for inorganic COPCs. This is consistent with other Agency guidance (U.S. EPA 1998), which recommends a range of 1 to 5 and a value of 3 as a "reasonable first estimate." This range has been used for organic matter, phosphorus, and other soil-bound COPCs (U.S. EPA 1998); however, no sources or references were provided for this range. *ER* is generally higher in sandy soils than in silty or loamy soils (U.S. EPA 1998). The following uncertainty is associated with this variable: The default *ER* value may not accurately reflect site-specific conditions; therefore, *kse* may be over- or underestimated to an unknown extent. Using county-specific *ER* values will reduce the extent of any uncertainties.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

TABLE B-4-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

Variable	Description	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to of other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	mL water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; θ_{sw} can be estimated as the midpoint between a soil's field capacity and wilting point, if a representative watershed soil can be identified. However, we recommend the use of 0.2 mL/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1993) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *kse* may be under- or overestimated to a small extent, based on the limited range of values.

TABLE B-4-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 5)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source for a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the sources that recommend assuming that the loss resulting from erosion (*kse*) is zero because contaminated soil erodes both onto the site and away from the site.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities.* April.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document is the source of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document recommends (1) a default *BD* value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988), and (2) a default θ_{sw}, value of 0.2 (mL water/cm^3 soil).

TABLE B-4-3

COPC LOSS CONSTANT DUE TO SOIL EROSION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the source of a range of COPC enrichment ratio, *ER*, values. The recommended range, 1 to 5, was used for organic matter, phosphorous, and other soul-bound COPCs. This document recommends a value of 3 as a "reasonable first estimate," and states that COPC enrichment occurs because lighter soil particles erode more quickly than heavier soil particles. Lighter soil particles have higher ratios of surface area to volume and are higher in organic matter content. Therefore, concentration of organic COPCs, which is a function of the organic carbon content of sorbing media, is expected to be higher in eroded material than in *in situ* soil.

This document is also a source of the following:

- A range of θ_{sw} values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for Z_s, for tilled and untilled soil
- The equations in Tables B-1-3 and B-1-5.

TABLE B-4-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the COPC loss constant due to runoff of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksr*.
(2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate *ksr*.

Equation

$$ksr = \frac{RO}{\theta_{sw} \cdot Z_s} \cdot \left(\frac{1}{1 + \left(Kd_s \cdot BD / \theta_{sw} \right)} \right)$$

Variable	Description	Units	Value
ksr	COPC loss constant due to runoff	yr^{-1}	
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994b), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures for estimating the amount of surface runoff, such as those based on the U.S. Soil Conservation Service curve number equation (CNE). U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksr* may be under- or overestimated to an unknown degree.

TABLE B-4-4

COPC LOSS CONSTANT DUE TO RUNOFF
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils), recommended by U.S. EPA (1998) (no source or reference is provided for this range), and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksr* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting lower concentrations within the Z_s. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	mL water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

TABLE B-4-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. “Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils.” *Journal of Contaminant Hydrology*. Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil) for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994), and NC DEHNR (1997) as a reference to calculate average annual runoff, *RO*. This reference provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these values are total contributions and not only surface runoff, U.S. EPA (1994) recommends that the volumes be reduced by 50 percent in order to estimate surface runoff.

Hillel, D. 1980. Fundamentals of Soil Physics. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that dry soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the source documents that cites the use of Table B-4-4; however, this document is not the original source of this equation (this source is unknown). This document also recommends the following:

- Estimating annual current runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as using the U.S. Soil Conservation Service curve number equation (CNE); U.S. EPA (1985) is cited as an example of such a procedure.
- Default value of 0.2 (mL water/cm^3 soil) for soil volumetric water content (θ_{sw})

TABLE B-4-4

COPC LOSS CONSTANT DUE TO RUNOFF (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water—Part I (Revised. 1985)*. Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate site-specific surface runoff.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents a range of values for soil mixing zone depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes*. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities. Offices of Emergency and Remedial Response. Office of Solid Waste. December 14.

This document recommends the following:

- Estimation of average annual runoff, *RO*, by using the *Water Atlas of the United States* (Geraghty et al. 1973)
- Default soil dry bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on the mean for loam soil that is taken from Carsel et al. (1988)
- Default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of soil volumetric water content, θ_{sw}, values of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) (the original source of, or reference for, these values is not identified)
- A range of values for soil mixing depth, Z_s, for tilled and untilled soil (the original source of, or reference for, these values is not identified)
- Using the *Water Atlas of the United States* (Geraghty, Miller, Van der Leeden, and Troise 1973) to calculate average annual runoff, *RO*

TABLE B-4-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC loss constant resulting from leaching of soil. Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksl*.

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksl*.

(3) The original source of this equation has not been identified. U.S. EPA (1998) presents the equation as shown here. U.S. EPA (1994a) and NC DEHNR (1997) replaced the numerator as shown with "*q*", defined as average annual recharge (cm/yr).

Equation

$$ksl = \frac{P + I - OR - E_v}{\theta_{sw} \cdot Z_s \cdot \left[1.0 + \left(BD \cdot Kd_s / \theta_{sw}\right)\right]}$$

Variable	Description	Units	Value
ksl	COPC loss constant due to leaching	yr^{-1}	
P	Average annual precipitation	cm/yr	**18.06 to 164.19** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (U.S. Bureau of Census 1987; Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. We recommend using site-specific data. The following uncertainty is associated with this variable: To the extent that a site is not located near an established meteorological data station, and site-specific data are not available, default average annual precipitation data may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated. However, average annual precipitation data are reasonably available; therefore, we expect uncertainty introduced by this variable to be minimal.

TABLE B-4-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
I	Average annual irrigation	cm/yr	**0 to 100** This variable is site-specific. This range is based on information presented in U.S. EPA (1998), representing data for 69 selected cities (Baes et al. 1984). The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual irrigation information is not available, default values (generally based on the closest comparable location) may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
RO	Average annual surface runoff from pervious areas	cm/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994a), and NC DEHNR (1997), you can estimate *RO* by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), you can also use more detailed, site-specific procedures, such as those based on the U.S. Soil Conservation Service CNE. U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.
E_v	Average annual evapotranspiration	cm/yr	**35 to 100** This variable is site-specific. This range is based on information presented in U. S. EPA (1998), representing data from 69 selected cities. The 69 selected cities are not identified; however, they appear to be located throughout the continental United States. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual evapotranspiration information is not available, default values may not accurately reflect site-specific conditions. As a result, *ksl* may be under- or overestimated to an unknown degree.

TABLE B-4-5

COPC LOSS CONSTANT DUE TO LEACHING
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. We recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range of 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b) and NC DEHNR (1997). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, *ksl* may be under- or overestimated to a small extent, based on the limited range of values.
Z_s	Soil depth mixing zone	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in lower concentrations within the Z_s. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.

TABLE B-4-5

COPC LOSS CONSTANT DUE TO LEACHING
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen and R.W. Shor. 1984. "A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture." Prepared for the U.S. Department of Energy under Contract No. DEAC05-840R21400.

For the continental United States, as cited in U.S. EPA (1998), this document is the source of a series of maps showing: (1) average annual precipitation (*P*), (2) average annual irrigation (*I*), and (3) average annual evapotranspiration isolines.

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994a) as the source for a mean soil bulk density value, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Geraghty, J.J., D.W. Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center, Port Washington, New York.

This document is cited by U.S. EPA (1998), U.S. EPA (1994a), and NC DEHNR (1997) as a reference for calculating *RO*. This document provides maps with isolines of annual average surface runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these volumes are total contributions and not only surface runoff, U.S. EPA (1994a) recommends that the volumes be reduced by 50 percent in order to estimate average annual surface runoff.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

This document is cited by U.S. EPA (1998) for the statement that *BD* is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

TABLE B-4-5

COPC LOSS CONSTANT DUE TO LEACHING (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 5)

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the source documents that cites the use of the equation in Table B-4-5. However, the document is not the original source of this equation. This document also recommends the following:

- Estimation of average annual surface runoff, *RO* (cm/yr), by using either the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as the U.S. Soil Conservation Service CNE; U.S. EPA 1985 is cited as an example of such a procedure.
- A default value of 0.2 (mL water/cm^3 soil) for soil volumetric water content, θ_{sw}

U.S. Bureau of the Census. 1987. *Statistical Abstract of the United States:* 1987. 107th edition. Washington, D.C.

This document is a source of average annual precipitation (*P*) information for 69 selected cities, as cited in U.S. EPA (1998); these 69 cities are not identified.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Groundwater*. Part I (Revised 1985). Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

This document is cited by NC DEHNR (1997) as an example of the use of the U.S. Soil Conservation Service CNE to estimate *RO*.

U.S. EPA. 1990. *Interim Final Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Environmental Criteria and Assessment Office. Office of Research and Development. EPA 600-90-003. January.

This document presents ranges of (1) average annual precipitation, (2) average annual irrigation, and (3) average annual evapotranspiration. This document cites Baes et al. (1984) and U.S. Bureau of the Census (1987) as the original sources of this information.

U.S. EPA. 1994a. *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.

U.S. EPA. 1994b. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures.* External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents values for soil mixing depth, Z_s, for tilled and untilled soil, as cited in U.S. EPA (1993).

This document recommends (1) a default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil), based on U.S. EPA (1993), and (2) a default soil bulk density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions.*" Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is one of the reference source documents for the equation in Table B-1-5. The original source of this equation is not identified. This document also presents a range of values for soil mixing depth, *Z*, for tilled and untilled soil; the original source of these values is not identified.

TABLE B-4-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the COPC loss constant from soil due to volatilization, and was obtained from *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (U.S. EPA 1998). The soil loss constant due to volatilization (*ksv*) is based on gas equilibrium coefficients and gas phase mass transfer. The first order decay constant, *ksv*, is obtained by adapting the Hwang and Falco equation for soil vapor phase diffusion (Hwang and Falco 1986).

Uncertainties associated with this equation include the following:

(1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting in a greater mixing depth. This uncertainty may overestimate *ksv*.

(2) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This uncertainty may underestimate *ksv*.

Equation

$$ksv = \left[\frac{3.1536 \times 10^{7} \cdot H}{Z_s \cdot Kd_s \cdot R \cdot T_a \cdot BD}\right] \cdot \left(\frac{D_a}{Z_s}\right) \cdot \left[1 - \left(\frac{BD}{\rho_{soil}}\right) - \theta_{sw}\right]$$

Variable	Definition	Units	Value
ksv	Constant for COPC loss due to volatilization	yr^{1}	
$3.1536 \times 10^{+07}$	Units conversion factor	s/yr	
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Values for this variable, estimated by using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, *ksv* may be under- or overestimated.

TABLE B-4-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 5)

Variable	Definition	Units	Value
Z_s	Soil mixing zone depth	cm	**2 to 20** We recommend the following values for Z_s: Soil — Depth (cm) — Reference Untilled — 2 — Brzuzy et al. (1995) Tilled — 20 — U.S. EPA (1998) U.S. EPA (1992) recommended values of 1 cm (for untilled) and 20cm (for tilled soil). These values are consistent with U.S. EPA (1998), which further states that leaching soluble compounds might lead to movement below a 1-cm depth. A default value of 2 cm for untilled soil mixing depth is based on a study that profiled dioxin measurements within soil (Brzuzy et al. 1995). A default value of 20 cm for tilled soil mixing depth is based on U.S. EPA (1998). The following uncertainties are associated with this variable: (1) For soluble COPCs, leaching might lead to movement to below 2 centimeters in untilled soils, resulting a greater mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (2) Deposition to hard surfaces may result in dust residues that have negligible dilution compared to other residues. This uncertainty may underestimate Cs and Cs_{tD}.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
R	Universal gas constant	atm-m^3/mol-K	**8.205×10^{-5}** There are no uncertainties associated with this parameter.

TABLE B-4-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 5)

Variable	Definition	Units	Value
T_a	Ambient air temperature	K	**298** This variable is site-specific. U.S. EPA (1998) also recommends an ambient air temperature of 298 K. The following uncertainty is associated with this variable: To the extent that site-specific or local values for the variable are not available, default values may not accurately represent site-specific conditions. We expect the uncertainty associated with the selection of a single value from within the temperature range at a single location to be more significant than the uncertainty associated with choosing a single ambient temperature to represent all localities. In other words, the range of average ambient temperatures across the country is generally less than the temperature range at an individual site.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended soil bulk density value may not accurately represent site-specific soil conditions.
ρ_{soil}	Solids particle density	g/cm^3	**2.7** We recommend the use of this value, based on Blake and Hartage (1996) and Hillel (1980). The solids particle density will vary with location and soil type.
D_a	Diffusivity of COPC in air	cm^2/s	**Varies** This value is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The default D_a values may not accurately represent the behavior of COPCs under site-specific conditions. However, we expect the degree of uncertainty to be minimal.

TABLE B-4-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 5)

Variable	Definition	Units	Value
θ_{sw}	Soil volumetric water content	mL/cm^3 soil	**0.2** This variable depends on the available water and on soil structure. You can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: The default θ_{sw} values may not accurately reflect site-specific or local conditions; therefore, *ksv* may be under- or overestimated to a small extent, based on the limited range of values.

REFERENCES AND DISCUSSION

Blake, GR. and K.H. Hartge. 1996. *Particle Density. Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods.* Second Edition. Arnold Klute, Ed. American Society of Agronomy, Inc. Madison, WI., p. 381.

Carsel, R.F., R.S, Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Vol. 2. Pages 11-24.

This document is cited by U.S. EPA (1994b) as the source of a mean soil bulk density value, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics.* Academic Press, Inc. New York, New York.

Hwang S. T. and Falco, J. W. 1986. "Estimation of multimedia exposures related to hazardous waste facilities", In: *Pollutants in a Multimedia Environment.* Yoram Cohen, Ed. Plenum Publishing Corp. New York.

Miller, R.W. and D.T. Gardiner. 1998. In: *Soils in Our Environment.* J.U. Miller, Ed. Prentice Hall. Upper Saddle River, NJ. pp. 80-123.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures*. External Review Draft. Office of Research and Development. Washington, D.C. EPA/600/6-88/005Cc. June.

This document presents value for soil, mixing depth, Z_s, for tilled and untilled soil.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response. Office of Solid Waste. December 14.

TABLE B-4-6

COPC LOSS CONSTANT DUE TO VOLATILIZATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 5)

This document recommends a default soil density, *BD*, value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil that is taken from Carsel et al. (1988).

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends the following:

- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil; however, the source or basis for these values is not identified
- A default ambient air temperature of 298 K
- A range of soil volumetric water content, θ_{sw}

TABLE B-4-7

TOTAL WATER BODY LOAD (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 2)

Description

This equation calculates the total average water body load from wet and dry vapor and particle deposition, runoff, and erosion loads. The limitations and uncertainties associated with this equation include the following:

(1) Uncertainties associated with variables in equations presented in Tables B-4-8, B-4-9, B-4-10, B-4-11, and B-4-12 that are site-specific. These variables include *Q, Dytwv, Dytwp*, A_w, *Cywv*, A_I, A_L, *Cs*, and X_e. Values for many of these variables are estimated through the use of mathematical models and the uncertainties associated with values for these variables may be significant in some cases (Bidleman 1988).

(2) We expect the uncertainties associated with the remaining variables in equations presented in Tables B-4-8, B-4-9, B-4-10, B-4-11, and B-4-12 to be less significant, primarily because of the narrow ranges of probable values for these variables or because values for these variables (such as Kd_s) were estimated using well-established methods.

Equation

$$L_T = L_{DEP} + L_{dif} + L_{RI} + L_R + L_E$$

Variable	Description	Units	Value
L_T	Total COPC load to the water body	g/yr	
L_{DEP}	Total (wet and dry) particle phase and vapor phase COPC direct deposition load to water body	g/yr	**Varies** This variable is COPC- and site-specific, and calculated using the equation presented in Table B-4-8. Uncertainty associated with this variable include the following: Most of the uncertainty associated with the variables in the equation in Table B-4-8, specifically those associated with *Q, Dytwv, Dytwp*, and A_w, are site-specific and may be significant in some cases.
L_{dif}	Vapor phase COPC diffusion load to water body	g/yr	**Varies** This variable is calculated using equation presented in Table B-4-12. Uncertainty associated with this variable include the following: Most of the uncertainty associated with the variables in the equation in Table B-4-12, specifically those associated with *Q, Cywv*, and A_w, are site-specific.

TABLE B-4-7

TOTAL WATER BODY LOAD (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 2)

Variable	Description	Units	Value
L_{RI}	Runoff load from impervious surfaces	g/yr	**Varies** This variable is calculated using the equation presented in Table B-4-9. Uncertainty associated with this variable include the following: Most of the uncertainty associated with the variables in this equation, specifically those associated with *Q, Dytwv, Dytwp*, and A_I, are site-specific.
L_R	Runoff load from pervious surfaces	g/yr	**Varies** This variable is calculated using equation presented in Table B-4-10. Uncertainties associated with this variable include the following: (1) Most of the uncertainties associated with the variables in the equation in Table B-4-10, specifically those for A_L, A_I, and *Cs*, are site-specific. (2) Uncertainties associated with the remaining variable in the equation in Table B-4-10 are not expected to be significant, primarily because of the narrow ranges of probable values for these variables or the use of well-established estimation procedures (Kd_s).
L_E	Soil erosion load	g/yr	**Varies** This variable is calculated using equation presented in Table B-4-11. Uncertainties associated with this variable include the following: (1) Most of the uncertainties associated with the variables in the equation in Table B-4-11, specifically those for X_e,, A_L, A_I, and Cs, are site-specific. (2) Uncertainties associated with the remaining variables in the equation in Table B-4-11 are not expected to be significant, primarily because of the narrow range of probable values for these variables or the use of well-established estimation procedures (Kd_s).

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion in Table B-1-1.

TABLE B-4-8

DEPOSITION TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the average load to the water body from direct deposition of wet and dry particles and wet vapors onto the surface of the water body. Uncertainties associated with this equation include the following:

(1) Most of the uncertainties associated with the variables in this equation, specifically those associated with Q, $Dytwv$, $Dytwp$, and A_w, are site-specific.

(2) It is calculated assuming a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower calculated F_v value. However, F_v would likely to be only a few percent lower.

Equation

$$L_{DEP} = Q \cdot [\ F_v \cdot Dytwv + (1 - F_v) \cdot Dytwp] \cdot A_w$$

For mercury modeling

$$L_{DEP_{(Mercury)}} = (0.48Q_{(Total)}) \cdot [\ F_{v_{(Hg^{2+})}} \cdot Dytwv + (1 - F_{v_{(Hg^{2+})}}) \cdot Dytwp] \cdot A_w$$

Use $0.48Q$ for total mercury (to account for loss to the global cycle) and F_v = 0.85 in the mercury modeling equation.

Variable	Description	Units	Value
L_{DEP}	Total (wet and dry) particle phase and vapor phase direct deposition load to water body	g/yr	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.

TABLE B-4-8

DEPOSITION TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources. It would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Dytwv	Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dytwp	Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
A_w	Water body surface area	m^2	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.

TABLE B-4-8

DEPOSITION TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion in Table B-1-1.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is a reference source for the equation in Table B-4-8. This document also recommends by using the equations in Bidleman (1988) to calculate F_v values for all organics other than dioxins (PCDD/PCDFs). However, the document does not present a recommendation for dioxins. Finally, this document states that metals are generally entirely in the particulate phase ($F_v = 0$) except for mercury, which is assumed to be entirely in the vapor phase. The document does not state whether F_v for mercury should be calculated by using the equations in Bidleman (1988).

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is a reference source for Equation B-4-8. This document also presents values for organic COPCs that range from 0.27 to 1. F_v values for organics other than PCDD/PCDFs are calculated by using the equations presented in Bidleman (1988). The F_v value for PCDD/PCDFs is assumed to be 0.27, based on U.S. EPA (no date). Finally, this document presents F_v values for inorganic COPCs equal to 0, based on the assumption that these COPCs are nonvolatile and assumed to be 100 percent in the particulate phase and 0 percent in the vapor phase.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-9

IMPERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the average runoff load to the water body from impervious surfaces in the watershed from which runoff is conveyed directly to the water body.

Uncertainties associated with this equation include the following:

(1) Most of the uncertainties associated with the variables in this equation, specifically those associated with Q, $Dytwv$, $Dytwp$, and A_I, are site-specific.

(2) The equation assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.

Equation

$$L_{RI} = Q \cdot \left[F_v \cdot Dytwv + (1.0 - F_v) \cdot Dytwp\right] \cdot A_I$$

For Mercury modeling

$$L_{RI(Mercury)} = (0.48\, Q_{(Total)}) \cdot \left[F_{v(Hg^{2+})} \cdot Dytwv + (1.0 - F_{v(Hg^{2+})}) \cdot Dytwp\right] \cdot A_I$$

Use $0.48Q$ for total mercury (to account for loss to the global cycle) and $F_v = 0.85$ in the mercury modeling equation.

Variable	Description	Units	Value
L_{RI}	Runoff load from impervious surfaces	g/yr	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific (see Chapters 2 and 3). Uncertainties associated with this variable are site-specific.

TABLE B-4-9

IMPERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) Calculations assume a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable *c* (Junge constant) is constant for all chemicals; however, the value of *c* depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of *c* to vary, uncertainty is introduced if a constant value of *c* is used to calculate F_v.
Dytwv	Unitized yearly (water body or watershed) average total (wet and dry) deposition from vapor phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
Dytwp	Unitized yearly (water body or watershed) average total (wet and dry) deposition from particle phase	s/m^2-yr	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
A_I	Impervious watershed area receiving COPC deposition	m^2	**Varies** This variable is site-specific. Uncertainties associated with this variable are site-specific.

TABLE B-4-9

IMPERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. *"Atmospheric Processes."* *Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion see References and Discussion in Table B-1-1.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is a reference source for the equation in Table B-4-9. This document also recommends using the equations in Bidleman (1988) to calculate F_v values for all organics other than dioxins (PCDD/PCDFs). However, the document does not present a recommendation for dioxins. Finally, this document states that generally metals are entirely in the particulate phase ($F_v = 0$) except for mercury, which is assumed to be entirely in the vapor phase. The document does not state whether F_v for mercury should be calculated using the equations in Bidleman (1988).

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is a reference source for the equation in Table B-4-9. This document also presents F_v values for organic COPCs that range form 0.27 to 1. F_v values for organics other than PCDD/PCDFs are calculated using the equations presented in Bidleman (1988). The F_v value for PCDD/PCDFs is assumed to be 0.27, based on Lorber (no date). Finally, this document presents F_v values for inorganic COPCs equal to 0, based on the assumption that these COPCs are nonvolatile and 100 percent in the particle phase (and 0 percent in the vapor phase).

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-10

PERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the average runoff load to the water body from pervious soil surfaces in the watershed. Uncertainty associated with this equation includes the following:

> To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, L_R may be under- or overestimated to an unknown degree.

Equation

$$L_R = RO \cdot (A_L - A_I) \cdot \frac{Cs \cdot BD}{\theta_{sw} + Kd_s \cdot BD} \cdot 0.01$$

Variable	Description	Units	Value
L_R	Runoff load from pervious surfaces	g COPC/yr	
RO	Average annual surface runoff from pervious areas	cm water/yr	**Varies** This variable is site-specific. According to U.S. EPA (1998), U.S. EPA (1994), and NC DEHNR (1997), average annual surface runoff, *RO*, can be estimated by using the *Water Atlas of the United States* (Geraghty et al. 1973). According to NC DEHNR (1997), more detailed, site-specific procedures for estimating the amount of surface runoff, such as those based on the U.S. Soil Conservation Service CNE may also be used. U.S. EPA (1985) is cited as an example of such a procedure. The following uncertainty is associated with this variable: To the extent that site-specific or local average annual surface runoff information is not available, default or estimated values may not accurately represent site-specific or local conditions. As a result, *RO* may be under- or overestimated to an unknown degree.
A_L	Total watershed area receiving COPC deposition	m^2	**Varies** This variable is site-specific. See Chapter 4 for procedures to calculate this variable. Uncertainties associated with this variable are site-specific.
A_I	Impervious watershed area receiving COPC deposition	m^2	**Varies** This variable is site-specific. See Chapter 4 for procedures to calculate this variable. Uncertainties associated with this variable are site-specific.

TABLE B-4-10

PERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and is calculated using the equation presented in Table B-4-1. Uncertainties associated with this variable are site-specific.
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default *BD* value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended range of soil bulk density values may not accurately represent site-specific soil conditions.
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if a representative watershed soil can be identified. However, we recommend using a default value of 0.2 mL/cm^3; this value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with other U.S. EPA (1994b) and NC DEHNR (1997) guidance. The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, K_R may be under- or overestimated to a limited extent.
Kd_s	Soil-water partition coefficient	cm^3 water/g soil	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.
0.01	Units conversion factor	cm^2 water-kg soil-g COPC/m^2-gsoil-mgCOPC	

TABLE B-4-10

PERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Volume 2: pages 11-24.

Geraghty, J.J., D.W Miller, F. Van der Leeden, and F.L. Troise. 1973. *Water Atlas of the United States*. Water Information Center. Port Washington, New York.

This document is cited by U.S. EPA (1993), U.S. EPA (1994c), and NC DEHNR (1997) as a reference for calculating average annual runoff, *RO*. Specifically, this reference provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and ground water recharge. Because these volumes are total contributions and not only surface runoff, U.S. EPA (1994c) notes that they need to be reduced to estimate surface runoff. U.S. EPA (1994c) recommends a reduction of 50 percent.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Pres, Inc. New York.

This document is cited by U.S. EPA (1990) for the statement that soil bulk density, *BD*, is affected by soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is one of the source documented that cites the use of the equation in Table B-4-10; however, the document is not the original source of this equation. This document also recommends the following:

- Estimation of average annual runoff, *RO* (cm/yr), by using the *Water Atlas of the United States* (Geraghty et al. 1973) or site-specific procedures, such as the U.S. Soil Conservation Service CNE; U.S. EPA (1985) is cited as an example of the use of the CNE
- A default value of 0.2 (mL water/cm^3 soil) for soil volumetric content (θ_{sw})

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedures for Toxic and Conventional Pollutants in Surface and Ground Water - Part I (Revised - 1985)*. Environmental Research Laboratory. Athens, Georgia. EPA/600/6-85/002a. September.

U.S. EPA. 1994. *Revised Draft Guidance of Performing Screening Level Risk Analysis 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.

This document recommends (1) a default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988), and (2) a default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil), based on U.S. EPA (1993).

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-10

PERVIOUS RUNOFF LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document cites Hillel (1980) for the statement that only soil bulk density, *BD*, is affected by the soil structure, such as loosened or compaction of the soil, depending on the water and clay content of the soil.

This document is also a source of the following:

- A range of soil volumetric water content (θ_{sw}) values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil

TABLE B-4-11

EROSION LOAD TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the load to the water body from soil erosion. Uncertainties associated with this equation include the following:

(1) Uncertainties associated with the variables X_e, A_S, A_I, and Cs, are site-specific and may be significant in some cases.

(2) Uncertainties associated with the remaining variables aren't expected to be significant, primarily because of the narrow ranges of probable values for these variables or the use of well-established estimation procedures (Kd_s).

Equation

$$L_E = X_e \cdot (A_L - A_I) \cdot SD \cdot ER \cdot \frac{Cs \cdot Kd_s \cdot BD}{\theta_{sw} + Kd_s \cdot BD} \cdot 0.001$$

For mercury modeling, the erosion load to water body is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) using their respective Cs values and Kd_s values.

Variable	Description	Units	Value
L_E	Soil erosion load	g COPC/yr	
X_e	Unit soil loss	kg soil/m^2-yr	**Varies** This variable is site-specific, and calculated using the equation presented in Table B-4-13. The following uncertainty is associated with this variable: All of the equation variables are site-specific. Using default values rather than site-specific values, for any or all or these variables, will result in estimates of unit soil loss, X_e, that are under- or overestimated to some degree. The range of X_e calculated on the basis of default values spans slightly more than one order of magnitude (0.6 to 36.3 kg/m^2-yr).
A_L	Total watershed area receiving deposition	m^2	**Varies** This variable is site-specific (see Chapter 4). Uncertainties associated with this variable are site-specific.
A_I	Area of impervious watershed receiving deposition	m^2	**Varies** This variable is site-specific (see Chapter 4). Uncertainties associated with this variable are site-specific.

TABLE B-4-11

EROSION LOAD TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
SD	Watershed sediment delivery ratio	unitless	**Varies** This value is site-specific and calculated using equation in Table B-4-14. The following uncertainty is associated with this variable: The recommended default values for the variables *a* and *b* (empirical intercept coefficient and empirical slope coefficient, respectively) are average values, based on a review of sediment yields from various watersheds. These default values may not accurately represent site-specific watershed conditions and, therefore, may contribute to under- or over estimating L_E.
ER	Soil enrichment ratio	unitless	**1 or 3** COPC enrichment occurs because (1) lighter soil particles erode more quickly than heavier soil particles and (2) concentrations of organic COPCs—a function of organic carbon content of sorbing media—are expected to be higher in eroded material than *in situ* soil (U.S. EPA 1998). In the absence of site-specific data, we recommend a default value of 3 for organic COPCs and 1 for inorganic COPCs. This is consistent with other Agency guidance (U.S. EPA 1998), which recommends a range of 1 to 5 and a value of 3 as a "reasonable first estimate". This range has been used for organic matter, phosphorus, and other soil-bound COPCs (U.S. EPA 1998); however, no sources or references were provided for this range. *ER* is generally higher in sandy soils than in silty or loamy soils (U.S. EPA 1998). The following uncertainty is associated with this variable: The default *ER* value may not accurately reflect site-specific conditions; therefore, L_E may be over- or underestimated to an unknown, but relatively small, extent. Using county-specific *ER* values will reduce the extent of any uncertainties.
Cs	Average soil concentration over exposure duration	mg COPC/kg soil	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-4-1. Uncertainties are site-specific.
Kd_s	Soil-water partition coefficient	mL water/g soil (or cm^3 water/g soil)	**Varies** This variable is COPC-specific. We discuss this variable in detail and offer COPC-specific values in Appendix A-2. The following uncertainty is associated with this variable: Uncertainties associated with this parameter will be limited if Kd_s values are calculated as described in Appendix A-2.

TABLE B-4-11

EROSION LOAD TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
BD	Soil bulk density	g soil/cm^3 soil	**1.5** This variable is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil (Hillel 1980), as summarized in U.S. EPA (1998). U.S. EPA (1994c) recommended a default BD value of 1.5 g soil/cm^3 soil, based on a mean value for loam soil obtained from Carsel et al. (1988). U.S. EPA (1998) stated that a value of 1.5 would suffice for most uses, if site-specific information was unavailable. The following uncertainty is associated with this variable: The recommended BD value may not accurately represent site-specific soil conditions; and may under- or overestimate site-specific soil conditions to an unknown degree.
θ_{sw}	Soil volumetric water content	mL water/cm^3 soil	**0.2** This variable is site-specific, and depends on the available water and on soil structure; you can estimate θ_{sw} as the midpoint between a soil's field capacity and wilting point, if you can identify a representative watershed soil. However, we recommend using 0.2 ml/cm^3 as a default value. This value is the midpoint of the range 0.1 (very sandy soils) to 0.3 (heavy loam/clay soils) recommended by U.S. EPA (1998) (no source or reference is provided for this range) and is consistent with U.S. EPA (1994b). The following uncertainty is associated with this variable: The default θ_{sw} value may not accurately reflect site-specific or local conditions; therefore, L_E may be under- or overestimated to a small extent, based on the limited range of values.
0.001	Units conversion factor	mgCOPC/ g COPC	

TABLE B-4-11

EROSION LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

REFERENCES AND DISCUSSION

Carsel, R.F., R.S. Parrish, R.L. Jones, J.L. Hansen, and R.L. Lamb. 1988. "Characterizing the Uncertainty of Pesticide Leaching in Agricultural Soils." *Journal of Contaminant Hydrology.* Volume 2. Pages 11-24.

This document is the source for a mean soil bulk density, *BD*, of 1.5 (g soil/cm^3 soil) for loam soil.

Hillel, D. 1980. *Fundamentals of Soil Physics*. Academic Press, Inc. New York.

This document is cited by U.S. EPA (1998) for the statement that soil bulk density, *BD*, is affected by the soil structure, such as looseness or compaction of the soil, depending on the water and clay content of the soil.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is cited as one of the sources for the range of *BD* values, and the default value for the volumetric soil water content.

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document recommends (1) a default soil bulk density value of 1.5 (g soil/cm^3 soil), based on a mean value for loam soil from Carsel et al. (1988), and (2) a default soil volumetric water content, θ_{sw}, value of 0.2 (mL water/cm^3 soil).

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the source of a range of COPC enrichment ratio, *ER*, values. The recommended range, 1 to 5, was used for organic matter, phosphorous, and other soul-bound COPCs. This document recommends a value of 3 as a "reasonable first estimate," and states that COPC enrichment occurs because lighter soil particles erode more quickly than heavier soil particles. Lighter soil particles have higher ratios of surface area to volume and are higher in organic matter content. Therefore, concentration of organic COPCs, which is a function of the organic carbon content of sorbing media, is expected to be higher in eroded material than in *in situ* soil.

This document is also a source of the following:

- A range of soil volumetric water content (θ_{sw}) values of 0.1 ml water/cm^3 soil (very sandy soils) to 0.3 ml water/cm^3 soil (heavy loam/clay soils). However, no source or reference is provided for this range.
- A range of values for soil mixing zone depth, Z_s, for tilled and untilled soil

TABLE B-4-12

DIFFUSION LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the load to the water body due to vapor phase diffusion. Uncertainties associated with this equation include the following:

(1) Most of the uncertainties associated with the variables K_v, Q, $Cywv$, and A_w, are site-specific.

(2) This equation assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.

Equation

$$L_{dif} = \frac{K_v \cdot Q \cdot F_v \cdot Cywv \cdot A_w \cdot 1\times10^{-06}}{\frac{H}{R \cdot T_{wk}}}$$

For Mercury modeling

$$L_{dif_{(Mercury)}} = \frac{K_v \cdot (0.48Q_{(Total)}) \cdot F_{v_{(Hg^{2+})}} \cdot Cywv \cdot A_w \cdot 1\times10^{-06}}{\frac{H}{R \cdot T_{wk}}}$$

Use *0.48Q* for total mercury (to account for loss to the global cycle) and F_v = 0.85 in the mercury modeling equation.

Variable	Description	Units	Value
L_{dif}	Vapor phase diffusion load to water body	g COPC/yr	
K_v	Overall transfer rate coefficient	m/yr	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-4-19. Uncertainties associated with this variable are site-specific.

TABLE B-4-12

DIFFUSION LOAD TO WATER BODY
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
Q	COPC-specific emission rate	g COPC/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are site-specific.
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. Values are also presented in U.S. EPA (1994), RTI (1992), and NC DEHNR (1997). Values are based on the work of Bidleman (1988), as cited in U.S. EPA (1994) and NC DEHNR (1997). U.S. EPA (1994) presents values for organic COPCs that range from 0.27 to 1. All values presented by U.S. EPA (1994) for inorganic COPCs are given as 0. Uncertainties associated with this variable include the following: (1) This equation assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c issued to calculate F_v.
$Cywv$	Unitized yearly average air concentration from vapor phase	μg-s/g-m^3	**Varies** This variable is COPC- and site-specific, and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are site-specific.
A_w	Water body surface area	m^2	**Varies** This variable is site-specific (see Chapter 4). Uncertainties associated with this variable are site-specific. However, we expect that the uncertainty associated with this variable will be limited, because maps, aerial photographs, and other resources from which water body surface areas can be measured, are readily available.
10^{-6}	Units conversion factor	g/μg	

TABLE B-4-12

DIFFUSION LOAD TO WATER BODY (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Values for this variable, estimated by using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, L_{Dif} may be under- or overestimated to a limited degree.
R	Universal gas constant	atm-m^3/mol-K	**8.205 x 10^{-5}**
T_{wk}	Water body temperature	K	**298** This variable is site-specific. We recommend using this default value in the absence of site-specific information, consistent with U.S. EPA (1998) and U.S. EPA (1994). The following uncertainty is associated with this variable: To the extent that the default water body temperature value does not accurately represent site-specific or local conditions, L_{dif} will be under- or overestimated.

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. *"Atmospheric Processes." Environmental Science and Technology.* Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion in Table B-1-1.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is a reference source for the equation in Table B-4-12. This document also recommends using the equations in Bidleman (1988) to calculate F_v values for all organics other than dioxins (PCDD/PCDFs). However, the document does not present a recommendation for dioxins. This document also states that metals are generally entirely in the particulate phase ($F_v = 0$), except for mercury, which is assumed to be entirely in the vapor phase. The document does not state whether F_v for mercury should be calculated by using the equations in Bidleman (1988); We assume that this is the case.

U.S. EPA 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is cited as the reference source for $T_{wk,}$ water body temperature (298 K); however, no references or sources are identified for this value. This document is a reference source for the equation in Table B-4-8. This document also presents values for organic COPCs that range from 0.27 to 1. F_v values for organics other than PCDD/PCDFs are calculated by using the equations presented in Bidleman (1988). The F_v value for PCDD/PCDFs is assumed to be 0.27, based on Lorber (no date). Finally, this document presents F_v values for inorganic COPCs equal to 0, based on the assumption that these COPCs are nonvolatile and 100 percent in the particulate phase and 0 percent in the vapor phase.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends a range (10°C to 20°C; 283 K to 303 K) for water body temperature, T_{wk}. No source was identified for this range.

TABLE B-4-13

UNIVERSAL SOIL LOSS EQUATION (USLE)
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the soil loss rate from the watershed by using the Universal Soil Loss Equation (USLE); the result is used in the soil erosion load equation in Table B-4-11. Estimates of unit soil loss, X_e, should be determined specific to each watershed evaluated. Information on determining site- and watershed-specific values for variables used in calculating X_e is provided in U.S. Department of Agriculture (U.S. Department of Agriculture 1997) and U.S. EPA guidance (U.S. EPA 1985). Uncertainties associated with this equation include the following:

(1) All of the equation variables are site-specific. Use of default values will result in estimates of unit soil loss, X_e, that are under- or overestimated to some unknown degree.

Equation

$$X_e = RF \cdot K \cdot LS \cdot C \cdot PF \cdot \frac{907.18}{4047}$$

Variable	Description	Units	Value
X_e	Unit soil loss	kg/m^2-yr	
RF	USLE rainfall (or erosivity) factor	yr^{-1}	**50 to 300** This value is site-specific and is derived on a storm-by-storm basis. As cited in U.S. EPA (1998), average annual values were compiled regionally by Wischmeier and Smith (1978); the recommended range reflects these compiled values. The following uncertainty is associated with this variable: The range of average annual rainfall factors (50 to 300) from Wischmeier and Smith (1978) may not accurately reflect site-specific conditions. Therefore, unit soil loss, X_e, may be under- or overestimated.
K	USLE erodibility factor	ton/acre	**Varies** This value is site-specific. We recommend using current guidance (U.S. Department of Agriculture 1997; U.S. EPA 1985) in determining watershed-specific values for this variable based on site-specific information. A default value of 0.39, as cited in NC DEHNR (1997) and U.S. EPA (1994), was based on a soil organic matter content of 1 percent (Droppo et al. 1989), and chosen to be representative of a whole watershed, not just an agricultural field. The following uncertainty is associated with this variable: Using a site-specific USLE soil erodibility factor, K, may cause unit soil loss, X_e, to be under- or overestimated to some unknown degree.

TABLE B-4-13

UNIVERSAL SOIL LOSS EQUATION (USLE)
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
LS	USLE length-slope factor	unitless	**Varies** This value is site-specific. We recommend using current guidance (U.S. Department of Agriculture 1997; U.S. EPA 1985) in determining watershed-specific values for this variable based on site-specific information. A value of 1.5 as cited in NC DEHNR (1997) and U.S. EPA (1994), reflects a variety of possible distance and slope conditions (U.S. EPA 1988), and was chosen to be representative of a whole watershed, not just an agricultural field. The following uncertainty is associated with this variable: A site-specific USLE length-slope factor, LS, may not accurately represent site-specific conditions. Therefore, unit soil loss, X_e, may be under- or overestimated to some unknown degree.
C	USLE cover management factor	unitless	**Varies** This value is site-specific. We recommend using current guidance (U.S. Department of Agriculture 1997; U.S. EPA 1985) in determining watershed-specific values for this variable based on site-specific information. The range of values up to 0.1 reflect dense vegetative cover, such as pasture grass; values from 0.1 to 0.7 reflect agricultural row crops; and a value of 1.0 reflects bare soil (U.S. EPA 1998). U.S. EPA (1993) recommended a value of 0.1 for both grass and agricultural crops. This range of values was also cited in NC DEHNR (1997). However, U.S. EPA (1994) and NC DEHNR (1997) both recommended a default value of 0.1 to be representative of a whole watershed, not just an agricultural field. The following uncertainty is associated with this variable: The USLE cover management factor, C, value determined may not accurately represent site-specific conditions. Therefore, the value for C may result in the under- or overestimation of unit soil loss, X_e.
PF	USLE supporting practice factor	unitless	**Varies** This value is site-specific. We recommend using current guidance (U.S. Department of Agriculture 1997; U.S. EPA 1985) in determining watershed-specific values for this variable based on site-specific information. A default value of 1.0, which conservatively represents the absence of any erosion or runoff control measures, was cited in NC DEHNR (1997) and U.S. EPA (1998; 1994). The following uncertainty is associated with this variable: Using a site-specific USLE supporting practice factor, PF, may result in under- or overestimating X_e depending on the actual extent that there are erosion or runoff control measures in the vicinity of the watershed evaluated.
907.18	Units conversion factor	kg/ton	
4047	Units conversion factor	m^2/acre	

TABLE B-4-13

UNIVERSAL SOIL LOSS EQUATION (USLE)
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

REFERENCES AND DISCUSSION

Droppo, J.G. Jr., D.L. Strenge, J.W. Buck, B.L. Hoopes, R.D. Brockhaus, M.B. Walter, and G. Whelan. 1989. *Multimedia Environmental Pollutant Assessment System (MEPAS) Application Guidance: Volume 2-Guidelines for Evaluating MEPAS Input Parameters*. Pacific Northwest Laboratory. Richland, Washington. December.

This document is cited by U.S. EPA 1994 and NC DEHNR 1997 as the reference source for a USLE erodibility factor value of 0.36, based on a soil organic matter content of 1 percent.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document recommended the following:

- A USLE erodibility factor, *K*, value of 0.36 ton/acre
- A USLE length-slope factor, *LS*, value of 1.5 (unitless)
- A range of USLE cover management factor, *C*, values of 0.1 to 1.0; it also recommended a value of 0.1 to be representative of a whole watershed, not just an agricultural field.
- A USLE supporting practice factor, *PF*, value of 1.0

U.S. Department of Agriculture. 1997. *Predicting Soil Erosion by Water: A Guide to Conservation Planning With the Revised Universal Soil Loss Equation (RUSLE)*. Agricultural Research Service, Agriculture Handbook Number 703. January.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water—Part I (Revised)*. ORD. Athens, Georgia. EPA/600/6-85/002a.

U.S. EPA. 1988. *Superfund Exposure Assessment Manual.* Office of Solid Waste. Washington, D.C. April.

This document is cited by U.S. EPA 1994 and NC DEHNR 1997 as the reference source for the USLE length-slope factor, *LS*, value of 1.5. This value reflects a variety of possible distance and slope conditions and was chosen to be representative of a whole watershed, not just an agricultural field.

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. Working Group Recommendations.* Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document recommends the following:

- A USLE cover management factor, *C*, of 0.1 for both grass and agricultural crops
- A USLE supporting practice factor, *PF*, of 1.0, based on the assumed absence of any erosion or runoff control measures

TABLE B-4-13

UNIVERSAL SOIL LOSS EQUATION (USLE)
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1994. *Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes.* Office of Emergency and Remedial Response. Office of Solid Waste. December.

This document recommends the following:

- A USLE erodibility factor, *K*, value of 0.36 ton/acre
- A USLE length-slope factor, *LS*, value of 1.5 (unitless)
- A range of USLE cover management factor, *C*, values of 0.1 to 1.0; it recommends a default value of 0.1 to be representative of a whole watershed, not just an agricultural field.
- A USLE supporting practice factor, *PF*, value of 1.0

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document cites Wischmeier and Smith (1978) as the source of average annual USLE rainfall factors, *RF*, and states that annual values range from less than 50 for the arid western United States to greater than 300 for the southeast.

This document discusses the USLE cover management factor. This factor, *C*, primarily reflects how erosion is influenced by vegetative cover and cropping practices, such as planting across slope rather than up and down slope. This document discusses a range of *C* values for 0.1 to 1.0; values greater than 0.1 but less than 0.2 are appropriate for agricultural row crops, and a value of 1.0 is appropriate for sites mostly devoid of vegetation.

Wischmeier, W.H., and D.D. Smith. 1978. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning.* Agricultural Handbook No. 537. U.S. Department of Agriculture Washington, D.C.

This document is cited by U.S. EPA (1998) as the source of average annual USLE rainfall factors, *RF*, compiled regionally. According to U.S. EPA (1998), annual values range from less than 50 for the arid western United States to greater than 300 for the southeast.

TABLE B-4-14

SEDIMENT DELIVERY RATIO
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the sediment delivery ratio for the watershed; the result is used in the soil erosion load equation in Table B-4-11.

Uncertainties associated with this equation include the following:

(1) The recommended default empirical intercept coefficient (*a*) values are average values based on various studies of sediment yields from various watersheds. Therefore, these default values may not accurately represent site-specific watershed conditions. As a result, using these default values may under- or overestimate the watershed sediment delivery ratio, *SD*.

(2) The recommended default empirical slope coefficient (*b*) value is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, using this default value may under- or overestimate the watershed sediment delivery ratio, *SD*.

Equation

$$SD = a \cdot (A_L)^{-b}$$

Variable	Description	Units	Value
SD	Watershed sediment delivery ratio	unitless	
a	Empirical intercept coefficient	unitless	**0.6 to 2.1** This variable is site-specific and is determined on the basis of the watershed area (Vanoni 1975), as cited in U.S. EPA (1998): Watershed Area (sq. miles) — "*a*" Coefficient (unitless) ≤ 0.1 — 2.1 1 (>0.1 but ≤ 1.0) — 1.9 10 (>1.0 but ≤ 10) — 1.4 100 (>10 but ≤ 100) — 1.2 1,000 (>100) — 0.6 Note: 1 sq. mile = 2.59×10^6 m^2 Using these values is consistent with U.S. EPA (1994a), U.S. EPA (1994b), and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended default empirical intercept coefficient, *a*, values are average values based on various studies of sediment yields from various watersheds. Therefore, these default values may not accurately represent site-specific watershed conditions. As a result, using these default values may under- or overestimate the watershed sediment delivery ratio, *SD*.

TABLE B-4-14

SEDIMENT DELIVERY RATIO (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
A_L	Total watershed area receiving deposition	m^2	**Varies** This variable is site-specific (see Chapter 4). Uncertainties associated with this variable are site-specific.
b	Empirical slope coefficient	unitless	**0.125** As cited in U.S. EPA (1998), this variable is an empirical constant based on the research of Vanoni (1975), which concludes that sediment delivery ratios vary approximately with negative one-eighth ($^-1/8$) power of the drainage area. The use of this value is consistent with U.S. EPA (1994a), U.S. EPA (1994b), and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended default empirical slope coefficient, *b*, value is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, use of this default value may under- or overestimate the watershed sediment delivery ratio, *SD*.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the reference source documents for the empirical intercept coefficient, *a*, and empirical slope coefficient, *b*, values. This document cites U.S. EPA (1993) as the source of its information.

U.S. EPA. 1994a. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April.

This document is cited as one of the reference source documents for the empirical intercept coefficient, *a*, and empirical slope coefficient, *b*, values. This document does not identify Vanoni (1975) as the source of its information.

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is cited as one of the reference source documents for the empirical intercept coefficient, *a*, and the empirical slope coefficient, *b*, values. This document cites U.S. EPA (1993) as the source of its information.

TABLE B-4-14

SEDIMENT DELIVERY RATIO
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is cited as one of the reference source documents for the empirical intercept coefficient, *a*, and empirical slope coefficient, *b*, values. This document cites Vanoni (1975) as its source of information.

Vanoni, V.A. 1975. *Sedimentation Engineering*. American Society of Civil Engineers. New York, New York. Pages 460-463.

This document is cited by U.S. EPA (1998) as the source of the equation in Table B-4-14 and the empirical intercept coefficient, *a*, and empirical slope coefficient, *b*, values. Based on various studies of sediment yields from watersheds, this document concludes that the sediment delivery ratios vary approximately with negative one-eighth ($^{-}1/8$) power of the drainage ratio. U.S. EPA has not completed a review of this document.

TABLE B-4-15

TOTAL WATER BODY CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the total water body concentration, including the water column and the bed sediment.

Uncertainties associated with this equation include the following:

(1) The default variable values recommended for use in the equation in Table B-4-15 may not accurately represent site-specific water body conditions. The degree of uncertainty associated with the variables Vf_x, A_w, d_{wc}, *and* d_{bs} is expected to be limited either because the probable ranges for these variables are narrow or information allowing accurate estimates is generally available.

(2) Uncertainty associated with f_{wc} is largely the result of uncertainty associated with default organic carbon (OC) content values and may be significant in specific instances. Uncertainties associated with the total core load into water body (L_T) and overall total water body core dissipation rate constant (k_{wt}) may also be significant in some instances because of the summation of many variable-specific uncertainties.

Equation

$$C_{wtot} = \frac{L_T}{Vf_x \cdot f_{wc} + k_{wt} \cdot A_w \cdot (d_{wc} + d_{bs})}$$

Variable	Description	Units	Value
C_{wtot}	Total water body COPC concentration, including water column and bed sediment	g COPC/m^3 water body (equivalent to mg COPC/L water body)	
L_T	Total COPC load to the water body, including deposition, runoff, and erosion	g/yr	**Varies** This variable is COPC- and site-specific, and calculated using the equation in Table B-4-7. Uncertainties associated with L_{DEP}, L_{Dif}, L_{RI}, L_R, and L_E, as presented in the equation in Table B-4-7, are also associated with L_T.

TABLE B-4-15

TOTAL WATER BODY CONCENTRATION
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
Vf_x	Average volumetric flow rate through water body	m^3/yr	**Varies** This variable is site-specific. The following uncertainty is associated with this variable: Default average volumetric flow rate (Vf_x) information may not accurately represent site-specific conditions, especially for those water bodies for which flow rate information is not readily available. Therefore, using default Vf_x values may contribute to under- or overestimating total water body COPC concentration, C_{wtot}.
f_{wc}	Fraction of total water body COPC concentration in the water column	unitless	**0 to 1** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-16. The following uncertainty is associated with this variable: The default values for the variables in the equation in Table B-4-16 may not accurately represent site- and water body - specific conditions. However, the range of several variables—including d_{bs}, C_{BS}, and θ_{bs}—is relatively narrow. Other variables, such as d_{wc} and d_z, can be reasonably estimated on the basis of generally available information. The largest degree of uncertainty may be introduced by the default medium-specific organic carbon (*OC*) content values. Because *OC* content values may vary widely in different locations in the same medium, using default values may result in insignificant uncertainty in specific cases.
k_{wt}	Overall total water body dissipation rate constant	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-17. The following uncertainty is associated with this variable: All of the variables in the equation in Table B-4-17 are site-specific; therefore, using default values for any or all of these variables will contribute to under- or overestimating C_{wtot}. The degree of uncertainty associated with the variable K_b is expected to be under one order of magnitude and is associated largely with the estimation of the unit soil loss, X_e, values for the variables f_{wc}, K_v, and f_{bs} are dependent on medium-specific estimates of *OC* content. Because *OC* content can vary widely for different locations in the same medium, uncertainty associated with these three may be significant in specific instances.
A_w	Water body surface area	m^2	**Varies** This variable is site-specific. The value you select represents an average value for the entire year. See Chapter 4 for procedures to determine this variable. Uncertainties associated with this variable are site-specific. However, we expect that the uncertainty associated with this variable will be limited because maps, aerial photographs, and other resources from which water body surface areas can be measured, are readily available.

TABLE B-4-15

TOTAL WATER BODY CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
d_{wc}	Depth of water column	m	**Varies** This variable is site-specific. The value you select represents an average value for the entire year. The following uncertainty is associated with this variable: Depth of water column, d_{wc}, values may not accurately reflect site-specific conditions, especially for those water bodies for which depth of water column information is unavailable or outdated. Therefore, using d_{wc} values may contribute to under-or overestimating total water body COPC concentration, C_{wtot}.
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. The value you select represents an average value for the entire year. We recommend a default upper benthic sediment depth of 0.03 meter, which is consistent with U.S. EPA (1994) and NC DEHNR (1997) guidance. This value was cited by U.S. EPA (1993); however, no reference was presented. U.S. EPA (1998) suggests a range of values, from 0.01 to 0.05 meters. The following uncertainty is associated with this variable: Default d_{bs} values may not accurately represent site-specific water body conditions. However, based on the narrow recommended range, we expect any uncertainty introduced to be limited.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is also cited as one of the reference source documents for the default depth of upper benthic layer value. The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993) as its source of information for the range of values for the depth of the upper benthic layer.

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September 24.

This document is cited by NC DEHNR (1997) and U.S. EPA (1994) as the source of the range and default value for the depth of the upper benthic layer (d_{bs}).

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 15.

This document is cited as one of the reference source documents for the default depth of the upper benthic layer value. The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993) as its source of information for the range of values for the depth of the upper benthic layer.

TABLE B-4-15

TOTAL WATER BODY CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is cited as the source of a range of values for the depth of the upper benthic layer (d_{bs}).

TABLE B-4-16

FRACTION IN WATER COLUMN AND BENTHIC SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the fraction of total water body concentration occurring in the water column and the bed sediments.

Uncertainties associated with this equation include the following:

> The default variable values may not accurately represent site-specific water body conditions. However, the range of several variables—including d_{bs}, C_{BS}, and θ_{bs}—is relatively narrow. Other variables, such as d_{wc} and d_z, can be reasonably estimated on the basis of generally available information. The largest degree of uncertainty may be introduced by the default medium-specific OC content values. *OC* content values can vary widely for different locations in the same medium. Therefore, the use of default values may introduce significant uncertainty in some cases.

Equations

$$f_{wc} = \frac{(1 + Kd_{sw} \cdot TSS \cdot 1\times10^{-6}) \cdot d_{wc}/d_z}{(1 + Kd_{sw} \cdot TSS \cdot 1\times10^{-6}) \cdot d_{wc}/d_z + (\theta_{bs} + Kd_{bs} \cdot C_{BS}) \cdot d_{bs}/d_z}$$

$$f_{bs} = 1 - f_{wc}$$

Variable	Description	Units	Value
f_{wc}	Fraction of total water body COPC concentration in the water column	unitless	
f_{bs}	Fraction of total water body COPC concentration in benthic sediment	unitless	

TABLE B-4-16

FRACTION IN WATER COLUMN AND BENTHIC SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
Kd_{sw}	Suspended sediments/surface water partition coefficient	L water/kg suspended sediment (or cm^3 water/kg suspended sediment)	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Kd_{sw} values in the HHRAP companion database are based on default OC contents for surface water and soil. Kd_{sw} values based on default values may not accurately reflect site- and water body-specific conditions and may under- or overestimate actual Kd_{sw} values. You can reduce uncertainty associated with Kd_{sw} by using site-specific and medium-specific OC estimates to calculate Kd_{sw}.
TSS	Total suspended solids concentration	mg/L	**2 to 300** This variable is site-specific. We recommend using site- and waterbody specific measured values, representative of long-term average annual values for the water body of concern (see Chapter 5). A value of 10 mg/L was cited by NC DEHNR (1997) and U.S. EPA (1993) in the absence of site-specific measured data. The following uncertainty is associated with this variable: Limitation on measured data used for determining a water body specific total suspended solids (TSS) value may not accurately reflect site- and water body-specific conditions long term. Therefore, the TSS value may contribute to under- or overestimating f_{wc}.
1×10^{-6}	Units conversion factor	kg/mg	
d_{wc}	Depth of water column	m	**Varies** This variable is site-specific. The value you select represents an average value for the entire year. The following uncertainty is associated with this variable: Depth of water column, d_{wc}, values may not accurately reflect site-specific conditions, especially for those water bodies for which depth of water column information is unavailable or outdated. Therefore, using d_{wc} values may contribute to under- or overestimating total water body COPC concentration, C_{wtot}.

TABLE B-4-16

FRACTION IN WATER COLUMN AND BENTHIC SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. The value you select represents an average value for the entire year. We recommend a default upper benthic sediment depth of 0.03 meter, which is consistent with U.S. EPA (1994) and NC DEHNR (1997) guidance. This value was cited by U.S. EPA (1993); however, no reference was presented. U.S. EPA (1998) suggests a range of values, from 0.01 to 0.05 meter. The following uncertainty is associated with this variable: A default d_{bs} value may not accurately represent site-specific water body conditions. However, we expect any uncertainty introduced to be limited on the basis of the narrow recommended range.
d_z	Total water body depth	m	**Varies** This variable is site-specific. We recommend using the following equation to calculate total water body depth, consistent with NC DEHNR (1997): $d_z = d_{wc} + d_{bs}$ The following uncertainty is associated with this variable: Calculating this variable sums the concentrations associated with the two variables d_{wc} and d_{bs}. Because most of the total water body depth (d_z) is made up of the depth of the water column (d_{wc}), and we don't expect the uncertainties associated with d_{wc} to be significant, we likewise don't expect the total uncertainties associated with d_z to be significant.
C_{BS}	Bed sediment concentration (or bed sediment bulk density)	g/cm^3 (equivalent to kg/L)	**1.0** This variable is site-specific. We recommend a default value of 1.0, consistent with U.S. EPA (1998), which states that this value should be reasonable for most applications. The recommended default value is also consistent with other U.S. EPA (1994), and NC DEHNR (1997) guidance. The following uncertainty is associated with this variable: The recommended default value may not accurately represent site- and water body-specific conditions. Therefore, the variable f_{wc} may be under- or overestimated. Based on th narrow recommended range, we expect the under- or overestimation will be limited.

TABLE B-4-16

FRACTION IN WATER COLUMN AND BENTHIC SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
θ_{bs}	Bed sediment porosity	$L_{water}/L_{sediment}$	**0.6** This variable is site-specific. We recommend a default bed sediment porosity of 0.6 (by using a C_{BS} value of 1 g/cm^3 and a solid density (ρ_s) value of 2.65 kg/L) calculated by using the following equation (U.S. EPA 1998): $\theta_{bs} = 1 - C_{BS}/\rho_s$ This is consistent with other U.S. EPA (1994), and NC DEHNR (1997) guidance. The following uncertainty is associated with this variable: Calculation of this variable combines the uncertainties associated with the two variables, C_{BS} and ρ_s, used in the calculation. To the extent that the recommended default values of C_{BS} and ρ_s don't accurately represent site- and water body-specific conditions, θ_{bs} will be under- or overestimated.
Kd_{bs}	Bed sediment/sediment pore water partition coefficient	L water/kg bottom sediment (or cm^{3}water/g bottom sediment)	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The Kd_{bs} values in the HHRAP companion database are based on default *OC* contents for sediment and soil. Kd_{bs} values based on default *OC* values may not accurately represent site- and water body-specific conditions and may under- or overestimate actual Kd_{bs} values. Uncertainty associated with this variable will be reduced if site- and water body-specific *OC* estimates are used to calculate Kd_{bs}.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of

- the range of Kd_s values
- *TSS* values. This document cites U.S. EPA (1993b) as its source of information.
- the equation for calculating total water body depth. No source of this equation was identified.
- the default value for bed sediment porosity. This document cites U.S. EPA (1993b) as its source of information.
- the default value for depth of the upper benthic layer. The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993b) as its source of information for the range of values for the depth of the upper benthic layer.
- the default bed sediment concentration. This document cites U.S. EPA (1993b) as its source of information.

TABLE B-4-16

FRACTION IN WATER COLUMN AND BENTHIC SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document is cited by NC DEHNR (1997) as the source of the *TSS* value. This document is also cited by NC DEHNR (1997) and U.S. EPA (1994) as the source of the default bed sediment porosity value and the equation used to calculate C_{BS}, and the range for the depth of the upper benthic layer values.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April.

This document is cited as one of the reference source documents for
- the default value for bed sediment porosity. This document cites U.S. EPA (1993b) as its source of information.
- the default value for depth of the upper benthic layer. The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993b) as its source of information for the range of values for the depth of the upper benthic layer.
- the default bed sediment concentration. This document cites U.S. EPA (1993b) as its source of information.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is cited as one of the sources of
- the equation for calculating bed sediment porosity (θ_{bs}); no source of this equation was identified.
- the range of the bed sediment concentration (C_{BS}); no original source of this range was identified.

TABLE B-4-17

OVERALL TOTAL WATER BODY DISSIPATION RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 2)

Description

This equation calculates the overall COPC dissipation rate in surface water due to volatilization and benthic burial.

Uncertainties associated with this equation include the following:

(1) All of the variables in the equation in Table B-4-17 are site-specific. Therefore, using default values for any or all of these variables will contribute to under- or overestimating k_{wt}. We expect the uncertainty associated with the variable k_b to be one order of magnitude at most. This uncertainty is associated with the estimation of the unit soil loss, X_e (a component of k_b). Values for the variables f_{wc}, k_v, and f_{bs} are dependent on medium-specific estimates of medium-specific *OC* content. Because *OC* content can vary widely for different locations in the same medium, uncertainty associated with these three variables may be significant in specific instances.

Equation

$$k_{wt} = f_{wc} \cdot k_v + f_{bs} \cdot k_b$$

Variable	Description	Units	Value
k_{wt}	Overall total water body dissipation rate constant	yr^{-1}	
f_{wc}	Fraction of total water body COPC concentration in the water column	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-16. Uncertainties associated with this variable include the following: (1) The default variable values we recommend you use in the equation in Table B-4-16 may not accurately represent site-specific water body conditions. However, the ranges of several component variables—including d_{bs}, C_{BS}, and θ_{sw}—are moderate (factors of 5, 3, and 2, respectively). We therefore expect the degree of uncertainty associated with these variables to be moderate. You can reasonably estimate other variables, such as d_{wc} and d_z, using generally available information. We therefore expect the degree of uncertainty associated with these variables to be relatively small. (2) The largest degree of uncertainty may be introduced by the default medium-specific *OC* content values. *OC* content values are often not readily available and can vary widely for different locations in the same medium. Therefore, the degree of uncertainty may be significant in specific instances.

TABLE B-4-17

OVERALL TOTAL WATER BODY DISSIPATION RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 2)

Variable	Description	Units	Value
k_v	Water column volatilization rate constant	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-18. Uncertainties associated with this variable include the following: (1) All of the variables in the equation in Table B-4-18 are site-specific. Therefore, using default values for any or all of these variables could contribute to under- or overestimating k_v. (2) We expect the degree of uncertainty associated with k_v components d_z and *TSS* to be minimal, either because information needed to estimate these variables is generally available or because the range of likely values is narrow. (3) Values for the variable k_v and Kd_{sw} are dependent on medium-specific estimates of *OC* content. Because *OC* content can vary widely for different locations in the same medium, uncertainty associated with these two variables may be significant in specific instances.
f_{bs}	Fraction of total water body COPC concentration in benthic sediment	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-16. Uncertainties associated with this variable include the following: (1) The default variable values we recommend you use in the equation in Table B-4-16 may not accurately represent site-specific water body conditions. However, the ranges of several components—including d_{bs}, C_{BS}, and θ_{sw}—are relatively narrow. We therefore expect the degree of uncertainty associated with these variables to be relatively small. You can reasonably estimate other components, such as d_{wc} and d_z, using generally available information. (2) The largest degree of uncertainty may be introduced by the default medium-specific *OC* contact values. *OC* content values are often not readily available and can vary widely for different locations in the same medium. Therefore, the degree of uncertainty may be significant in specific instances.
k_b	Benthic burial rate constant	yr^{-1}	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-22. Uncertainties associated with this variable include the following: (1) All of the variables in the equation in Table B-4-22 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating K_b. (2) The degree of uncertainty associated with each of these variables is as follows: (1) X_e—about one order of magnitude at most, (2) C_{BS}, d_{bs}, Vf_x, *TSS,* and A_w—limited because of the narrow recommended ranges for these variables or because resources to estimate variable values are generally available, and (3) A_L and *SD*—very site-specific, degree of uncertainty unknown.

TABLE B-4-18

WATER COLUMN VOLATILIZATION LOSS RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the water column COPC loss rate constant due to volatilization. Uncertainty associated with this equation includes the following:

> All of the variables in the equation in Table B-4-18 are site-specific. Therefore, using default values for any or all of these variables will contribute to under- or over estimating k_v. We expect the uncertainty associated with the variables d_{wc}, d_{bs}, and d_z to be minimal, either because information necessary to estimate these variables is generally available or because the range of probable values is narrow. Values for the variables K_v and Kd_{sw} are dependent on medium-specific estimates of *OC* content. Because *OC* content can vary widely for different locations in the same medium, uncertainty associated with these two variables may be significant in specific instances.

Equation

$$k_v = \frac{K_v}{d_z \cdot (1 + Kd_{sw} \cdot TSS \cdot 10^{-6})}$$

Variable	Description	Units	Value
k_v	Water column volatilization rate constant	yr^{-1}	
K_v	Overall COPC transfer rate coefficient	m/yr	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-19. Uncertainties associated with this variable include the following: (1) All of the variables in the equation in Table B-4-19—except *R*, the universal gas constant, which is well-established—are site-specific. Therefore, using default values, for any or all these variables, could contribute to under- or overestimating K_v. (2) We expect the degree of uncertainty associated with the variables *H* and T_{wk} to be minimal. Values for *H* are well-established, and T_{wk} will likely vary less than 10 percent of the default value. (3) The uncertainty associated with the variables K_L and K_G is attributable largely to medium-specific estimates of organic carbon, *OC*, content. Because *OC* content values can vary widely for different locations in the same medium, using default values may generate significant uncertainty in specific instances. Finally, the origin of the recommended temperature correction factor, θ, value is unknown. The degree of associated uncertainty is therefore also unknown.

TABLE B-4-18

WATER COLUMN VOLATILIZATION LOSS RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
d_z	Total water body depth	m	**Varies** This variable is site-specific. We recommend using the following equation to calculate d_z, consistent with NC DEHNR (1997): $d_z = d_{wc} + d_{bs}$ The following uncertainty is associated with this variable: Calculating this variable sums the concentrations associated with the two variables d_{wc} and d_{bs}. Because most of the total water body depth (d_z) is made up of the depth of the water column (d_{wc}), and we don't expect the uncertainties associated with d_{wc} to be significant, we likewise don't expect the total uncertainties associated with d_z, to be significant.
d_{wc}	Depth of water column	m	**Varies** This variable is site-specific. The following uncertainty is associated with this variable: Default values for depth of water column, d_{wc}, may not accurately reflect site-specific conditions, especially for water bodies for which depth of water column information is unavailable or outdated. Therefore, using default d_{wc} values may contribute to under- or overestimating total water body COPC concentration, C_{wtot}. However, we don't expect the degree of under- or overestimating to be significant.
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. The value you select represents an average value for the entire year. We recommend a default upper benthic sediment depth of 0.03 meter, which is consistent with U.S. EPA (1994) and NC DEHNR (1997) guidance. This value was cited by U.S. EPA (1993); however, no reference was presented. U.S. EPA (1998) suggests a range of values, from 0.01 to 0.05 meter. The following uncertainty is associated with this variable: A default d_{bs} value may not accurately represent site-specific water body conditions. However, we expect any uncertainty introduced to be limited on the basis of the narrow recommended range.

TABLE B-4-18

WATER COLUMN VOLATILIZATION LOSS RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
Kd_{sw}	Suspended sediments/surface water partition coefficient	L water/kg suspended sediments	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The Kd_{sw} values presented in the HHRAP companion database were calculated on the basis of default *OC* contents for surface water and soil. Kd_{sw} values based on default values may not accurately reflect site-and water body-specific conditions and may under- or overestimate actual Kd_{sw} values. You can reduce uncertainty associated with this variable by using site-specific and medium-specific *OC* estimates to calculate Kd_{sw}.
TSS	Total suspended solids concentration	mg/L	**2 to 300** This variable is site-specific. We recommend using site- and waterbody specific measured values, representative of long-term average annual values for the water body of concern (see Chapter 5). A value of 10 mg/L was cited by NC DEHNR (1997) and U.S. EPA (1993) in the absence of site-specific measured data. The following uncertainty is associated with this variable: Limitation on measured data used to determine a water body specific total suspended solids (*TSS*) value may not accurately reflect site- and water body-specific conditions long term. Therefore, the *TSS* value may contribute to under-or overestimating f_{wc}.
1×10^{-6}	Units conversion factor	kg/mg	

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as the source of the equation for calculating total water body depth. No source of this equation was identified. This document is also cited as one of the sources of the range of Kd_s values and an assumed *OC* value of 0.075 for surface water. This document is also cited as one of the sources of *TSS*. This document cites U.S. EPA (1993b) as its source of information.

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of the range and default value for the depth of the upper benthic layer (d_{bs}). This document is also cited by NC DEHNR (1997) as the source of the *TSS* value.

TABLE B-4-18

WATER COLUMN VOLATILIZATION LOSS RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facility Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facility*. April 15.

This document is cited as one of the reference source documents for the default value of the depth of the upper benthic layer. The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993b) as its source of information.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

TABLE B-4-19

OVERALL COPC TRANSFER RATE COEFFICIENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the overall transfer rate of contaminants from the liquid and gas phases in surface water.

Uncertainties associated with this equation include the following:

(1) All of the variables in the equation in Table B-4-19—except *R*, the universal gas constant, which is well-established—are site-specific. Therefore, using default values for any or all of these variables will contribute to under- or overestimating K_v.

(2) We believe the degree of uncertainty associated with the variables *H* and T_{wk} to be minimal. Values for *H* are well-established, and T_{wk} will likely vary less than 10 percent of the default value.

(3) The uncertainty associated with the variables K_v and K_G is attributable largely to medium-specific estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, using default values may generate significant uncertainty in specific instances. Finally, the origin of the recommended value is unknown; therefore, the degree of associated uncertainty is also unknown.

Equation

$$K_v = \left[K_L^{-1} + \left(K_G \cdot \frac{H}{R \cdot T_{wk}} \right)^{-1} \right]^{-1} \cdot \theta^{(T_{wk} - 293)}$$

Variable	Description	Units	Value
K_v	Overall COPC transfer rate coefficient	m/yr	

TABLE B-4-19

OVERALL COPC TRANSFER RATE COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
K_L	Liquid phase transfer coefficient	m/yr	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-20. Uncertainties associated with this variable include the following: All of the variables in the equation in Table B-4-20 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating K_v. The degree of uncertainty associated with these variables is as follows: a) We assume the uncertainty associated with six variables—D_w, u, d_z, ρ_a, ρ_w, and μ_w—is minimal or insignificant, either because of narrow recommended ranges for these variables or because information to estimate variable values is generally available. b) No original sources were identified for the equations used to derive recommended values or specific recommended values for variables C_d, k, and λ_z. Therefore, the degree and direction of any uncertainties associated with these variables are unknown. c) Uncertainties associated with the variable W are site-specific.
K_G	Gas phase transfer coefficient	m/yr	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-21. Uncertainties associated with this variable include the following: All of the variables in the equation in Table B-4-21, with the exception of k, are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating K_G. The degree of uncertainty associated with each of these variables is as follows: a) We assume the uncertainty associated with the variables D_a, μ_a, and ρ_a, is minimal or insignificant, because these variables have been extensively studied, and equation procedures are well-established. b) No original sources were identified for equations used to derive recommended values or specific recommended values for variables C_d, k, and d_z. Therefore, the degree and direction of any uncertainties are unknown. c) Uncertainties associated with the variable W are site-specific and cannot be readily estimated.
H	Henry's Law constant	atm-m^3/mol	**Varies** This variable is COPC-specific. We discuss this variable in detail in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Values for this variable, estimated using the parameters and algorithms in Appendix A-2, may under- or overestimate the actual COPC-specific values. As a result, K_v may be under- or overestimated to a limited degree.

TABLE B-4-19

OVERALL COPC TRANSFER RATE COEFFICIENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

Variable	Description	Units	Value
R	Universal gas constant	atm-m^3/mol-K	**8.205×10^{-5}** There are no uncertainties associated with this constant.
T_{wk}	Water body temperature	K	**298** This variable is site-specific. We recommend using this default value when site-specific information is not available; this is consistent with U.S. EPA (1994) and U.S. EPA (1998). The following uncertainty is associated with this variable: To the extent that the default water body temperature value does not accurately represent site- and water body-specific conditions, K_v, will be under- or overestimated.
θ	Temperature correction factor	unitless	**1.026** This variable is site-specific. We recommend using this default value when site-specific information is not available. This is consistent with U.S. EPA (1994) and U.S. EPA (1998). The following uncertainty is associated with this variable: The purpose and sources of this variable and the recommended value are unknown.

REFERENCES AND DISCUSSION

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is cited as the reference source for water body temperature (T_{wk}) and temperature correction factor (θ). This document apparently cites U.S. EPA (1993a) as its source of information.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is the reference source for the equation in Table B-4-19. This document also recommends the T_{wk} value of 298 K (298 K = 25°C) and the θ value of 1.026. No source was identified for these values.

TABLE B-4-20

LIQUID PHASE TRANSFER COEFFICIENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 5)

Description

This equation calculates the rate of COPC transfer from the liquid phase for a flowing or quiescent water body.

Uncertainties associated with this equation include the following:

(1) We assume uncertainly associated with the following six variables is minimal or insignificant: D_w, u, d_z, ρ_a, ρ_w, and μ_w.
(2) No original sources were identified for equations used to derive recommended values or specific recommended values for the following three variables: C_d, k, and d_z. Therefore, the degree and duration of any uncertainties associated with these variables is unknown.
(3) Uncertainties associated with the variable W are site-specific.

Equation

For flowing streams or rivers

$$K_L = \sqrt{\frac{(1 \times 10^{-4}) \cdot D_w \cdot u}{d_z}} \cdot 3.1536 \times 10^7$$

For quiescent lakes or ponds

$$K_L = (C_d^{0.5} \cdot W) \cdot (\frac{\rho_a}{\rho_w})^{0.5} \cdot \frac{k^{0.33}}{\lambda_z} \cdot (\frac{\mu_w}{\rho_w \cdot D_w})^{-0.67} \cdot 3.1536 \times 10^7$$

Variable	Description	Units	Value
K_L	Liquid phase transfer coefficient	m/yr	

TABLE B-4-20

LIQUID PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 5)

Variable	Description	Units	Value
D_w	Diffusivity of COPC in water	cm^2/s	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The default D_w values may not accurately represent the behavior of COPCs under water body-specific conditions. However, we expect the degree of uncertainty to be minimal.
u	Current velocity	m/s	**Varies** This variable is site-specific, and relates to the volumetric flow rate of the waterbody evaluated. The following uncertainty is associated with this variable: Sources of values for this variable are reasonably available for most large surface water bodies. Estimated values for this variable may be necessary for smaller water bodies; uncertainty will be associated with these estimates. We don't expect the degree of uncertainty associated with this variable to be significant.
d_z	Total water body depth	m	**Varies** This variable is site-specific, and, in most cases, represents the average mean across the waterbody evaluated. We recommend that you calculate this value using the following equation, consistent with U.S. EPA (1994) and NC DEHNR (1997): $d_z = d_{wc} + d_{bs}$ No reference was cited for this recommendation. The following uncertainty is associated with this variable: Calculating this variable sums the concentrations associated with the two variables d_{wc} and d_{bs}. Because most of the total water body depth (d_z) is made up of the depth of the water column (d_{wc}), and the uncertainties associated with d_{wc} are not expected to be significant, we likewise don't expect the total uncertainties associated with d_z to be significant.
3.1536×10^7	Units conversion factor	s/yr	

TABLE B-4-20

LIQUID PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 5)

Variable	Description	Units	Value
C_d	Drag coefficient	unitless	**0.0011** This variable is site-specific. We recommend a default value of 0.0011, consistent with U.S. EPA (1994), NC DEHNR (1997), and U.S. EPA (1998). The following uncertainty is associated with this variable: The original source of this variable value is unknown. Therefore, any uncertainties associated with its use are also unknown.
W	Average annual wind speed	m/s	**3.9** Consistent with U.S. EPA (1998), we recommend a default value of 3.9 m/s. See Chapter 3 for guidance regarding the references and methods used to determine a site-specific value that is consistent with air dispersion modeling. The following uncertainty is associated with this variable: To the extent that site-specific or local values for this variable are not available, default values may not accurately represent site-specific conditions. The uncertainty associated with the selection of a single value from within the range of windspeeds at a single location may be more significant than the uncertainty associated with choosing a single windspeed to represent all locations.
ρ_a	Density of air	g/cm^3	**0.0012** We recommend this default value when site-specific information is not available. This is consistent with U.S. EPA (1994) and NC DEHNR (1997), both of which cite Weast (1979) as the source of this value. This value applies at standard conditions (25°C or 298 K and 1 atm or 750 mm Hg). The density of air will vary with temperature.
ρ_w	Density of water	g/cm^3	**1** We recommend this default value, consistent with U.S. EPA (1994) and NC DEHNR (1997), both of which cite Weast (1979) as the source of this value. This value applies at standard conditions (25°C or 298 K and 1 atm or 750 mm Hg). There is no significant uncertainty associated with this variable.
k	von Karman's constant	unitless	**0.4** This value is a constant. We recommend using this value, consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The original source of this variable value is unknown. Therefore, any uncertainties associated with its use are also unknown.

TABLE B-4-20

LIQUID PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 5)

Variable	Description	Units	Value
λ_z	Dimensionless viscous sublayer thickness	unitless	**4** This value is site-specific. We recommend using this default value when site-specific information is not available; consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The source of the value for this variable is unknown. Therefore, any uncertainties associated with its use cannot be quantified.
μ_w	Viscosity of water corresponding to water temperature	g/cm-s	**1.69 x 10^{-02}** We recommend this default value, consistent with U.S. EPA (1994) and NC DEHNR (1997), which both cite Weast (1979) as the source of this value. This value applies at standard conditions (25°C or 298 K and 1 atm or 760 mm Hg). There is no significant uncertainty associated with this variable.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the range of D_w values and assumed C_d, ρ_a, ρ_w, k, α_λ, and μ_w values of 0.0011, 1.2 x 10^{-3}, 1, 0.4, 4, and 1.69 x 10^{-2}, respectively. This document cites (1) Weast (1979) as its source of information regarding ρ_a, ρ_w, and μ_w; and (2) U.S. EPA (1993a) as its source of information regarding $C_{d,}$ k, and d_z.

U.S. EPA. 1993a. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions*. Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September 24.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of the recommended drag coefficient (C_d) value of 0.0011 and the recommended von Karman's constant (k) value of 0.4. The original sources of variable values are not identified.

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is cited as one of the sources of the range of D_w values and assumed C_d, ρ_a, ρ_w, k, λ_z, and μ_w values of 0.0011, 1.2 x 10^{-3}, 1, 0.4, 4, and 1.69 x 10^{-2}, respectively. This document cites (1) Weast (1979) as its source of information regarding ρ_a, ρ_w, and μ_w; and (2) U.S. EPA (1993a) as its source of information regarding $C_{d,}$ k, and d_z.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-20

LIQUID PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 5 of 5)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends a value of 0.0011 for the drag coefficient (C_d) variable and a value of 0.4 for von Karman's constant (k). No sources are cited for these values.

Weast, R. C. 1979. *CRC Handbook of Chemistry and Physics*. 60th ed. CRC Press, Inc. Cleveland, Ohio.

This document is cited as the source of ρ_a, ρ_w, and μ_w variables of 1.2 x 10^{-3}, 1, and 1.69 x 10^{-2}, respectively.

TABLE B-4-21

GAS PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the rate of COPC transfer from the gas phase for a flowing or quiescent water body. Uncertainties associated with this equation include the following:

(1) Minimal or insignificant uncertainty is assumed to be associated with the variables D_a, μ_a, and ρ_a.

(2) No original sources were identified for equations used to derive recommended values or specific recommended values for variables C_d, k, and λ_z. Therefore, the degree and direction of any uncertainties associated with these variables are unknown.

(3) Uncertainties associated with the remaining variables are site-specific.

Equation

Flowing streams or rivers

$$K_G = 36500 \ m/yr$$

Quiescent lakes or ponds

$$K_G = (C_d^{0.5} \cdot W) \cdot \frac{k^{0.33}}{\lambda_z} \cdot \left(\frac{\mu_a}{\rho_a \cdot D_a}\right)^{-0.67} \cdot 3.1536\times10^7$$

Variable	Description	Units	Value
K_G	Gas phase transfer coefficient	m/yr	
C_d	Drag coefficient	unitless	**0.0011** This variable is site-specific. We recommend using this default value when site-specific information is not available, consistent with U.S. EPA (1994), NC DEHNR (1997), and U.S. EPA (1998). The following uncertainty is associated with this variable: The original source of this variable is unknown. Therefore, any uncertainties associated with its use are also unknown.

TABLE B-4-21

GAS PHASE TRANSFER COEFFICIENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
W	Average annual wind speed	m/s	**3.9** Consistent with U.S. EPA (1998), we recommend a default value of 3.9 m/s. See Chapter 3 of the HHRAP for guidance regarding the references and methods used to determine a site-specific value that is consistent with air dispersion modeling. The following uncertainty is associated with this variable: To the extent that site-specific or local values for this variable are not available, default values may not accurately represent site-specific conditions. The uncertainty associated with the selection of a single value from within the range of windspeeds at a single location may be more significant than the uncertainty associated with choosing a single windspeed to represent all locations.
k	von Karman's constant	unitless	**0.4** This value is a constant. We recommend using this value, consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The original source of this variable is unknown. Therefore, any uncertainties associated with its use are also unknown.
λ_z	Dimensionless viscous sublayer thickness	unitless	**4** This value is site-specific. We recommend using this default value when site-specific information is not available, consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The original source of this variable is unknown. Therefore, any uncertainties associated with its use are also unknown.
μ_a	Viscosity of air	g/cm-s	**1.81×10^{-04}** We recommend using this default value when site-specific information is not available, consistent with U.S. EPA (1994) and NC DEHNR (1997), both of which cite Weast (1979) as the source of their information. There is no significant uncertainty associated with this variable.
ρ_a	Density of air	g/cm^3	**0.0012** We recommend using this default value when site-specific information is not available, consistent with U.S. EPA (1994) and NC DEHNR (1997), both of which cite Weast (1979) as the source of this value. This value applies at standard conditions (25°C or 298 K and 1 atm or 760 mm Hg) . The density of air will vary with temperature.

TABLE B-4-21

GAS PHASE TRANSFER COEFFICIENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
D_a	Diffusivity of COPC in air	cm^2/s	**Varies** This variable is COPC-specific. We discuss this variable in detail in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The recommended D_a values may not accurately represent the behavior of COPCs under water body-specific conditions. However, we expect the degree of uncertainty to be minimal.
3.1536×10^7	Units conversion factor	s/yr	

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the variables ρ_a, k, λ_z , and μ_a values of 1.2×10^{-3}, 0.4, 4, and 1.81×10^{-04}, respectively. This document cites (1) Weast (1979) as its source of information for ρ_a and μ_a, and (2) U.S. EPA (1993a) as its source of information for k and λ_z.

U.S. EPA. 1993a. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustion Emissions*. Working Group Recommendations. Office of Solid Waste, and Office of Research and Development. Washington, D.C. September.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of (1) the recommended drag coefficient (C_d) value of 0.0011, (2) the recommended von Karman's constant (k) value of 0.4, and (3) the recommended dimensionless viscous sublayer thickness (λ_z) value of 4. The original sources of these variable values are not identified.

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document is cited as one of the sources of the variables ρ_a, k, λ_z , and μ_a values of 1.2×10^{-3}, 0.4, 4, and 1.81×10^{-04}, respectively. This document cites (1) Weast (1979) as its source of information for ρ_a and μ_a, and (2) U.S. EPA (1993a) as its source of information for k and λ_z.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment*. Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-4-21

GAS PHASE TRANSFER COEFFICIENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document recommends (1) a value of 0.0011 for the drag coefficient (C_d) variable, (2) a value of 0.4 for von Karman's constant (K), and (3) a value of 4 for the dimensionless viscous sublayer thickness (λ_z) variable. The original sources of the variable values are not identified.

Weast, R.C. 1979. *CRC Handbook of Chemistry and Physics.* 60th ed. CRC Pres, Inc. Cleveland, Ohio.

This document is cited as the source of ρ_a, ρ_w, and μ_a variables of 1.2 x 10^{-3}, 1, and 1.69 x 10^{-2}, respectively.

TABLE B-4-22

BENTHIC BURIAL RATE CONSTANT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the water column loss constant due to burial in benthic sediment.

Uncertainties associated with this equation include the following:

(1) All of the variables in the equation in Table B-4-22 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating k_b. The degree of uncertainty associated with each of these variables is as follows: (a) X_e—about one order of magnitude at the most, (b) C_{BS}, d_{bs}, V_{fx}, TSS, and A_w—limited because of the narrow recommended ranges for these variables or because resources to estimate variable values are generally available, (c) A_L and SD—very site-specific, degree of uncertainty unknown.

Based on the possible ranges for the input variables to this equation, values of k_b can range over about one order of magnitude.

Equation

$$k_b = \left(\frac{X_e \cdot A_L \cdot SD \cdot 1 \times 10^3 - Vf_x \cdot TSS}{A_w \cdot TSS} \right) \cdot \left(\frac{TSS \cdot 1 \times 10^{-6}}{C_{BS} \cdot d_{bs}} \right)$$

Variable	Description	Units	Value
k_b	Benthic burial rate constant	yr^{-1}	
X_e	Unit soil loss	kg/m^2-yr	**Varies** This variable is site-specific and is calculated using the equation in Table B-4-13. The following uncertainty is associated with this variable: All of the variables in the equation used to calculate unit soil loss, X_e, are site-specific. Using default values rather than site-specific values, for any or all of the equation variables, will result in estimates of X_e that under- or overestimate the actual value. We expect the degree or magnitude of any under- or overestimation to be about one order of magnitude or less.
A_L	Total watershed area receiving deposition	m^2	**Varies** This variable is site-specific (see Chapter 4). Uncertainties associated with this variable are site-specific.

TABLE B-4-22

BENTHIC BURIAL RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
SD	Watershed sediment delivery ratio	unitless	**Varies** This value is site-specific and is calculated using the equation in Table B-4-14. Uncertainties associated with this variable include the following: (1) The default values for empirical intercept coefficient (*a*) that we recommend for use in the equation in Table B-4-14, are average values based on various studies of sediment yields from various watersheds. Therefore, these default values may not accurately represent site-specific watershed conditions. As a result, using these default values may contribute to under- or overestimating the benthic burial rate constant, k_b. (2) The default value for empirical slope coefficient (*b*) that we recommend for use in the equation in Table B-4-14 is based on a review of sediment yields from various watersheds. This single default value may not accurately represent site-specific watershed conditions. As a result, using this default value may contribute to under-or overestimating k_b.
1×10^3	Units conversion factor	g/kg	
Vf_x	Average volumetric flow rate through water body	m^3/yr	**Varies** This variable is site-specific. We recommend using site- and waterbody-specific measured values, representative of long-term average annual values for the water body of concern. The following uncertainty is associated with this variable: Default average volumetric flow rate (Vf_x) values may not accurately represent site-specific water body conditions. Therefore, using such default values may contribute to under- or overestimating k_b. However, we expect that the uncertainty associated with this variable to be limited, because resources such as maps, aerial photographs, and gauging station measurements—from which Vf_x can be estimated—are generally available.
TSS	Total suspended solids concentration	mg/L	**2 to 300** This variable is site-specific. We recommend using site- and waterbody specific measured values, representative of long-term average annual values for the water body of concern (see Chapter 5). A value of 10 mg/L was cited by NC DEHNR (1997), and U.S. EPA (1993) in the absence of site-specific measured data. The following uncertainty is associated with this variable: Limitation on measured data used for determining a water body specific total suspended solids (*TSS*) value may not accurately reflect site- and water body-specific conditions long term. Therefore, the *TSS* value may contribute to under-or overestimating f_{wc}.

TABLE B-4-22

BENTHIC BURIAL RATE CONSTANT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
A_w	Water body surface area	m^2	**Varies** This variable is site-specific. The value selected represents an average value for the entire year. See Chapter 4 for guidance regarding the references and methods used to determine this value. Uncertainties associated with this variable are site-specific. However, we expect that the uncertainty associated with this variable will be limited, because maps, aerial photographs—and other resources from which water body surface area, A_w, can be measured—are readily available.
1×10^{-6}	Units conversion factor	kg/mg	
C_{BS}	Bed sediment concentration	g/cm^3	**1.0** This variable is site-specific. We recommend a default value of 1.0, consistent with U.S. EPA (1998), which states that this value should be reasonable for most applications. No reference is cited for this recommendation. The recommended default value is also consistent with U.S. EPA (1994), and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended value may not accurately represent site-specific water body conditions.
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. The value selected represents an average value for the entire year. We recommend a default upper-benthic sediment depth of 0.03 meters, which is based on the center of the range cited by U.S. EPA (1993). This value is also consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended default value for depth of upper benthic sediment layer, d_{bs}, may not accurately represent site-specific water body conditions. Therefore, use of this default value may contribute to the under- or overestimation of k_b. However, the degree of uncertainty associated with this variable is expected to be limited because of the narrow recommended range.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the range of all recommended specific C_{BS} and d_{bs} values. This document cites U.S. EPA (1993a) as its source.

TABLE B-4-22

BENTHIC BURIAL RATE CONSTANT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustion Emissions*. Working Group Recommendations. Office of Solid Waste, and Office of Research and Development. Washington, D.C. September.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of (1) the recommended drag coefficient (C_d) value of 0.0011, (2) the recommended von Karman's constant (k) value of 0.4, and (3) the recommended dimensionless viscous sublayer thickness (λ_z) value of 4. The original sources of these variable values are not identified.

U.S. EPA 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 15.

This document is cited as one of the reference sources for the d_{bs} value. The recommended value is the midpoint of an acceptable range. This document is also cited as one of the reference source documents for the default C_{BS} value. This document cites U.S. EPA (1993a) as its source.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

TABLE B-4-23

TOTAL WATER COLUMN CONCENTRATION
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the total water column concentration of COPCs including (1) both dissolved COPCs and (2) COPCs sorbed to suspended solids. Uncertainties associated with this equation include the following:

(1) All of the variables in the equation in Table B-4-23 are COPC- and site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating C_{wctot}.

We expect the degree of uncertainty associated with the variables d_{wc} and d_{bs} to be minimal either because information for estimating a variable (d_{wc}) is generally available or because the probable range for a variable (d_{bs}) is narrow. The uncertainty associated with the variables f_{wc} and C_{wtot} is associated with estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, the uncertainty associated with using default *OC* values may be significant in specific cases.

Equation

$$C_{wctot} = f_{wc} \cdot C_{wtot} \cdot \frac{d_{wc} + d_{bs}}{d_{wc}}$$

Variable	Description	Units	Value
C_{wctot}	Total COPC concentration in water column	mg COPC/L water column	
f_{wc}	Fraction of total water body COPC concentration in the water column	unitless	**0 to 1** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-16. The following uncertainty is associated with this variable: The default variable values we recommend you use in the equation in Table B-4-16 may not accurately represent site-specific water body conditions. However, the ranges of several variables—including d_{bs}, C_{BS}, and θ_{sw}—are relatively narrow. Therefore, we expect the uncertainty to be relatively small. You can reasonably estimate other variables, such as d_{wc} and d_z, using generally available information. The largest degree of uncertainty may be introduced by the default medium-specific *OC* content values. *OC* content values are often not readily available and can vary widely for different locations in the same medium. Therefore, default values may not adequately represent site-specific conditions.

TABLE B-4-23

TOTAL WATER COLUMN CONCENTRATION
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
C_{wtot}	Total waterbody COPC concentration including water column and bed sediment	mg COPC/L water body (or g COPC/m^3 water body)	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-15. The following uncertainty is associated with this variable: The default variable values we recommend you use in the equation in Table B-4-15 may not accurately represent site-specific water body conditions. We expect the degree of uncertainty associated with variables Vf_x, A_w, d_w, and d_{bs} to be limited either because the probable ranges for variables are narrow or information allowing accurate estimates is generally available. Uncertainty associated with f_{wc} is largely the result of water body-associated default *OC* content values, and may be significant in specific instances. Uncertainties associated with the total COPC load into water body (L_t) and overall total water body COPC dissipation rate constant (k_{wt}) may also be significant in some instances because of the combination of many variable-specific uncertainties.
d_{wc}	Depth of water column	m	**Varies** This variable is site-specific. The following uncertainty is associated with this variable: Default values for depth of water column, d_{wc}, may not accurately reflect site-specific water body conditions. Therefore, using default values may contribute to under- or overestimating C_{wctot}. However, we expect the degree of uncertainty associated with this variable to be limited, because information regarding this variable is generally available.
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. We recommend a default upper-benthic sediment depth of 0.03 meters, which is based on the center of a range cited by U.S. EPA (1993) This value is consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended default value for depth of upper benthic sediment layer, d_{bs}, may not accurately represent site-specific water body conditions. Therefore, using this default value may contribute to under- or overestimating C_{wctot}. However, we expect the degree of uncertainty associated with this variable to be limited because of the narrow recommended range.

TABLE B-4-23

TOTAL WATER COLUMN CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the range of d_{bs} values. This document cites U.S. EPA (1993a) as its source.

U.S. EPA. 1993. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as one of the sources of the ranges of d_{bs} values. No original source of this range was identified.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facility*. April.

This document is cited as one of the reference sources for the default value for depth of upper benthic layer (d_{bs}). The recommended value is the midpoint of an acceptable range. This document cites U.S. EPA (1993) as the source of its information. The degree of uncertainty associated with the variables d_{wc} and d_{bs} is expected to be minimal either because information for estimating these variables is generally available (d_{wc}) or the probable range for a variable (d_{bs}) is narrow. Uncertainty associated with the variables f_{wc} and C_{wtot} is largely associated with the use of default *OC* content values. Because *OC* content is known to vary widely in different locations in the same medium, use of default medium-specific values can result in significant uncertainty in some instances.

TABLE B-4-24

DISSOLVED PHASE WATER CONCENTRATION
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates the concentration of COPC dissolved in the water column. Uncertainties associated with this equation include the following:

(1) The variables in the equation in Table B-4-24 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating C_{dw}. We expect the degree of uncertainty associated with *TSS* to be relatively small, because information regarding reasonable site-specific values for this variable are generally available or it can be easily measured. On the other hand, the uncertainty associated with the variables C_{wctot} and Kd_{sw} is associated with estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, using default *OC* values may result in significant uncertainty in specific cases.

Equation

$$C_{dw} = \frac{C_{wctot}}{1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6}}$$

For mercury modeling,

$$C_{dw(Mercury)} = \frac{C_{wctot(Hg^{2+})}}{1 + Kd_{sw(Hg^{2+})} \cdot TSS \cdot 1 \times 10^{-6}}$$

Use the equation above to calculate the C_{dw} mercury value. Apportion into the divalent mercury (Hg^{2+}) and methyl mercury (MHg) forms based on the assumed 85% Hg^{2+} and 15% MHg speciation split in the water body (see Chapter 2) using the correlations below. Elemental mercury (Hg^0) occurs in very small amounts in the vapor phase and does not exist in the particle or particle-bound phase. Therefore, assume elemental mercury in the water body is negligible or zero, and evaluate it for the direct inhalation pathway only (Table B-5-1).

$C_{dw\ (Hg2+)}$ = $0.85\ C_{\mathrm{dw\ (Mercury)}}$
$C_{dw\ (Mhg)}$ = $0.15\ C_{\mathrm{dw\ (Mercury)}}$
$C_{dw\ (Hg0)}$ = 0.0

Evaluate divalent and methyl mercury as individual COPCs to determine C_{fish} (Tables B-4-26 and B-4-27) for calculating COPC intake from fish in Table C-1-4, and in evaluating COPC intake from drinking water (Table C-1-5). Calculate $C_{dw\ (Mercury)}$ as above using the corresponding fate and transport parameters for mercuric chloride (Hg^{2+}) provided in Appendix A-2, and determine $C_{dw\ (Hg2+)}$ and $C_{dw\ (Mhg)}$ as calculated above.

TABLE B-4-24

DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
C_{dw}	Dissolved phase water concentration	mg COPC/L water	
C_{wctot}	Total COPC concentration in water column	mg COPC/L water column	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-23. The following uncertainty is associated with this variable: All of the variables in the equation in Table B-4-23 are COPC- and site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating C_{wctot}. We expect the degree of uncertainty associated with the variables d_{wc} and d_{bs} to be minimal either because information for estimating a variable (d_{wc}) is generally available or because the probable range for a variable (d_{bs}) is narrow. The uncertainty associated with the variables f_{wc} and C_{wtot} is associated with estimates of Organic Carbon, *OC*, content. Because *OC* content values can vary widely for different locations in the same medium, using default *OC* values may result in significant uncertainty in specific cases.
Kd_{sw}	Suspended sediments/surface water partition coefficient	L water/kg suspended sediment	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: Values contained in Appendix A-2 for Kd_{sw} are based on default *OC* content values for surface water and soil. Because *OC* content can vary widely for different locations in the same medium, the uncertainty associated with estimated Kd_{sw} values based on default *OC* content values may be significant in specific cases.
TSS	Total suspended solids concentration	mg/L	**2 to 300** This variable is site-specific. We recommend using site- and waterbody specific measured values, representative of long-term average annual values for the water body of concern (see Chapter 5). A value of 10 mg/L was cited by NC DEHNR (1997) and U.S. EPA (1993b) in the absence of site-specific measured data. The following uncertainty is associated with this variable: Limitation on measured data used for determining a water body specific total suspended solids (*TSS*) value may not accurately reflect site- and water body-specific conditions long term. Therefore, the *TSS* value may contribute to under-or overestimating f_{wc}.
1×10^{-6}	Units conversion factor	kg/mg	

TABLE B-4-24

DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 3)

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the range of Kd_s values and the *TSS* value of 10. This document cites U.S. EPA (1993) as its sources of information regarding *TSS*, and (2) RTI (1992) as its source regarding Kd_s.

U.S. EPA. 1993. *Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document is cited by U.S. EPA (1994) and NC DEHNR (1997) as one of the sources of the range of Kd_s value and the assumed *OC* value of 0.075 for surface water. The generic equation for calculating partition coefficients (soil, surface water, and bed sediments) is as follows: $Kd_{ij} = K_{ocj} * OC_i$. K_{oc} is a chemical-specific value; however, *OC* is medium-specific. The range of Kd_s values was based on an assumed *OC* value of 0.01 for soil. Therefore, the Kd_{sw} values were estimated by multiplying the Kd_s values by 7.5, because the *OC* value for surface water is 7.5 times greater than the *OC* value for soil. This document is also cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of the recommended *TSS* value.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April 15.

This document is cited as one of the sources of the range of Kd_s values, citing RTI (1992) as its source of information.

TABLE B-4-25

COPC CONCENTRATION SORBED TO BED SEDIMENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 1 of 4)

Description

This equation calculates the concentration of COPCs sorbed to bed sediments.

Uncertainties associated with this equation include the following:

(1) The default variable values recommended for use in the equation in Table B-4-25 may not accurately represent site-specific water body conditions. We expect the degree of uncertainty associated with variables θ_{bs}, C_{BS}, d_{wc}, and d_{bs} to be limited either because the probable ranges for these variables are narrow or because information allowing reasonable estimates is generally available.

(2) Uncertainties associated with variables f_{bs}, C_{wtot} and Kd_{bs} are largely associated with the use of default *OC* content values in their calculation. The uncertainty may be significant in specific instances, because *OC* content is known to vary widely in different locations in the same medium.

Equation

$$C_{sb} = f_{bs} \cdot C_{wtot} \cdot \frac{Kd_{bs}}{\theta_{bs} + Kd_{bs} \cdot C_{BS}} \cdot \frac{d_{wc} + d_{bs}}{d_{bs}}$$

Variable	Description	Units	Value
C_{sb}	Concentration sorbed to bed sediment	mg COPC/kg sediment	
f_{bs}	Fraction of total water body COPC concentration that occurs in the benthic sediment	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-16. The following uncertainty is associated with this variable: The default values for the variables in the equation in Table B-4-16 may not accurately represent site- and water body-specific conditions. However, the range of several variables—including d_{bs}, C_{BS}, and θ_{bs}—is relatively narrow. You can reasonably estimate other variables, such as d_{wc} and d_z, using generally available information. The largest degree of uncertainty may be introduced by the default medium-specific *OC* content values. Because *OC* content values may vary widely in different locations in the same medium, using default values may result in significant uncertainty in specific cases.

TABLE B-4-25

COPC CONCENTRATION SORBED TO BED SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 2 of 4)

Variable	Description	Units	Value
C_{wtot}	Total water body concentration including water column and bed sediment	mg COPC/L water body (or g COPC/cm^3 water body)	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-15. The following uncertainty is associated with this variable: (1) The default variable values may not accurately represent site-specific water body conditions. We expect the degree of uncertainty associated with variables Vf_x, A_w, d_{wc}, and d_{bs} to be limited either because the probable ranges for these variables are narrow or information allowing reasonable estimates is generally available. (2) Uncertainty associated with f_{wc} is largely the result of uncertainty associated with default *OC* content values and may be significant in specific instances. Uncertainties associated with the variable L_T and K_{wt} may also be significant because of the combination of many variable-specific uncertainties.
Kd_{bs}	Bed sediment/sediment pore water partition coefficient	L water/kg bed sediment (or cm^3 water/g bed sediment)	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. The following uncertainty is associated with this variable: The default Kd_{bs} values in Appendix A-2 are based on default *OC* content values for sediment and soil. Because medium-specific *OC* content may vary widely at different locations in the same medium, the uncertainty associated with Kd_{bs} values calculated by using default *OC* content values may be significant in specific instances.
θ_{bs}	Bed sediment porosity	unitless ($L_{pore\ volume}/L_{sediment}$)	**0.6** This variable is site-specific. We recommend a default bed sediment porosity of 0.6 (using a C_{BS} value of 1 g/cm^3 and a solids density (ρ_s) value of 2.65 kg/L), calculated using the following equation (U.S. EPA 1998): $\theta_{bs} = 1 - C_{BS}/\rho_s$ This also is consistent with U.S. EPA (1993), U.S. EPA (1994), and NC DEHNR (1997). The following uncertainty is associated with this variable: To the extent that the recommended default values of C_{BS} and ρ_s don't accurately represent site- and water body-specific conditions, θ_{bs} will be under- or overestimated to some degree. However, we expect the degree of uncertainty to be minimal, based on the narrow range of recommended values.

TABLE B-4-25

COPC CONCENTRATION SORBED TO BED SEDIMENT (CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 3 of 4)

Variable	Description	Units	Value
C_{BS}	Bed sediment concentration (or bed sediment bulk density)	g/cm^3	**1.0** This variable is site-specific. We recommend a default value of 1.0, consistent with U.S. EPA (1998), which states that this value should be reasonable for most applications. No reference is cited for this recommendation. This is also consistent with U.S. EPA (1993), U.S. EPA (1994), and NC DEHNR (1997). The following uncertainty is associated with this variable: The recommended default value for θ_{bs} may not accurately represent site- and water body-specific conditions. Therefore, the variable C_{sb} may be under- or overestimated to a limited degree, as indicated by the narrow range of recommended values.
d_{wc}	Depth of water column	m	**Varies** This variable is site-specific. The following uncertainty is associated with this variable: Default d_{wc} values may not accurately reflect site-specific conditions. Therefore, using these values may contribute to under- or overestimating the variable C_{sb}. However, we expect the degree of uncertainty to be minimal, because resources allowing reasonable water body-specific estimates of d_{wc} are generally available.
d_{bs}	Depth of upper benthic sediment layer	m	**0.03** This variable is site-specific. We recommend a default upper-benthic sediment depth of 0.03 meters, which is based on the center of a range cited by U.S. EPA (1998). This value is consistent with U.S. EPA (1994) and NC DEHNR (1997). The following uncertainty is associated with this variable: Default d_{bs} values may not accurately reflect site-specific conditions. Therefore, using these values may contribute to under- or overestimating C_{sb}. However, we expect the degree of uncertainty to be small, based on the narrow recommended range of default values.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the sources of the range of Kd_s values and an assumed *OC* value of 0.04 for sediment. This document cites RTI (1992) as its source of information regarding Kd_s values. This document is also cited as one of the reference source documents for the default value for bed sediment porosity(θ_{sw}). This document cites U.S. EPA (1993a; 1993b) as its source of information. This document is also cited as one of the reference source documents for the default value for depth of the upper benthic layer. The default value is the

TABLE B-4-25

COPC CONCENTRATION SORBED TO BED SEDIMENT
(CONSUMPTION OF DRINKING WATER AND FISH EQUATIONS)

(Page 4 of 4)

midpoint of an acceptable range. This document cites U.S. EPA (1993a) and U.S. EPA (1993b) as its source of information for the range of values for the depth of the upper benthic layer. This document is also cited as one of the reference source documents for the default bed sediment concentration (C_{BS}). This document cites U.S. EPA (1993a; 1993b) as its source.

U.S. EPA. 1993b. *Addendum: Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Working Group Recommendations. Office of Solid Waste and Office of Research and Development. Washington, D.C. September.

This document is cited by NC DEHNR (1997) and U.S. EPA (1994) as the source of the default bed sediment porosity value (θ_{sw}), the default bed sediment concentration value (C_{BS}), and the range for depth of upper benthic layer (d_{bs}) values.

U.S. EPA. 1994. *Draft Guidance for Performing Screening Level Risk Analysis at Combustor Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April.

This document is cited as one of the sources of the range of Kd_s values and an assumed OC value of 0.04 for sediment. This document cites RTI (1992) as its source of information regarding Kd_s values. This document is cited as one of the reference source documents for the default value for bed sediment porosity (θ_{sw}). This document cites U.S. EPA (1993a; 1993b) as its source. This document is also cited as one of the reference source documents for the default value for depth of upper benthic layer (d_{bs}). The default value is the midpoint of an acceptable range. This document cites U.S. EPA (1993a) and U.S. EPA (1993b) as its source of information for the range of values for the depth of the upper benthic layer. This document is also cited as one of the reference source documents for the default bed sediment concentration (C_{BS}). This document cites U.S. EPA (1993b) as its source.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

This document is also cited as the source of the equation for calculating bed sediment porosity (θ_{sw}). No source of this equation was identified. This document was also cited as the source for the range of the bed sediment concentration (C_{BS}). No source of this range was identified.

TABLE B-4-26

FISH CONCENTRATION FROM BIOCONCENTRATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates fish concentration, from dissolved COPCs, by using a bioconcentration factor. Uncertainty associated with this equation include the following:

Calculating C_{dw} is dependent on default values for two variables C_{wctot} and Kd_{sw}. Values for these two variables are, in turn, dependent on default medium-specific *OC* content values. Because *OC* content can vary widely at different locations in the same medium, significant uncertainty may be associated with C_{wctot} and Kd_{sw} and, in turn, C_{dw} in specific instances.

Equation

$$C_{fish} = C_{dw} \cdot BCF_{fish}$$

Variable	Description	Units	Value
C_{fish}	Concentration of COPC in fish	mg COPC/kg FW tissue	
C_{dw}	Dissolved phase water concentration	mg COPC/L	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-24. Uncertainties associated with this variable include the following: (1) The variables in the equation in Table B-4-24 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to the under- or overestimating C_{dw}. We expect the degree of uncertainty associated with *TSS* to be relatively small, because information regarding reasonable site-specific values for this variable is generally available or can be easily measured. (2) The uncertainty associated with the variables C_{wctot} and Kd_{sw} is dependent on estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, the uncertainty associated with using different *OC* content values may be significant in specific cases.

TABLE B-4-26

FISH CONCENTRATION FROM BIOCONCENTRATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
BCF_{fish}	Bioconcentration factor for COPC in fish	unitless ([mg COPC/kg FW tissue]/[mg COPC/kg feed])	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. As explained in Appendix A-2, we recommend using *BCFs* for organic COPCs with log K_{ow} less than 4.0 and *BAFs* (rather than *BCFs*) for organic COPCs with log K_{ow} of 4.0 or greater. For organics with a log K_{ow} value of less than 4.0 and all metals (except lead and mercury), we obtained values from U.S. EPA (1998) or, when measured values were not available, derived from the correlation equation presented by Lyman et al. (1982). The following uncertainty is associated with this variable: The COPC-specific *BCF* values may not accurately represent site-specific water body conditions, because estimates of *BCFs* and *BAFs* can vary, based on experimental conditions.

REFERENCES AND DISCUSSION

Ellgenhausen, H. J., A. Guth, and H.O. Esser. 1980. "Factors Determining the Bioaccumulation Potential of Pesticides in the Individual Compartments of Aquatic Food Chains." *Ecotoxicology Environmental Safety*. Vol. 4. P. 134.

*BCF*s for pesticides and polycyclic aromatic hydrocarbons (PAHs) with log K_{ow} less than 5.5 were apparently calculated by using the following equation derived for pesticides from this document:

$$\log BCF = 0.83 \cdot \log K_{ow} - 1.71$$

where

BCF = Bioconcentration factor for COPC in fish(unitless)
K_{ow} = Octanol-water partition coefficient (unitless)

Lyman, W.J., W.F. Reehl, and D.H. Rosenblatt. 1982. *Handbook of Chemical Property Estimation Methods: Environmental Behavior of Organic Compounds.* McGraw-Hill Book Company. New York, New York.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document cites Ogata et al. (1984), U.S. EPA (1994, 1995) as its sources of the equations used to calculate *BCF*s fish:

TABLE B-4-26

FISH CONCENTRATION FROM BIOCONCENTRATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 3 of 3)

Ogata, M.K., Y. Ogino Fijusaw, and E. Mano. 1984. "Partition Coefficients as a Measure of Bioconcentration Potential of Crude Oil Compounds in Fish and Shellfish." *Bulletin of Environmental Contaminant Toxicology*. Vol. 33. P. 561.

*BCF*s for compounds with $log\ K_{ow}$ less than 5.5 were calculated by using the following equation derived for aromatic compounds from this document:

$$log\ BCF = 0.71 \cdot log\ K_{ow} - 0.92$$

where

BCF = Bioconcentration factor for COPC in fish (unitless)
K_{ow} = Octanol-water partition coefficient (unitless)

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

U.S. EPA. 1995. *Review Draft Development of Human-Health Based and Ecologically - Based Exit Criteria for the Hazardous Waste Identification Project. Volumes I and II.* Office of Solid Waste. March.

This document recommends that the following references be used:

- *BCFs* for organic COPCs with log K_{ow} less than 4.0 should be based on equations presented in Thomann, R.V. 1989. "Bioaccumulation Model of Organic Chemical Distribution in Aquatic Food Chains." *Environmental Science and Technology*-23(b): 699-707.
- *BAFs* for organic COPCs with log K_{ow} greater than or equal to 4.0 and less than 6.5 are estimated on the basis of models presented in Thomann (1989) - see above - for the limnetic ecosystem, or for the littoral ecosystem, based on the following document:
    - Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. "An Equilibrium Model of Organic Chemical Accumulation in Aquatic Food Webs with Sediment Interaction." *Environmental Toxicology and Chemistry*. 11:615-629.
- For organics with log K_{ow} greater than or equal to 6.5, a default *BAF* of 1,000 was assumed on the basis of an analysis of available data on polycyclic aromatic hydrocarbons (PAH), and the following document:
    - Stephan, C.E. et al. 1993. "*Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lake Initiative.*" Office of Research and Development. U.S. EPA Research Laboratory. PB93-154672. Springfield, Virginia.
- *BCFs* for inorganics were obtained from various literature sources and the AQUIRE electronic database.

All *BCFs* and *BAFs* were corrected to 5 percent lipid, reflecting a typical value for a fish fillet.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December. Environmental Criteria and Assessment Office. ORD. Cincinnati, Ohio.

U.S. EPA. 1999. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Peer Review Draft. Office of Solid Waste. August.

TABLE B-4-27

FISH CONCENTRATION FROM BIOACCUMULATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates fish concentration from dissolved COPC concentration by using a bioaccumulation factor. Uncertainty associated with this equation include the following:

> Calculating C_{dw} uses on default values for variables F_{water} and C_{wtot}. Values for these two variables, in turn, depend on default medium-specific *OC* content values. Because *OC* content can vary widely at different locations in the same medium, significant uncertainty may be associated with F_{water} and C_{wtot}, and, in turn, C_{wt} in specific instances.

Equation

$$C_{fish} = C_{dw} \cdot BAF_{fish}$$

For mercury modeling, the concentration of COPC in fish is calculated for divalent mercury (Hg^{2+}) and methyl mercury (MHg) as shown in the following equations:

$$C_{fish_{(Hg^{2+})}} = C_{dw_{(Hg^{2+})}} \cdot BAF_{fish_{(Hg^{2+})}}$$

$$C_{fish_{(MHg)}} = C_{dw_{(MHg)}} \cdot BAF_{fish_{(MHg)}}$$

Variable	Description	Units	Value
C_{fish}	Concentration of COPC in fish	mg COPC/kg FW tissue	

TABLE B-4-27

FISH CONCENTRATION FROM BIOACCUMULATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
C_{dw}	Dissolved phase water concentration	mg COPC/L	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-24. Uncertainties associated with this variable include the following: (1) The variables in the equation in Table B-4-24 are site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to under- or overestimating C_{dw}. We expect the degree of uncertainty associated with *TSS* to be relatively small, because information regarding reasonable site-specific values for this variable is generally available or can be easily measured. (2) The uncertainty associated with the variables C_{wctot} and Kd_{sw} depends on estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, the uncertainty associated with using different *OC* content values may be significant in specific cases.
BAF_{fish}	Bioaccumulation factor for COPC in fish	L/kg FW tissue	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. As discussed in Appendix A-2, BAF_{fish} values were adjusted for dissolved water concentrations. We obtained BAFs for all organics with a log K_{ow} greater than or equal to 4.0 from U.S. EPA (1998), which cites U.S. EPA (1995a), U.S. EPA (1995b), and U.S. EPA (1994b). We calculated the BAF_{fish} value for lead as a geometric mean of data from various literature sources described in U.S. EPA (1998). We don't expect Elemental mercury to deposit significantly onto soils and surface water; therefore, assume no transfer of elemental mercury to fish. Assume that all mercury in fish exists or is converted to the methyl mercury (organic) form after uptake into the fish tissue. For this HHRAP, we use the BAF_{fish} value for methyl mercury listed in U.S. EPA (1997) for a trophic level 4 fish. The following uncertainty is associated with this variable: The COPC-specific *BAF* values may not accurately represent site-specific water body conditions, because estimates of *BAFs* can vary, based on experimental conditions.

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document cites the following documents as its sources of information regarding *BAF*s:

TABLE B-4-27

FISH CONCENTRATION FROM BIOACCUMULATION FACTORS USING DISSOLVED PHASE WATER CONCENTRATION (CONSUMPTION OF FISH EQUATIONS)

(Page 3 of 3)

U.S. EPA. 1993. "*Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative*." Office of Research and Development, U.S. Environmental Research Laboratory. Duluth, Minnesota. March.

This study presents three methods for estimating *BAFs*, in the following order of preference (first to last): (1) measured *BAF*; (2) measured *BCF* multiplied by a food-chain multiplier estimated from log K_{ow}; and (3) *BAF* estimated from log $K_{ow.}$

U.S. EPA 57 Federal Register 20802. 1993. "Proposed Water Quality Guidance for the Great Lakes System." April.

This document recommends using *BAF*s for compounds with a log K_{ow} greater than 5.5.

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

U.S. EPA. 1995a. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project. Volumes I and II.* Office of Solid Waste. March 3.

This document recommends that the following references be used.

- *BAFs* for organic COPCs with log K_{ow} greater than 4.0 but less than 6.5 should be calculated from the following references for the limetic ecosystem and the literal ecosystem, respectively.
  - Thomann, R.V. 1989. "Bioaccumulation Model of Organic Chemical Distribution in Aquatic Food Chains." *Environmental Science and Technology*. 23(6):699-707.
  - Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. "An Equilibrium Model of Organic Chemical Accumulation in Aquatic Food Webs with Sediment Interaction." *Environmental Toxicology and Chemistry*. 11:6115-629.
- *BAFs* for compounds with logK_{ow} greater than 6.5 were allowed to equal 1,000, based on an analysis of available data on PAHs and the following document:
  - Stephan, C.E. et al. 1993. "Derivation of Proposed Human Health and Wildlife Bioaccumulation Factors for the Great Lakes Initiative." Office of Research and Development, U.S. Environmental Research Laboratory. PB93-154672. Springfield, Virigina.

All *BAFs* were corrected to 5 percent lipid, reflecting a typical value for a fish fillet.

U.S. EPA. 1995b. *Great Lakes Water Quality Initiative. Technical Support Document for the Procedure to Determine Bioaccumulation Factors.* Office of Water. EPA-820-B-95-005. March.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment.* Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

U.S. EPA. 1998. "*Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions*." Update to EPA/600/6-90/003. Office of Research and Development, National Center for Environmental Assessment, U.S. EPA. EPA/600/R-98/137. December.

U.S. EPA. 1999. *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities.* Peer Review Draft. Office of Solid Waste. August.

TABLE B-4-28

FISH CONCENTRATION FROM BIOTA-TO-SEDIMENT ACCUMULATION FACTORS USING COPC SORBED TO BED SEDIMENT (CONSUMPTION OF FISH EQUATIONS)

(Page 1 of 3)

Description

This equation calculates fish concentration from bed sediment concentration, by using a biota-to-sediment accumulation factor (*BSAF*). Uncertainties associated with this equation include the following:

(1) Calculation of C_{sb} is largely dependent on default medium-specific *OC* content values. Because *OC* content can vary widely within a medium, significant uncertainty may be associated with estimates of C_{sb} in specific instances.
(2) Lipid content varies between different species of fish. Therefore, use of a default f_{lipid} value results in a moderate degree of uncertainty.
(3) Some species of fish have limited, if any, contact with water body sediments. Therefore, use of *BSAFs* to estimate the accumulation of COPCs in these species may be significantly uncertain.

Equation

$$C_{fish} = \frac{C_{sb} \cdot f_{lipid} \cdot BSAF}{OC_{sed}}$$

Variable	Description	Units	Value
C_{fish}	Concentration of COPC in fish	mg COPC/kg FW tissue	
C_{sb}	Concentration of COPC sorbed to bed sediment	mg COPC/kg bed sediment	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-25. Uncertainties associated with this variable include the following: (1) The default variable values recommended for use in the equation in Table B-4-25 may not accurately represent site-specific water body conditions. We expect the degree of uncertainty associated with variables θ_{bs}, *TSS*, d_{wc}, and d_{bs} to be limited either because the probable ranges for these variables are narrow or information allowing reasonable estimates is generally available. (2) Uncertainty associated with variables f_{bs}, C_{wtot}, and Kd_{bs} is largely associated with the use of default *OC* content values. Because *OC* content is known to vary widely in different locations in the same medium, use of default medium-specific values can result in significant uncertainty in some instances.

TABLE B-4-28

FISH CONCENTRATION FROM BIOTA-TO-SEDIMENT ACCUMULATION FACTORS USING COPC SORBED TO BED SEDIMENT (CONSUMPTION OF FISH EQUATIONS)

(Page 2 of 3)

Variable	Description	Units	Value
f_{lipid}	Fish lipid content	unitless	**0.03 to 0.07** We recommend this default range of values to be representative of warm water non-salmonoid fish (3 percent lipid content) at the low end and cold water salmonoid game fish at the high end (7 percent lipid content). Examples of non-salmonoid fish that may have lipid percentages in the edible portion at the lower end of the range would be catfish, northern pike, and walleye. U.S. EPA (1994a) and U.S. EPA (1994b) recommended values of 7 percent, which was originally cited by Cook et al. (1991). A value of 3 percent lipid content for the edible portion is provided by U.S. EPA (2000). The following uncertainty is associated with this variable: (1) Lipid content may vary between different species of fish. Therefore, using a default f_{lis} value may result in under- or overestimating C_{fish}.
$BSAF$	Biota-to-sediment accumulation factor	unitless ([mg COPC/kg lipid tissue]/[mg COPC/kg sediment])	**Varies** This variable is COPC-specific. We discuss this variable in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. These factors are applied only to PCDDs, PCDFs, and polychlorinated biphenyls (PCBs), consistent with NC DEHNR (1997), U.S. EPA (1992), U.S. EPA (1994), and U.S. EPA (1995). Uncertainty is associated with this variable: The greatest uncertainty associated with using *BSAFs* is that some species of fish have limited, if any, contact with water body sediments. Any accumulation of compounds into the tissue of these fishes is almost entirely the result of contact with surface water. Therefore, using *BSAFs* to estimate COPC accumulation in these species may be uncertain.
OC_{sed}	Fraction of organic carbon in bottom sediment	unitless	**0.04** This variable is site-specific. We recommend a default value of 0.04, the midpoint of the range (0.03 to 0.05), if site-specific information is not available. This is consistent with other U.S. EPA (1994b) and NC DEHNR (1997) guidance. The following uncertainty is associated with this variable:: The recommended OC_{sed} value may not accurately represent site-specific water body conditions. However, as indicated by the probable range of values for this parameter, we expect any uncertainty to be limited in most cases.

TABLE B-4-28

FISH CONCENTRATION FROM BIOTA-TO-SEDIMENT ACCUMULATION FACTORS USING COPC SORBED TO BED SEDIMENT (CONSUMPTION OF FISH EQUATIONS)

(Page 3 of 3)

REFERENCES AND DISCUSSION

Cook, P.M., D.W. Duehl, M.K. Walker, and R.E. Peterson. 1991. *Bioaccumulation and Toxicity of TCDD and Related Compounds in Aquatic Ecosystems*. In Gallo, M.A., R.J. Scheuplein, and K.A. Van Der Heijden (eds). *Banbury Report 35: Biological Basis for Risk Assessment of Dioxins and Related Compounds*. Cold Spring Harbor Laboratory Press. 0-87969-235-9/91.

This document is cited by U.S. EPA (1992) and U.S. EPA (1994) as the source of the fish lipid content value.

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document is cited as one of the reference source documents for biota-to-sediment factors for PCBs and dioxins. This document cites U.S. EPA (1992) as its source. This document is also cited as one of the reference documents for the default value for fraction *OC* in bottom sediment. The default value is the midpoint of the range obtained from U.S. EPA (1993). No source of this recommendation was identified.

This document is cited as one of the reference source documents for the fish lipid content value. The document cites Cook, Duehl, Walker, and Peterson (1991) as its original source of information. This document is also cited by U.S. EPA (1994) and NC DEHNR (1997) as the source of the *BSAF*s. *BSAF* values from this document were either measured values or estimates based on a whole fish lipid content of 7 percent. Specifically, *BSAF* values from this document must be evaluated because of the difficult experimental methods used to derive them.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume III: Site-specific Assessment Procedures*. External Review Draft. Office of Research and Development. Washington. D.C. EPA/600/6-88/005Cc. June.

U.S. EPA. 1994b. *Draft Guidance for Performing Screening Level Risk Analysis at Combustion Facilities Burning Hazardous Wastes. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. April.

This document is cited as one of the reference source documents for the fish lipid content value. The document cites Cook, Duehl, Walker, and Peterson (1991) as its original source of information. This document is also cited as one of the reference source documents for biota-to-sediment factors for PCBs and dioxins. This document cites U.S. EPA (1992) as its source of information. This document is also cited as one of the reference documents for the default fraction *OC* in bottom sediment value. The default value is the midpoint of the range obtained from U.S. EPA (1993). No source of this recommendation was identified.

U.S. EPA. 1995. *Review Draft Development of Human Health-Based and Ecologically-Based Exit Criteria for the Hazardous Waste Identification Project. Volumes I and II.* Office of Solid Waste. March.

U.S. EPA. 2000. *Draft Methodology for Deriving Ambient Water Quality Criteria for Protection of Human Health. Technical Support Document Volume III: Bioaccumulation Part 1 - Development of National Bioaccumulation Factors.* Office of Science and Technology in the Office of Water.

This document is cited as one of the reference source documents for the fish lipid content value.

TABLE B-5-1

AIR CONCENTRATION
(DIRECT INHALATION EQUATION)

(Page 1 of 3)

Description

This equation calculates the air concentration of a COPC based on the fraction in vapor phase and the fraction in particle phase.

Uncertainties associated with this equation include the following:

(1) Most of the uncertainties associated with the variables in this equation—specifically, those associated with variables *Q, Cyv*, and *Cyp*—are site-specific.

(2) In calculation of F_v, the equation assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, the use of the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than the S_T value for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.

Equation

For all COPCs (except mercury)

$$C_a = Q \cdot \left[F_v \cdot Cyv + (1.0 - F_v) \cdot Cyp\right]$$

Air concentration is calculated using (1) 0.002Q and $F_v = 1.0$ for elemental mercury (Hg^0) and (2) *0.48Q* and $F_v = 0.85$ for divalent mercury (Hg^{2+}). Elemental mercury is evaluated only for the inhalation exposure pathway (see discussion in Chapter 2).

$$C_{a(Hg^0)} = 0.002\,Q_{(Total)} \cdot \left[F_{v(Hg^0)} \cdot Cyv + \left(1.0 - F_{v(Hg^0)}\right) \cdot Cyp\right]$$

$$C_{a(Hg^{2+})} = 0.48\,Q_{(Total)} \cdot \left[F_{v(Hg^{2+})} \cdot Cyv + \left(1.0 - F_{v(Hg^{2+})}\right) \cdot Cyp\right]$$

Variable	Description	Units	Value
C_a	Air concentration	µg/m^3	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 for guidance on calculating this variable. Uncertainties associated with this variable are COPC- and site-specific.

TABLE B-5-1

AIR CONCENTRATION
(DIRECT INHALATION EQUATION)

(Page 2 of 3)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that $F_v = 0$ for all metals (except mercury). The following uncertainties are associated with this variable: (1) It uses a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Cyv	Unitized yearly air concentration from vapor phase	μg-s/g-m^3	**Varies** This variable is COPC- and site-specific and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are COPC- and site-specific.
Cyp	Unitized yearly air concentration from particle phase	μg-s/g-m^3	**Varies** This variable is COPC- and site-specific and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are COPC- and site-specific.

TABLE B-5-1

AIR CONCENTRATION
(DIRECT INHALATION EQUATION)

(Page 3 of 3)

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. "Atmospheric Processes." *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I*. Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document recommends using the equations in Bidleman (1988) to calculate F_v values for all organics other than dioxins (PCDD/PCDFs). However, this document does not present a recommendation for dioxins. This document also states that metals are generally entirely in the particulate phase ($F_v = 0$), except for mercury, which is assumed to be entirely in the vapor phase. The document does not state whether F_v for mercury should be calculated by using the equations in Bidleman (1988).

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document presents F_v values for organic COPCs that range from 0.27 to 1. F_v values for organics other than PCDD/PCDFs are calculated by using the equations presented in Bidleman (1988). The F_v value for PCDD/PCDFs is assumed to be 0.27. This value represents dioxin TEQs by weighting data for all dioxin and furan congeners with nonzero TEFs. This document presents F_v values for most inorganic COPCs equal to 0, based on the assumption that these COPCs are nonvolatile and assumed to be 100 percent in the particulate phase and 0 percent in the vapor phase.

U.S. EPA. 1997. *Mercury Study Report to Congress. Volume III: Fate and Transport of Mercury in the Environment*. Office of Air Quality and Planning and Standards and Office of Research and Development. EPA 452/R-97-005. December.

TABLE B-6-1

ACUTE AIR CONCENTRATION EQUATION
(ACUTE EQUATION)

(Page 1 of 3)

Description

This equation calculates the total air concentration of a COPC (hourly) based on the fraction in vapor phase and the fraction in particle phase.

Uncertainties associated with this equation include the following:

(1) Most of the uncertainties associated with the variables in this equation—specifically, those associated with variables Q, Chv, and Chp—are site-specific.

(2) In calculating F_v, the equation assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than the S_T value for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.

Equation

For all COPCs (except mercury)

$$C_{acute} = Q \cdot \left[F_v \cdot Chv + \left(1.0 - F_v\right) \cdot Chp \right]$$

Consistent with Table B-5-1, air concentration is calculated using (1) $0.002Q$ and $F_v = 1.0$ for elemental mercury (Hg^0) and (2) $0.48Q$ and $F_v = 0.85$ for divalent mercury (Hg^{2+}). Although calculated as separate species, acute toxicity benchmarks are not available for mercury and therefore, acute air concentration for each species should be summed for comparison to the acute toxicity benchmark for mercury.

$$C_{acute\left(Hg^0\right)} = 0.002\,Q_{(Total)} \cdot \left[F_{v\left(Hg^0\right)} \cdot Chv + \left(1.0 - F_{v\left(Hg^0\right)}\right) \cdot Chp \right]$$

$$C_{acute\left(Hg^{2+}\right)} = 0.48\,Q_{(Total)} \cdot \left[F_{v\left(Hg^{2+}\right)} \cdot Chv + \left(1.0 - F_{v\left(Hg^{2+}\right)}\right) \cdot Chp \right]$$

Variable	Description	Units	Value
C_{acute}	Acute air concentration	µg/m^3	
Q	COPC-specific emission rate	g/s	**Varies** This variable is COPC- and site-specific. See Chapters 2 and 3 for guidance regarding the calculation of this variable. Uncertainties associated with this variable are COPC- and site-specific.

TABLE B-6-1

ACUTE AIR CONCENTRATION EQUATION (ACUTE EQUATION)

(Page 2 of 3)

Variable	Description	Units	Value
F_v	Fraction of COPC air concentration in vapor phase	unitless	**0 to 1** This variable is COPC-specific. We discuss F_v in detail in Appendix A-2, and offer COPC-specific values in the HHRAP companion database. This range is based on values presented in Appendix A-2. Values are also presented in U.S. EPA (1994b) and NC DEHNR (1997). F_v was calculated using an equation presented in Junge (1977) for all organic COPCs, including PCDDs and PCDFs. U.S. EPA (1994c) states that F_v = 0 for all metals (except mercury). The following uncertainties are associated with this variable: (1) It assumes a default S_T value for background plus local sources, rather than an S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than that for background plus local sources, and it would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower. (2) According to Bidleman (1988), the equation used to calculate F_v assumes that the variable c (Junge constant) is constant for all chemicals; however, the value of c depends on the chemical (sorbate) molecular weight, the surface concentration for monolayer coverage, and the difference between the heat of desorption from the particle surface and the heat of vaporization of the liquid-phase sorbate. To the extent that site- or COPC-specific conditions may cause the value of c to vary, uncertainty is introduced if a constant value of c is used to calculate F_v.
Chv	Unitized hourly air concentration from vapor phase	µg-s/g-m^3	**Varies** This variable is COPC- and site-specific and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are COPC- and site-specific.
Chp	Unitized hourly air concentration from particle phase	µg-s/g-m^3	**Varies** This variable is COPC- and site-specific and is determined by air dispersion modeling (see Chapter 3). Uncertainties associated with this variable are COPC- and site-specific.

REFERENCES AND DISCUSSION

Bidleman, T.F. 1988. “Atmospheric Processes.” *Environmental Science and Technology*. Volume 22. Number 4. Pages 361-367.

For discussion, see References and Discussion, Table B-1-1.

Junge, C.E. 1977. *Fate of Pollutants in Air and Water Environments, Part I.* Suffet, I.H., Ed. Wiley. New York. Pages 7-26.

TABLE B-6-1

ACUTE AIR CONCENTRATION EQUATION
(ACUTE EQUATION)

(Page 3 of 3)

NC DEHNR. 1997. *Final NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

This document recommends using the equations in Bidleman (1988) to calculate F_v values for all organics other than dioxins (PCDD/PCDFs). However, this document does not present a recommendation for dioxins. This document also states that metals are generally entirely in the particulate phase ($F_v = 0$), except for mercury, which is assumed to be entirely in the vapor phase. The document does not state whether F_v for mercury should be calculated by using the equations in Bidleman (1988).

U.S. EPA. 1994. *Revised Draft Guidance for Performing Screening Level Risk Analysis 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.

This document presents F_v values for organic COPCs that range from 0.27 to 1. F_v values for organics other than PCDD/PCDFs are calculated by using the equations presented in Bidleman (1988). The F_v value for PCDD/PCDFs is assumed to be 0.27. This value represents dioxin TEQs by weighting data for all dioxin and furan congeners with nonzero TEFs. This document presents F_v values for most inorganic COPCs equal to 0, based on the assumption that these COPCs are nonvolatile and assumed to be 100 percent in the particulate phase and 0 percent in the vapor phase.

U.S. EPA. 1997. “Mercury Study Report to Congress.” Volume III. Draft. Office of Air Quality and Planning and Standards and Office of Research and Development. December.

APPENDIX C
RISK CHARACTERIZATION EQUATIONS

Table	Equation
C-1-1	COPC INTAKE FROM SOIL
C-1-2	COPC INTAKE FROM PRODUCE
C-1-3	COPC INTAKE FROM BEEF, MILK, PORK, POULTRY, AND EGGS
C-1-4	COPC INTAKE FROM FISH
C-1-5	COPC INTAKE FROM DRINKING WATER
C-1-6	TOTAL DAILY INTAKE
C-1-7	INDIVIDUAL CANCER RISK
C-1-8	HAZARD QUOTIENT
C-1-9	TOTAL CANCER RISK
C-1-10	TOTAL HAZARD INDEX
C-1-11	SEGREGATED HAZARD INDEX FOR SPECIFIC ORGAN EFFECTS
C-2-1	INHALATION CANCER RISK FOR INDIVIDUAL CHEMICALS
C-2-2	INHALATION HAZARD QUOTIENT FOR COPCS
C-2-3	TOTAL INHALATION CANCER RISK
C-2-4	HAZARD INDEX FOR INHALATION
C-3-1	CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK
C-3-2	AVERAGE DAILY DOSE TO THE EXPOSED INFANT
C-4-1	ACUTE HAZARD QUOTIENT

TABLE C-1-1

COPC INTAKE FROM SOIL

(Page 1 of 4)

Description

This equation calculates the daily intake of COPC from soil consumption. The soil concentration will vary with each scenario location, and the soil consumption rate varies for children and adults. Uncertainties associated with this equation include:

(1) Assume that the amount of soil intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{soil}.
(2) The standard assumptions regarding period exposed might not be representative of an actual exposure situation. This assumption may under- or overestimate I_{soil}.

Equation

$$I_{soil} = \frac{Cs \cdot CR_{soil} \cdot F_{soil}}{BW}$$

Variable	Description	Units	Value
I_{soil}	Daily intake of COPC from soil	mg/kg-day	
Cs	Average soil concentration over exposure duration	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-1-1. Cs varies based on whether the COPC is carcinogenic or noncarcinogenic. For carcinogenic COPCs, this value is equal to the soil concentration averaged over the exposure duration (Table B-1-1) (U.S. EPA 1994 and NC DEHNR 1997). For noncarcinogenic COPCs, this value is equal to the highest annual soil concentration occurring within the exposure duration. The highest annual soil concentration would most likely occur at the end of the time period of combustion (Table B-1-1) (U.S. EPA 1994 and NC DEHNR 1997). The following uncertainties are associated with this variable: (1) We assume that the time period for deposition of COPCs resulting from hazardous waste combustion is a conservative, long-term value. This assumption may overestimate Cs and Cs_{tD}. (2) Exposure duration values (T_2) are based on historical mobility studies and won't necessarily remain constant. Specifically, mobility studies indicate that most receptors that move remain in the vicinity of the combustion unit; however, it is impossible to accurately predict the probability that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. (3) Using a value of zero for T_1 doesn't account for exposure that may have occurred from historic operations and emissions from hazardous waste combustion. This may underestimate Cs and Cs_{tD}. (4) For soluble COPCs, leaching might lead to movement to below the mixing depth, resulting in lower concentrations within the mixing depth. This uncertainty may overestimate Cs and Cs_{tD}. (5) Deposition to hard surfaces may result in dust residues that have negligible dilution (as a result of potential mixing with *in situ* materials) compared to other residues. This may underestimate Cs and Cs_{tD}.

TABLE C-1-1

COPC INTAKE FROM SOIL

(Page 2 of 4)

Variable	Description	Units	Value
CR_{soil}	Consumption rate of soil	kg/day	**0.0001 to 0.0002** The soil consumption rate varies for the adult and child receptors (U.S. EPA 1997). Receptor — Intake Rate (kg/day) Adult — 0.0001 Child — 0.0002 U.S. EPA (1997) recommends using a child intake rate of 0.0002 kg/day for a child receptor as a conservative estimate of exposure. These values are consistent with U.S. EPA (1991). U.S. EPA (1997) references studies done by Hawley (1985) and Calabrese (1990) as the sources used to derive soil consumption rates. Uncertainties associated with this variable include: (1) Tracer studies have resulted in wide ranging estimates of the amount of soil and dust ingested by young children, making it difficult to identify a single value to use. It is also extremely difficult to separate the contribution of exposure resulting from exterior soil vs. interior dust. As a result, the intake rate is reported as the combined rate for soils and dusts. This uncertainty may under- or overestimate CR_{soil}. (2) Our recommended intake rates may not accurately represent behavioral characteristics since they are upper estimates. This uncertainty may overestimate CR_{soil}. (3) The intake rates represent normal mouthing tendencies. Some children exhibit abnormal mouthing (i.e. “pica”) behavior and would have much higher intake rates. This uncertainty may considerably underestimate CR_{soil}.
F_{soil}	Fraction of soil that is contaminated	unitless	**1.0** We recommend assuming that all consumed soil is contaminated. This is consistent with NC DEHNR (1997) and U.S. EPA (1994), which assume the fraction of consumed soil contaminated is 1.0 for all exposure scenarios. Uncertainty associated with this variable include: U.S. EPA guidance recommends the fraction of consumed soil contaminated is equal to 1.0. However, due to variations in the proximity of the receptor to the contaminated source, size of the contaminated source, receptors of concern, mobility of receptors, and nature of exposure, this assumption may underestimate or overestimate F_{soil}.

TABLE C-1-1

COPC INTAKE FROM SOIL

(Page 3 of 4)

Variable	Description	Units	Value
BW	Body weight	kg	**15 or 70** We recommend using default values of 70 (adults) and 15 (children). These default values are consistent with U.S. EPA (1991; 1994). Uncertainty associated with this variable include: These body weights represent the average weight of an adult and child. However, depending on the actual receptor, body weights may be higher or lower. These default values may overestimate or underestimate actual body weights. However, the degree of under- or overestimation is not expected to be significant.

REFERENCES AND DISCUSSION

Calabrese, E.J., Stanek, E.J., Gilbert, C.E., and Barnes, R.M. 1990. Preliminary adult soil ingestion estimates; results of a pilot study. *Regul. Toxicol. Pharmacol.* 12:88-95.

This document is cited by U.S. EPA (1997) as a source of information used to derive soil consumption rates.

Hawley, J.K. 1985. Assessment of health risk from exposure to contaminated soil. *Risk Analysis* 5:289-302.

This document is cited by U.S. EPA (1997) as a source of information used to derive soil consumption rates.

NC DEHNR. 1997. *North Carolina Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units. January.*

This document is one of the sources for the equation in Table C-1-1. This document also states that (1) for carcinogenic COPCs, *Cs* is equal to the soil concentration averaged over the exposure duration; however, no reference document is cited and (2) for noncarcinogenic COPCs, *Cs* is equal to the highest annual soil concentration occurring within the exposure duration; the highest annual soil concentration would occur at the end of the time period of emissions.

U.S. EPA. 1991. *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.6-03. Washington, D.C. March 21.

This document is cited as the reference source document of soil ingestion rates, and the exposure frequency and body weight variables.

U.S. EPA. 1994. *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.

This document is one of the sources for the equation in Table C-1-1. This document also states that (1) for carcinogenic COPCs, *Cs* is equal to the soil concentration averaged over the exposure duration; however, no reference document is cited and (2) for noncarcinogenic COPCs, *Cs* is equal to the highest annual soil concentration occurring within the exposure duration; the highest annual soil concentration would occur at the end of the time period of emissions.

TABLE C-1-1

COPC INTAKE FROM SOIL

(Page 4 of 4)

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for soil consumption rates.

TABLE C-1-2

COPC INTAKE FROM PRODUCE

(Page 1 of 4)

Description

This equation calculates the daily intake of COPC from ingestion of exposed aboveground, protected aboveground, and belowground produce. The consumption rate varies for children and adults, and for the type of produce. The concentration in exposed aboveground, protected aboveground, and belowground produce will also vary with each scenario location.

Consumption rates were derived from the *Exposure Factors Handbook* (U.S. EPA 1997). U.S. EPA (1997) presents consumption rates based on body weight; therefore, body weight is not included as a variable in the calculation of I_{ag}.

Uncertainties associated with this equation include the following:

(1) The amount of produce intake is assumed to be constant and representative of the exposed population. This assumption may under- or overestimate I_{ag}.
(2) The standard assumptions regarding period exposed may not be representative of any actual exposure situation. This assumption may under- or overestimate I_{ag}.

Equation

$$I_{ag} = [((Pd + Pv + Pr) \cdot CR_{ag}) + (Pr \cdot CR_{pp}) + (Pr_{bg} \cdot CR_{bg})] \cdot F_{ag}$$

Variable	Description	Units	Value
I_{ag}	Daily intake of COPC from produce	mg/kg-day DW	
Pd	Aboveground exposed produce concentration due to direct (wet and dry) deposition onto plant surfaces	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-2-7. Uncertainties associated with this variable include the following: (1) The recommended equation for calculating *kp* does not consider chemical degradation processes. Adding chemical degradation processes would decrease half-lifes and thereby increase *kp* values; plant concentration decreases as *kp* increases. Using a *kp* value that does not consider chemical degradation processes is protective. (2) Estimating other parameter values (for example, *Fw* and *Rp*) is based directly or indirectly on studies of vegetation other than aboveground produce (primarily grasses). Uncertainty is introduced to the extent that the calculated parameter values do not accurately represent aboveground produce-specific values.

TABLE C-1-2

COPC INTAKE FROM PRODUCE

(Page 2 of 4)

Variable	Description	Units	Value
Pv	Aboveground exposed produce concentration due to air-to-plant transfer	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-2-8. Uncertainties associated with this variable include the following: (1) The range of values for the variable *Bv* (air-to-plant biotransfer factor) is about 19 orders of magnitude for organic COPCs. (2) The algorithm used to calculate values for the variable F_v assumes a default value for the parameter S_T (Whitby's average surface area of particulates [aerosols]) of background plus local sources rather than an S_T value for urban sources. If a specific site is located in an urban area, the use of the latter S_T value may be more appropriate. The S_T value for urban sources is about one order of magnitude greater than that for background plus local sources and would result in a lower F_v value; however, the F_v value is likely to be only a few percent lower.
Pr	Aboveground exposed and protected produce concentration due to root uptake	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-2-9. Uncertainty associated with this variable include the following: Estimated COPC-specific soil-to-plant bioconcentration factors (*Br*) may not be representative of site-specific conditions.
Pr_{bg}	Belowground produce concentration due to root uptake	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-2-10. Uncertainty associated with this variable include the following: Estimated COPC-specific soil-to-plant bioconcentration factors (*Br*) may not be representative of site-specific conditions.

TABLE C-1-2

COPC INTAKE FROM PRODUCE

(Page 3 of 4)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr>
<td>CR_{ag};
CR_{pp};
CR_{bg}</td>
<td>Consumption rate of aboveground, protected aboveground, and belowground produce, respectively</td>
<td>kg/kg-day DW</td>
<td>This variable is site-specific. The recommended default values represent the total of the following produce-specific ingestion rates:
<table>
<tr><th></th><th colspan="6">Ingestion Rate (kg/kg-day DW)</th></tr>
<tr><th>Plant Type</th><th>Farmer</th><th>Farmer Child</th><th>Resident</th><th>Res. Child</th><th>Fisher</th><th>Fisher Child</th></tr>
<tr><td>Exposed Aboveground Produce (Cr_{ag})</td><td>0.00047</td><td>0.00113</td><td>0.00032</td><td>0.00077</td><td>0.00032</td><td>0.00077</td></tr>
<tr><td>Protected Aboveground Produce (Cr_{pp})</td><td>0.00064</td><td>0.00157</td><td>0.00061</td><td>0.00150</td><td>0.00061</td><td>0.00150</td></tr>
<tr><td>Belowground Produce (Cr_{bg})</td><td>0.00017</td><td>0.00028</td><td>0.00014</td><td>0.00023</td><td>0.00014</td><td>0.00023</td></tr>
</table>
We derived ingestion rates from U.S. EPA (1997), Tables 13-61, 13-62, 13-63, 13-64, and 13-65. The ingestion rates listed in U.S. EPA (1997) are derived from the 1987-1988 USDA National Food Consumption Survey and may be used to assess exposure to contaminants in foods grown at a specific site. The ingestion rates were adjusted for cooking and preparation loss as recommended by U.S. EPA (1997). The amount of preparation and cooking loss varied depending on type of food consumed. We assumed that no preparation and cooking loss occurs with exposed fruits because we further assumed this type of fruit is eaten in the raw form. In addition, ingestion rates for the child receptor represent a time-weighted mean from the respective tables.

Uncertainty associated with this variable include the following:
The recommended ingestion rates are based on national average home produced consumption rates. Site-specific ingestion rates may be higher or lower than those recommended. Therefore, using the recommended ingestion rates may under- or overestimate I_{ag}.</td>
</tr>
<tr>
<td>F_{ag}</td>
<td>Fraction of produce that is contaminated</td>
<td>unitless</td>
<td>1.0
This variable is site-specific. We recommend using the default value in the absence of site-specific information, consistent with U.S. EPA (1997).

Only that portion of the diet produced at home (and therefore exposed to facility emissions) is of consequence in the risk assessment. As detailed above (in Section 6.2.2.2), the consumption rates we recommend represent only the home-produced portion of the diet. Therefore, by using consumption rates specific to home produced foods, it is reasonable to assume that 100% of those home produced foods are contaminated.

The following uncertainty is associated with this variable:
Using default values may overestimate F_{ag}.</td>
</tr>
</table>

TABLE C-1-2

COPC INTAKE FROM PRODUCE

(Page 4 of 4)

REFERENCES AND DISCUSSION

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. *Review and Analysis of Parameters and Assessing Transport of Environmentally Released Radionuclides through Agriculture*. Oak Ridge National Laboratory. Oak Ridge, Tennessee.

This document is cited as a source for *Br* values.

U.S. EPA 1990. *Exposure Factors Handbook.* Office of Health and Environmental Assessment, Exposure Assessment Group. Washington, D.C. March.

This is the document cited as the source of the fraction of produce that is contaminated (F_{ag}) the adult resident, child resident, and fisher. U.S. EPA assumes that F_{ag} for the fisher child is the same as for the fisher.

U.S. EPA 1992. *Technical Support Document for Land Application of Sewage Sludge*. Volumes I and II. Office of Water. Washington, D.C. EPA 822/R-93-001a.

This document is cited as a source for plant uptake response slope factors.

U.S. EPA. 1994. *Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste*. Office of Emergency and Remedial Response. Office of Solid Waste.

This document is cited as the source of the fraction of produce that is contaminated (F_{ag}) for the farmer (U.S. EPA assumes that F_{ag} for the farmer child is the same as for the farmer).

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for produce consumption rates.

TABLE C-1-3

COPC INTAKE FROM BEEF, MILK, PORK, POULTRY, AND EGGS

(Page 1 of 4)

Description

This equation calculates the daily intake of COPCs from the ingestion of animal tissue (where the *i* in the equation refers to beef, milk, pork, poultry, or eggs). The consumption rate varies for children and adults and for the type of animal tissue (*j*). The concentration in the animal tissue will also vary with each scenario location.

Consumption rates were derived from the *Exposure Factors Handbook* (U.S. EPA 1997). U.S. EPA (1997) presents consumption rates based on body weight; therefore, body weight is not included as a variable in the calculation of I_i.

Uncertainties associated with this equation include the following:

(1) The amount of animal tissue intake is assumed to be constant and representative of the exposed population. This assumption may under- or overestimate $I_{i.}$

(2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_i.

Equation

$$I_i \;=\; A_i \cdot CR_i \cdot F_i$$

Variable	Description	Units	Value
I_i	Daily intake of COPC *i* from animal *j* tissue	mg/kg-day	
A_j	Concentration of COPC *i* in animal tissue *j*	mg/kg FW	**Varies** This variable is COPC- and site-specific, and is calculated using output from the equations in Tables B-3-10, B-3-11, B-3-12, B-3-13, and B-3-14. Uncertainties associated with this variable include the following: (1) Based on the information provided, A_{beef} and A_{pork} depend on estimated concentrations of COPCs in plant feeds and soil, and the biotransfer factors estimated for each constituent. To the extent the estimated concentrations and biotransfer factors do not reflect site-specific or local conditions, A_{beef} may be under- or overestimated. (2) Uptake of COPCs into chicken and eggs has typically been applied only to PCDDs and PCDFs but could possibly be used to calculate A_{chick} and A_{egg} resulting from other COPCs. (3) The assumption that 10 percent of a chicken's diet is soil may not represent site-specific or local conditions of chickens raised on farms. Stephens, Petreas, and Hayward (1992) and Stephens, Petreas, and Hayward (1995) suggest the percentage of soil in the diet of chickens raised under field conditions may be greater than 10 percent. Therefore, the concentration of COPCs in eggs, A_{egg}, and the concentration of COPCs in chicken, A_{chick}, may be underestimated.

TABLE C-1-3

COPC INTAKE FROM BEEF, MILK, PORK, POULTRY, AND EGGS

(Page 2 of 4)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr>
<td>CR_j</td>
<td>Consumption rate of animal tissue j</td>
<td>kg/kg-day FW</td>
<td>
Varies
This variable is site-specific. We derived our recommended ingestion rates of animal tissues (see the equation in Table C-1-4 for fish ingestion) from U.S. EPA (1997):

Animal Tissue Ingestion Rates (kg/kg-day FW)
<table>
<tr><th></th><th>Farmer</th><th>Farmer Child</th></tr>
<tr><td>Homegrown Beef</td><td>0.00122</td><td>0.00075</td></tr>
<tr><td>Homegrown Milk</td><td>0.01367</td><td>0.02268</td></tr>
<tr><td>Homegrown Poultry</td><td>0.00066</td><td>0.00045</td></tr>
<tr><td>Homegrown Eggs</td><td>0.00075</td><td>0.00054</td></tr>
<tr><td>Homegrown Pork</td><td>0.00055</td><td>0.00042</td></tr>
</table>

Ingestion rates were based on data from U.S. EPA (1997) Tables 13-28, 13-36, 13-43, 43-54, and 13-55. The ingestion rates listed in U.S. EPA (1997) were derived from the 1987-1988 USDA National Food Consumption Survey and may be used to assess exposure to contaminants in foods raised at a specific site. Prior to the adjustment for cooking and preparation loss, the mean individual meat consumption rates were weighted by age group. The ingestion rates were then adjusted for cooking and preparation loss as recommended in U.S. EPA (1997). The total preparation and cooking loss was in the range of 45 to 54 percent for beef, pork, and poultry.

In addition, ingestion rates for the child receptor represent a time-weighted mean from the respective tables. Where data for a specific age group was incomplete, the intake was extrapolated using data from the general population (Tables 11-11 and 11-13 of U.S. EPA 1997). See HHRAP Section 6.2.2.2 for a more detailed explanation of our recommended method for estimating age-appropriate consumption rates.

U.S. EPA (1997) provides information for total home produced dairy (Table 13-28 of U.S. EPA 1997), but does not specify intake for fluid milk.

For the metals mercury, selenium, and cadmium, the concentration in beef, milk, and pork, and the consumption rate are in kilograms dry weight per day. Wet-weight to dry-weight conversion information for beef, milk, and pork is presented in U.S. EPA (1997)

The following uncertainty is associated with this variable:
The recommended tissue-specific consumption rates may not accurately reflect site-specific local conditions. As a result, tissue-specific intakes may be under- or overestimated.
</td>
</tr>
</table>

TABLE C-1-3

COPC INTAKE FROM BEEF, MILK, PORK, POULTRY, AND EGGS

(Page 3 of 4)

Variable	Description	Units	Value
F_j	Fraction of animal tissue *j* that is contaminated	unitless	**1.0** This variable is site-specific. We recommend an F_j of 1.0 for all animal tissues consumed. This recommendation is consistent with NC DEHNR (1997). The following uncertainty is associated with this variable: The fraction of animal tissue that is contaminated is site-specific; therefore, any of the following may be under- or overestimated: variations in the proximity of the receptor to the contaminated source, size of the contaminated source, receptors of concern, mobility of receptors, and nature of exposure.

REFERENCE AND DISCUSSIONS

Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1992. "Biotransfer and Bioaccumulation of Dioxins and Dibenzofurans from Soil." Hazardous Materials Laboratory, California Department of Health Services. Berkeley, California. Presented at the 12th International Symposium on Dioxins and Related Compounds. August 24 through 28. University of Tampere, Tampere, Finland.

This document is cited as the source of the assumption that free-range chickens ingest soil as 10 percent of their diet and as the source of the dioxin and furan congeners-specific *BCF*s recommended by NC DEHNR (1997). However this document does not clearly reference or document the assumption that soil represents 10 percent of a free-range chicken's diet. The document appears to cite two other documents as supporting its assumption: (1) Chang, Hayward, Goldman, Harnly, Flattery and Stephens (1989) and (2) Petreas, Goldman, Hayward, Chang, Flattery, Wiesmuller, Stephens, Fry, and Rappe (1992).

Also, this document presents dioxin and furan congener-specific *BCF*s (thigh) for the low- exposure group after 80 days of a 178-day total exposure period. The chickens in the low-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 42 ppt I-TEQ. Chickens in the high-dose group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 458 ppt I-TEQ; *BCF* results were not presented from the high-dose group.

Stephens, R.D., M.X. Petreas, and D.G. Hayward. 1995. "Biotransfer and Bioaccumulation of Dioxins and Furans from Soil: Chickens as a Model for Foraging Animals." *The Science of the Total Environment.* Volume 175: 253-273.

This document is an expansion of the results originally presented in Stephens, Petreas, and Hayward (1992). In particular, this document suggests that the percentage of soil in the diet of chickens raised under field conditions is likely to be greater than 10 percent, the value that was used in the experimental study presented in this document.

Also, this document presents dioxin and furan congener-specific *BCF*s (thigh) under two exposure schemes; low exposure and high exposure. The white leghorn (Babcock D 300) chickens in the low group were fed a diet containing 10 percent soil with a PCDD/PCDF concentrations of 42 ppt I-TEQ. Chickens in the high group were fed a diet containing 10 percent soil with a PCDD/PCDF concentration of 460 ppt I-TEQ (some congeners were fortified by spiking).

The *BCF*s presented for low- and high-dose groups both represent averages of results from Day-80 and Day-164 of a total 178-day exposure period.

TABLE C-1-3

COPC INTAKE FROM BEEF, MILK, PORK, POULTRY, AND EGGS

(Page 4 of 4)

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for home produced beef, milk, pork, poultry, and egg consumption rates.

TABLE C-1-4

COPC INTAKE FROM FISH

(Page 1 of 4)

Description

This equation calculates the daily intake of COPCs from the ingestion of fish. Consumption rates were derived from the *Exposure Factors Handbook* (U.S. EPA 1997). U.S. EPA (1997) presents consumption rates based on body weight; therefore, body weight is not included as a variable in the calculation of I_{fish}.

The limitations and uncertainty introduced in calculating this value include the following:

(1) The amount of fish intake is assumed to be constant and representative of the exposed population. This assumption may under- or overestimate I_{fish}.
(2) The standard assumptions regarding period exposed may not be representative of any actual exposure situation. This assumption may under- or overestimate I_{fish}.

Equation

$$I_{fish} = C_{fish} \cdot CR_{fish} \cdot F_{fish}$$

Variable	Description	Units	Value
I_{fish}	Daily intake of COPC from fish	mg/kg-day	
C_{fish}	Concentration in fish	mg/kg	**Varies** This variable is COPC- and site-specific, and is calculated using output from the equations in Tables B-4-26 through B-4-28; the fish concentration will vary for each water body. The following uncertainty is associated with this variable: The methodology does not account for concentration variations across fish species. Different species may accumulate COPCs to different extents depending, for example, on their feeding habits and fat content. This may cause C_{fish} to be under- or overestimated.

TABLE C-1-4

COPC INTAKE FROM FISH

(Page 2 of 4)

Variable	Description	Units	Value
CR_{fish}	Consumption rate of fish	kg/kg-day FW	**Varies** The consumption rate varies for the receptor considered. The following home produced or caught ingestion rates for fish were derived from U.S. EPA (1997): Receptor — Ingestion Rate (kg/kg-day FW) Fisher — 0.00125 Fisher Child — 0.00088 We derived ingestion rates using data from U.S. EPA (1997) Table 13-23. The ingestion rates listed in U.S. EPA (1997) were derived from the 1987-1988 USDA National Food Consumption Survey and may be used to assess exposure to contaminants in foods grown, raised, or caught at a specific site. Prior to the adjustment for cooking and preparation loss, the mean individual fish consumption rates were weighted by age group. The ingestion rates were then adjusted for cooking and preparation loss as recommended in U.S. EPA (1997). The total preparation and cooking loss for fish was 41 percent. In addition, the ingestion rates for the child receptor represents a time-weighted mean. Where data for a specific age group was incomplete, the intake was extrapolated using data from the general population (Table 10-46 of U.S. EPA 1997). See HHRAP Section 6.2.2.2 for a more detailed explanation of our recommended method for estimating age-appropriate consumption rates. Uncertainties introduced by assumptions made to calculate this value include the following: (1) Our recommended intake rates don't necessarily take into account the types of fish that will be present in the water body. Separate intake rates are needed for freshwater and estuarine fish and shellfish, depending on the nature of the local surface water body. This assumption can overestimate or underestimate CR_{fish}. (2) These intake rates do not represent long behavior patterns, which is the focus of the exposure assessments used to support chronic health effects. This introduces uncertainty into the estimates of medians and other percentiles. This assumption can overestimate or underestimate CR_{fish}. (3) The intake rates represent total intake rates of home-caught fish. Where use of site-specific information would reveal the amount of fish consumed from waters within the study area, this information should be used. This assumption can overestimate or underestimate CR_{fish}.
F_{fish}	Fraction of fish that is contaminated	unitless	**1.0** We recommend using this default value if site-specific information is not available. The contaminated fraction will vary with each exposure scenario; however, NC DEHNR (1997) and U.S. EPA (1994) assume that this value equals 1.0 for the fisher. The following uncertainty is associated with this variable: Using 1.0 as a default value for fraction of fish that is contaminated assumes that receptors consume only contaminated fish; this assumption may overestimate F_{fish}.

TABLE C-1-4

COPC INTAKE FROM FISH

(Page 3 of 4)

REFERENCES AND DISCUSSION

NC DEHNR. 1997. *NC DEHNR Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Combustion Units.* January.

This document is one of the reference source documents for the equation in Table C-1-4.

U.S. EPA. 1994. *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.

This document is one of the reference source documents for the equation in Table C-1-4.

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for home-caught fish consumption rates.

TABLE C-1-5

COPC INTAKE FROM DRINKING WATER

(Page 1 of 3)

Description

This equation calculates the daily intake of COPC from drinking water. COPC intake from drinking water is calculated from the concentration of COPC dissolved in the water column of each surface water body or watershed identified as a drinking water source. The dissolved concentration is used to estimate COPC intake from drinking water because it is assumed the water is filtered prior to human consumption. The COPC concentration will vary for each water body. The consumption rate varies for children and adults.

Uncertainties associated with this equation include the following:

(1) The amount of drinking water intake is assumed to be constant and representative of the exposed population. This assumption may under- or overestimate I_{dw}.
(2) The standard assumptions regarding period exposed may not be representative of any actual exposure situation. This assumption may under- or overestimate I_{dw}.

Equation

$$I_{dw} = \frac{C_{dw} \cdot CR_{dw} \cdot F_{dw}}{BW}$$

Variable	Description	Units	Value
I_{dw}	Daily intake of COPC from drinking water	mg/kg-day	
C_{dw}	Dissolved phase water concentration	mg/L	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-4-24. Uncertainties associated with this variable include the following: All of the variables in the equation in Table B-4-24 are COPC- and site-specific. Therefore, using default values rather than site-specific values, for any or all of these variables, will contribute to the under- or overestimating C_{wt}. We expect the degree of uncertainty associated with the variables d_w and d_b to be minimal because information for estimating d_w is generally available and the probable range for d_b is narrow. The uncertainty associated with the variables F_{water} and C_{wtot} is associated with estimates of *OC* content. Because *OC* content values can vary widely for different locations in the same medium, using default *OC* values may result in significant uncertainty in specific cases.

TABLE C-1-5

COPC INTAKE FROM DRINKING WATER

(Page 2 of 3)

Variable	Description	Units	Value
CR_{dw}	Rate of consumption of drinking water	L/day	**0.67 or 1.4** This variable is site-specific. We recommend using default values of 0.67 (child) and 1.4 (adult) in the absence of site-specific data. Our recommended average adult consumption rate of drinking water is based on information cited in U.S. EPA (1997). For the child receptor, U.S. EPA (1997) provides recommended drinking water intake rates for various age groups in Table 3-30. The child default drinking water intake was derived using a time-weighted average for the age groups 0 to 6 years of age. The following uncertainty is associated with this variable: The average consumption rate of drinking water is based on the average intake observed from five studies. The number of studies conduct may underestimate or underestimate CR_{dw}.
F_{dw}	Fraction of drinking water that is contaminated	unitless	**1.0** This variable is site-specific. Consistent with U.S. EPA (1994), we recommend assuming 1.0 for the fraction of drinking water that is contaminated. The following uncertainty is associated with this variable: Some receptors may consume a fraction of their drinking water from sources unimpacted by facility emissions. Therefore, this assumption will likely overestimate F_{dw}.
BW	Body weight	kg	**15 or 70** This variable is site-specific. We recommend using default values of 15 (children) or 70 (adults) in the absence of site-specific information. These default values are consistent with U.S. EPA (1991; 1994). Uncertainties associated with this variable include: These body weights represent the average weight of an adult and child. However, depending on the receptor, the body weights may be higher or lower. These default values may overestimate or underestimate actual body weights. However, the degree of under- or overestimation is not expected to be significant.

TABLE C-1-5

COPC INTAKE FROM DRINKING WATER

(Page 3 of 3)

REFERENCES AND DISCUSSION

U.S. EPA. 1991. *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.6-03. Washington, D.C. March 21.

This document is cited as the reference source document of the exposure frequency and body weight variables.

U.S. EPA. 1994. *Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Wastes*. Office of Emergency and Remedial Response. Office of Solid Waste.

This document was cited as the source of the fraction of drinking water that is contaminated.

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document is the source for the drinking water consumption rates.

TABLE C-1-6

TOTAL DAILY INTAKE

(Page 1 of 3)

Description

This equation calculates the daily intake of COPC via all indirect exposure pathways. As discussed in Chapter 4 and Table 4-1, the indirect exposure pathways considered in the calculation of the total daily intake of COPCs are specific to the recommended exposure scenario evaluated and the representative exposure setting. Daily intake values from exposure pathways which are not evaluated in a respective exposure scenario may be assumed to be zero when calculating the total daily intake of COPC (*I*).

Uncertainties associated with this equation include the following:

(1) The uncertainties associated with estimates of total intake are those associated with each of the medium- or tissue-specific intakes.
(2) To the extent that medium- or tissue-specific intakes do not accurately represent site-specific local conditions, *I* may be under- or overestimated.

Equation

$$I = I_{soil} + I_{ag} + I_{beef} + I_{milk} + I_{fish} + I_{pork} + I_{poultry} + I_{eggs} + I_{dw}$$

Variable	Description	Units	Value
I	Total daily intake of COPC	mg/kg-day	
I_{soil}	Daily intake of COPC from soil	mg/kg-day	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-1. The value for this variable will vary for each receptor and each exposure scenario location. Uncertainties associated with this variable include the following: (1) We assume the amount of soil intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{soil}. (2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_{soil}.

TABLE C-1-6

TOTAL DAILY INTAKE

(Page 2 of 3)

Variable	Description	Units	Value
I_{ag}	Daily intake of COPC from aboveground produce	mg/kg-day DW	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-2. The value for this variable will vary for each receptor and each exposure scenario location. Uncertainties associated with this variable include the following: (1) We assume the amount of produce intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{ag}. (2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_{ag}.
I_{beef}; I_{milk}; I_{pork}; $I_{poultry}$; I_{eggs}	Daily intake of COPC from beef, milk, pork, poultry, and eggs	mg/kg-day FW	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-3. The value for this variable will vary for each receptor and each exposure scenario location. Uncertainties associated with this variable include the following: (1) We assume the amount of animal tissue intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{beef}, I_{milk}, I_{pork}, $I_{poultry}$, and I_{eggs}. (2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_{beef}, I_{milk}, I_{pork}, $I_{poultry}$, and I_{eggs}.
I_{fish}	Daily intake of COPC from fish	mg/kg-day FW	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-4. The value for this variable will vary for each water body evaluated. Uncertainties associated with this variable include the following: (1) We assume the amount of fish intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{fish}. (2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_{fish}.

TABLE C-1-6

TOTAL DAILY INTAKE

(Page 3 of 3)

Variable	Description	Units	Value
I_{dw}	Daily intake of COPC from drinking water	mg/kg-day	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-5. The value for this variable will vary for each water body evaluated. Uncertainties associated with this variable include the following: (1) We assume the amount of drinking water intake is constant and representative of the exposed population. This assumption may under- or overestimate I_{dw}. (2) The standard assumptions regarding period exposed may not be representative of an actual exposure situation. This assumption may under- or overestimate I_{dw}.

TABLE C-1-7

INDIVIDUAL CANCER RISK: CARCINOGENS

(Page 1 of 3)

Description

This equation calculates the individual cancer risk from indirect exposure to carcinogenic COPCs. The exposure duration varies for different scenarios. Uncertainties associated with this equation include the following:

(1) Default factors for exposure frequency and exposure duration are assumed to represent the highest exposure that is reasonably expected to occur at a site and, in practice, is estimated by combining upper-bound (90th to 95th percentile) values for these exposure parameters, but not all parameters. This assumption may over- or underestimate the $Cancer\ Risk_i$.

(2) Slope factors are used to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a COPC, and are accompanied by the weight of evidence classification to indicate the strength of the evidence that the agent is a human carcinogen. This classification potentially over- or underestimates $Cancer\ Risk_i$.

(3) Risk at low exposure levels is difficult to measure directly, either by animal experiments or by epidemiological studies. The development of a cancer slope factor generally entails applying a model to the available data set and using the model to extrapolate from the relatively high doses administered to experimental animals (or the exposures noted in epidemiological studies) to lower exposure levels expected for human contact in the environment. This approach may under- or overestimate *Oral CSF*.

Equation

$$Cancer\ Risk_i = \frac{I \cdot ED \cdot EF \cdot CSF}{AT \cdot 365}$$

Variable	Description	Units	Value
$Cancer\ Risk_i$	Individual lifetime cancer risk through indirect exposure to COPC carcinogen *i*	unitless	
I_i	Daily intake of COPC *i* from animal tissue *j*	mg COPC/kg BW-day	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-6. The value for this variable will vary for each exposure pathway and each exposure scenario location. The following uncertainty is associated with this variable: This variable is COPC- and site-specific. See the equation in Table C-1-6 regarding the calculation of and uncertainties associated with this variable.

TABLE C-1-7

INDIVIDUAL CANCER RISK: CARCINOGENS

(Page 2 of 3)

<table>
<tr><th>Variable</th><th>Description</th><th>Units</th><th>Value</th></tr>
<tr><td><i>ED</i></td><td>Exposure duration</td><td>yr</td><td>6, 30, or 40
This variable is exposure scenario-specific:
<table><tr><th>Exposure Scenario</th><th>ED</th></tr><tr><td>Farmer</td><td>40 (U.S. EPA 1994)</td></tr><tr><td>Farmer Child</td><td>6 (U.S. EPA 1989)</td></tr><tr><td>Adult Resident</td><td>30 (U.S. EPA 1989)</td></tr><tr><td>Child Resident</td><td>6 (U.S. EPA 1989)</td></tr><tr><td>Fisher</td><td>30 (U.S. EPA 1994)</td></tr><tr><td>Fisher Child</td><td>6 (U.S. EPA 1989)</td></tr></table>The following uncertainty is associated with this variable:
This exposure duration is a single value that represents the highest exposure that is reasonably expected to occur at a site. This assumption may overestimate <i>ED</i>.</td></tr>
<tr><td><i>EF</i></td><td>Exposure frequency</td><td>days/yr</td><td>350
This variable is site-specific. We recommend using this default value in the absence of site-specific information, consistent with U.S. EPA (1991).

The following uncertainty is associated with this variable:
This exposure frequency is a single value that represents the most frequent exposure that is reasonably expected to occur at a site, assuming 2 weeks of vacation or travel. This assumption may overestimate <i>EF</i>.</td></tr>
<tr><td><i>AT</i></td><td>Averaging time</td><td>yr</td><td>70
This variable is site-specific. We recommend using this default value in the absence of site-specific information, consistent with U.S. EPA (1989).

The following uncertainty is associated with this variable:
The recommendation for averaging time may not accurately represent site-specific time; specifically, this single value may under- or overestimate the length of time of exposure.</td></tr>
<tr><td><i>365</i></td><td>Units conversion factor</td><td>day/yr</td><td></td></tr>
</table>

TABLE C-1-7

INDIVIDUAL CANCER RISK: CARCINOGENS

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Variable	Description	Units	Value
Oral CSF	Oral Cancer Slope Factor	$(mg/kg\text{-}day)^{-1}$	**Varies** This variable is COPC-specific. We recommend using the companion database to the HHRAP as an initial source for this parameter. We also recommend checking the recommended hierarchy of sources of health benchmark values (discussed in Appendix A-2) to make sure the database values are still valid, especially if using database values result in risk estimates of concern. Uncertainties associated with this variable include the following: (1) Slope factors are used to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a COPC; and are accompanied by the weight of evidence classification to indicate the strength of the evidence that the agent is a human carcinogen. (2) Risk at low exposure levels is difficult to measure directly either by animal experiments or by epidemiological studies. Developing a cancer slope factor generally entails applying a model to the available data set and using the model to extrapolate from the relatively high doses administered to experimental animals (or the exposures noted in epidemiological studies) to the lower exposure levels expected for human contact in the environment. This approach may under- or overestimate *Oral CSF*.

REFERENCES AND DISCUSSION

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

This document is cited as the reference source document of the exposure duration for adult and child residents. This document is also cited as the reference source document for the averaging time for carcinogens.

U.S. EPA. 1991. *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.6-03. Washington, D.C.

This document is cited as the reference source document of the exposure frequency.

U.S. EPA. 1994. *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.

This document is cited as the reference source document of the exposure duration for the fisher and farmer.

TABLE C-1-8

HAZARD QUOTIENT: NONCARCINOGENS

(Page 1 of 3)

Description

This equation calculates the hazard quotient for indirect exposure to noncarcinogenic COPCs. The following uncertainty is associated with this equation.

> A chronic *RfD* is an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic *RfDs* are specifically developed to be protective for long-term exposure (from 7 years to a lifetime) to a compound. COPC-specific reference doses (*RfD*) are unlikely to underestimate a chemical potential for causing adverse effects.

Equation

$$HQ = \frac{I \cdot ED \cdot EF}{RfD \cdot AT \cdot 365}$$

Variable	Description	Units	Value
HQ	Hazard quotient	unitless	
I_i	Daily intake of COPC *i* from animal tissue *j*	mg COPC/ kg-day	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-6. The value for this variable will vary for each exposure pathway and each exposure scenario location. Uncertainties associated with this variable are site-specific.
ED	Exposure duration	yr	**6, 30, or 40** Consistent with U.S. EPA (1994b) and NC DEHNR (1997), we recommend using the following default values: Exposure Scenario — *ED* Farmer — 40 (U.S. EPA 1994a) Farmer Child — 6 (U.S. EPA 1989) Adult Resident — 30 (U.S. EPA 1989) Child Resident — 6 (U.S. EPA 1989) Fisher — 30 (U.S. EPA 1994a) Fisher Child — 6 (U.S. EPA 1989) Uncertainty associated with this variable includes: These exposure durations are single values that represent the highest exposure that is reasonably expected to occur at a site. These values may overestimate *ED* for some individuals.

TABLE C-1-8

HAZARD QUOTIENT: NONCARCINOGENS

(Page 2 of 3)

Variable	Description	Units	Value
EF	Exposure frequency	days/yr	**350** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This value is based on U.S. EPA (1991) and is consistent with U.S. EPA (1994b). Uncertainty associated with this variable includes: This exposure frequency is a single value that represents the most frequent exposure that is reasonably expected to occur at a site with two weeks of vacation or travel. This recommended value may overestimate *EF* for individuals who are away from their home for more than two weeks each year. On the other hand, some individuals (such as farmers) may remain at their home (or farm) for more than 350 days per year. In either case, the degree of over- or underestimation is not expected to be significant in most cases.
RfD	Reference Dose	mg/kg-day	**Varies** This variable is COPC-specific. We recommend using the companion database to the HHRAP as an initial source for this parameter. We also recommend checking the recommended hierarchy of sources of health benchmark values (discussed in Appendix A-2) to make sure the database values are still valid, especially if using database values result in risk estimates of concern. The following uncertainty is associated with this variable: A chronic *RfD* is an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic *RfDs* are specifically developed to be protective for long-term exposure (from 7 years to a lifetime) to a compound. COPC-specific *RfDs* are unlikely to underestimate a COPC's potential for causing adverse health effects.
365	Units conversion factor	day/yr	
AT	Averaging time	yr	**6, 30, or 40** This variable is site-specific and related to *ED*. Specifically, the *AT* for noncarcinogens is numerically the same as ED. This default value is consistent with U.S. EPA (1989), U.S. EPA (1991), and U.S. EPA (1994a). Uncertainty associated with this variable includes: Our recommended averaging time may not accurately represent site-specific time; specifically this single value may under- or overestimate the length of an average adult lifetime.

TABLE C-1-8

HAZARD QUOTIENT: NONCARCINOGENS

(Page 3 of 3)

REFERENCES AND DISCUSSION

NC DEHNR (1997). *Draft North Carolina Protocol for Performing Indirect Exposure Risk Assessments for Hazardous Waste Exposure Risk Assessments for Hazardous Waste Combustion Units*. January.

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

This document is cited as the reference source document of the exposure duration for adult and child residents. U.S. EPA OSW assumes that the recommended exposure duration for the child resident may also reasonably be applied to the farmer child and to the fisher child.

U.S. EPA. 1991. *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.6-03. Washington, D.C.

This document is cited as a source document for exposure frequency and averaging time.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-like Components - Volume III: Site-Specific Assessment Procedure*. Review Draft. Office of Research and Development. Washington D.C. EPA/600/6-88/005Cc. June.

This document is cited by U.S. EPA (1994b) as the same document for the recommended default exposure duration (*ED*) values for the farmer and fisher. The *ED* value of 40 years recommended for both the farmer and the fisher is based on the assumption that “farmers live in one location longer than the general population”.

U.S. EPA. 1994b. *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.

This document recommends the following:

- An exposure frequency of 350 days per year
- Receptor-specific exposure duration values as presented in U.S. EPA (1994a)— fisher (40 years) and farmer (40 years) and U.S. EPA (1989)—adult resident (30 years) and child resident (6 years)
- Adult and child body weights of 70 kg and 15 kg, respectively

TABLE C-1-9

TOTAL CANCER RISK: CARCINOGENS

(Page 1 of 1)

Description

For carcinogens, cancer risks are added across all carcinogenic COPCs. See Appendix A for identification of carcinogens. Uncertainty associated with this equation includes the following:

> *Total Cancer Risk* assumes that different carcinogens affect the same target organ to produce a cancer response, ignoring potential antagonistic or synergistic effects or disparate effects on different target organs.

Equation

$$Total\ Cancer\ Risk = \sum_i Cancer\ Risk_i$$

Variable	Description	Units	Value
Total Cancer Risk	Individual lifetime cancer risk through indirect exposure to all COPC carcinogens	unitless	
$Cancer\ Risk_i$	Individual lifetime cancer risk through indirect exposure to COPC carcinogen *i*	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-7. The value for this variable will vary for each exposure pathway. Uncertainties associated with this variable include the following: (1) We assume that the default values for exposure frequency and exposure duration represent the highest exposure that is reasonably expected to occur at a site. In practice, intakes are estimated by combining upper-bound (90th to 95th percentile) values for these exposure variables, but not for other parameters. This assumption is likely to overestimate intakes and the $Cancer\ Risk_i$. (2) Slope factors are used to estimate an upper-bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a potential carcinogen; and are accompanied by the weight of evidence classification to indicate the strength of the evidence that the agent is a human carcinogen. This classification potentially over- or underestimates risk. (3) Risk at low exposure levels is difficult to measure directly either by animal experiments or by epidemiological studies. Developing a cancer slope factor generally entails applying a model to the available data set and using the model to extrapolate from the relatively high doses administered to experimental animals (or the exposures noted in epidemiological studies) to lower exposure levels expected for human contact in the environment. This approach is likely to overestimate *CSF*.

TABLE C-1-10

TOTAL HAZARD INDEX: NONCARCINOGENS

(Page 1 of 1)

Description

For non-cancer health effects, hazard quotient for all COPCs, regardless of target organs, are summed to calculate a total *hazard index*. Uncertainties associated with this equation include the following:

(1) Assuming that different COPCs affect the same target organ to produce an adverse health effect, while ignoring potential antagonistic or synergistic effects or disparate effects on different target organs, may overestimate the total hazard index.
(2) Total hazard index assumes that a single individual in the exposure scenario is exposed to site-related contaminants at estimated exposure concentrations by all pathways that make up the scenario. It is unlikely, however, that a single individual will be exposed by each pathway in the exposure media. This assumption may overestimate the total hazard index.

Equation

$$Total\ Hazard\ Index = \Sigma_j\ HI_j$$

$$HI = \Sigma_i\ HQ_i$$

Variable	Description	Units	Value
Total Hazard Index	Total individual hazard index for all COPCs across all exposure pathways	unitless	
HI_j	Hazard Index for exposure pathway *j*	unitless	**Varies** This variable is COPC- and site-specific. The value for this variable will vary for each exposure pathway. Uncertainties associated with this variable are site-specific.
HQ_i	Hazard Quotient for COPC *i*	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-8. The value for this variable will vary for each exposure pathway. Uncertainties associated with this variable are site-specific.

TABLE C-1-11

SEGREGATED HAZARD INDEX FOR SPECIFIC ORGAN EFFECTS: NONCARCINOGENS

(Page 1 of 1)

Description

For non-cancer health effects, hazard quotients are added across COPCs when they target the same organ to calculate a segregated *hazard index*. See Appendix A-2 for identification of noncarcinogens and their associated target organ. Since segregation by critical effect requires the identification of all major effects, information in Appendix A-2 may not always represent the most current and complete information on COPC-specific major effects. Therefore, Appendix A-2 may require supplemental information about COPC-specific major effects. Uncertainties associated with this equation include the following:

(1) Target organ segregation is dependent upon the critical effect. Segregation by critical effect requires the identification of all major effects, not just those seen at higher doses. The segregation process may over- or underestimate the *hazard index*.

Equation

$$HI_j = \sum_i HQ_i$$

Variable	Description	Units	Value
HI_j	Hazard index for exposure pathway *j*	unitless	
HQ_i	Hazard quotient for COPC *i*	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-1-8. The value for this variable will vary for each exposure pathway. The following uncertainty is associated with this variable: (1) We assume that the default values for exposure frequency and exposure duration represent the highest exposure that is reasonably expected to occur as a site. In practice, intakes are estimated by combining upper-bound (90th to 95th percentile) values for these exposure variables, but not for other parameters. This equation is likely to overestimate intakes and HI_j. (2) Adverse health effects at low exposure levels are difficult to either directly either by animal experiments or by epidemiological studies. The development of *RfD*s generally entails applying uncertainty factors to extrapolate from the results of studies using high exposure doses to lower exposure doses expected for human contact in the environment. This approach is unlikely to underestimate and likely overestimate HI_j. The uncertainties associated with this variable are COPC- and site-specific and will vary for each exposure pathway.

TABLE C-2-1

INHALATION CANCER RISK FOR INDIVIDUAL CHEMICALS: CARCINOGENS

(Page 1 of 4)

Description

This equation calculates the excess lifetime individual cancer risk from the average daily intake via inhalation of a COPC carcinogen. Uncertainties associated with this equation include:

(1) COPC-specific *URF* values are unlikely to underestimate, and may overestimate, the carcinogenic potential of a COPC because of the choice of mathematical models and the use of uncertainty factors in the estimation of these values.
(2) The uncertainty associated with the variable C_a are largely site-specific.

Equation

$$EC = \frac{C_a \cdot EF \cdot ED}{AT \cdot 365\ days/year}$$

Variable	Description	Units	Value
$Cancer\ Risk_{inh(i)}$	Individual lifetime cancer risk through direct inhalation of COPC carcinogen *i*	unitless	
EC	Exposure concentration	$\mu g/m^3$	

TABLE C-2-1

INHALATION CANCER RISK FOR INDIVIDUAL CHEMICALS: CARCINOGENS

(Page 2 of 4)

Variable	Description	Units	Value
URF	Inhalation Unit Risk Factor	$(\mu g/m^3)^{-1}$	**Varies** This variable is COPC-specific. We recommend using the companion database to the HHRAP as an initial source for this parameter. We also recommend checking the recommended hierarchy of sources of health benchmark values (discussed in Appendix A-2) if using database values results in risk estimates of concern. The following general uncertainty is associated with this variable: COPC-specific inhalation unit risk factors (*URF*s) are generally estimated by fitting the results of studies conducted on laboratory animals with a mathematical model. The model generally recommended by U.S. EPA is the linearized multistage (LMS) model. U.S. EPA's position on assessing carcinogenic potential was updated (U.S. EPA 1996b) and should be considered along with recommended toxicological benchmarks and information (U.S. EPA 1995; 1996a). The LMS model assumes that there is no "safe dose" or threshold below which a COPC causing cancer at higher doses will no longer cause cancer in expected individuals. In other words, any exposure to a carcinogen may, through a series of stages, cause cancer in an exposed individual. Also, a series of uncertainty factors are applied to the results before the results are fitted with the LMS model. Applying uncertainty factors follows the underlying assumption that humans are, or may be, as sensitive or more sensitive to the carcinogenic effects of COPCs than the laboratory animals that were tested. As a result of the choice of models and the use of uncertainty factors, COPC-specific *URF*s are unlikely to underestimate a COPC's potential for causing cancer.
C_a	Total COPC air concentration	$\mu g/m^3$	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-5-1. Uncertainty associated with this variable includes: Calculated assuming a default S_T value for background plus local sources, rather than a S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than the S_T value for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.

TABLE C-2-1

INHALATION CANCER RISK FOR INDIVIDUAL CHEMICALS: CARCINOGENS

(Page 3 of 4)

Variable	Description	Units	Value
EF	Exposure frequency	days/yr	**350** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This value is based on U.S. EPA (1991) and is consistent with U.S. EPA (1994b). Uncertainties associated with this variable include: This exposure frequency is a single value that represents the most frequent exposure that is reasonably expected to occur at a site, with two weeks of vacation. This recommended value may overestimate EF for individuals who are away from their home for more than two weeks each year. On the other hand, individuals such as subsistence farmers may remain at their home (or farm) for more than 350 days per year. In either case, we don't expect a significant degree of over- or underestimation in most cases.
ED	Exposure duration	yr	**6, 30, or 40** We recommend reasonable maximum exposure (RME) values for T_2: Exposure Duration — RME — Reference Child Resident, Farmer Child, Fisher Child — 6 years — U.S. EPA (1997b) Adult Resident and Fisher — 30 years — U.S. EPA (1997b) Farmer — 40 years — U.S. EPA (1994b) U.S. EPA (1994c) recommended the following unreferenced values: Exposure Duration — Years Subsistence Farmer — 40 Adult Resident — 30 Subsistence Fisher — 30 Child Resident — 9 Uncertainties associated with this variable include the following: (1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate *Cs* and Cs_{tD}. (2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources. However, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate *Cs* and Cs_{tD}.

TABLE C-2-1

INHALATION CANCER RISK FOR INDIVIDUAL CHEMICALS: CARCINOGENS

(Page 4 of 4)

Variable	Description	Units	Value
AT	Averaging time	yr	**70** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This default value is consistent with U.S. EPA (1989), U.S. EPA (1991), and U.S. EPA (1994b). Uncertainties associated with this variable include: The recommendation for averaging time may not accurately represent site-specific time; specifically this single value may under- or overestimate the length of an average adult lifetime.

REFERENCES AND DISCUSSION

U.S. EPA. 1995. *Health Effects Assessment Summary Tables. Annual Update.* OHEA-ECAO-CIN-909. Environmental Criteria and Assessment Office, Office of Research and Development Cincinnati, Ohio.

This document represent U.S. EPA's secondary source of *Inhalation CSF* values.

U.S. EPA. 1996a. "Integrated Risk Information System (IRIS)". *Database on Toxicity Information Network (TOXNET).*

This reference represents U.S. EPA's primary source of *Inhalation CSF* values and other toxicity factors. This reference is updated periodically and should be reviewed prior to preparing a risk assessment.

U.S. EPA. 1996b. "Proposed Guidelines for Carcinogenic Risk Assessment." *Federal Register.* 61 FR 31667. Volume 61. Number 120. June 20.

This document proposes new guidelines for assessing the carcinogenicity of COPCs.

TABLE C-2-2

INHALATION HAZARD QUOTIENT FOR COPCS: NONCARCINOGENS

(Page 1 of 3)

Description

This equation calculates the *HQ* for inhalation exposures to COPCs that have noncancer health effects. Uncertainties associated with this equation include the following:

(1) COPC-specific reference concentrations (*RfC*) are unlikely to underestimate a COPC's potential for causing adverse health effects.
(2) Most of the uncertainties associated with the variables in the equation in Table B-5-1 (used to calculate C_a), specifically those associated with the variables *Q, Cyv*, and *Cyp*, are site-specific.

Equation

$$EC = \frac{C_a \cdot EF \cdot ED}{AT \cdot 365\ days/year}$$

Variable	Description	Units	Value
$HQ_{inh(i)}$	Hazard quotient for direct inhalation of COPC noncarcinogen *i*	unitless	
EC	Exposure concentration	µg/m^3	
0.001	Units conversion factor	mg/µg	

TABLE C-2-2

INHALATION HAZARD QUOTIENT FOR COPCS: NONCARCINOGENS

(Page 2 of 3)

Variable	Description	Units	Value
RfC	Reference concentration	mg/m^3	**Varies** This variable is COPC-specific, and values can be found in the HHRAP companion database. The following uncertainty is associated with this variable: COPC *RfC*s are generally estimated by applying a series of uncertainty factors to the results of studies conducted on laboratory animals. The application of uncertainty factors follows the underlying assumption that humans are, or may be, as sensitive or more sensitive to the harmful effects of COPCs than the laboratory animals that were tested. *RfC*s are designed to ensure that the general public, including sensitive subpopulations, will not experience adverse health effects as a result of exposure to a COPC at its *RfC*. As a result, COPC-specific *RfC*s are unlikely to underestimate a COPC's potential for causing adverse health effects.
C_a	Total COPC air concentration	$\mu g/m^3$	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-5-1.
EF	Exposure frequency	days/yr	**350** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This value is based on U.S. EPA (1991) and is consistent with U.S. EPA (1994b). Uncertainties associated with this variable include: This exposure frequency is a single value that represents the most frequent exposure that is reasonably expected to occur at a site, with two weeks of vacation. This recommended value may overestimate EF for individuals who are away from their home for more than two weeks each year. On the other had, individuals such as subsistence farmers may remain at their home (or farm) for more than 350 days per year. In either case, we don't expect a significant degree of over- or underestimation in most cases.

TABLE C-2-2

INHALATION HAZARD QUOTIENT FOR COPCS: NONCARCINOGENS

(Page 3 of 3)

Variable	Description	Units	Value
ED	Exposure duration	yr	**6, 30, or 40** We recommend reasonable maximum exposure (RME) values for T_2: Exposure Duration / RME / Reference Child Resident, Farmer Child, Fisher Child — 6 years — U.S. EPA (1997b) Adult Resident and Fisher — 30 years — U.S. EPA (1997b) Farmer — 40 years — U.S. EPA (1994b) U.S. EPA (1994c) recommended the following unreferenced values: Exposure Duration / Years Subsistence Farmer — 40 Adult Resident — 30 Subsistence Fisher — 30 Child Resident — 9 Uncertainties associated with this variable include the following: (1) Exposure duration rates are based on historical mobility rates and may not remain constant. This assumption may overestimate or underestimate *Cs* and Cs_{tD}. (2) Mobility studies indicate that most receptors that move remain in the vicinity of the emission sources. However, it is impossible to accurately predict the likelihood that these short-distance moves will influence exposure, based on factors such as atmospheric transport of pollutants. This assumption may overestimate or underestimate *Cs* and Cs_{tD}.
AT	Averaging time	yr	**6, 30, or 40** This variable is site-specific and related to *ED*. Specifically, the *AT* for noncarcinogens is numerically the same as ED. This default value is consistent with U.S. EPA (1989; 1991; 1994a). Uncertainty associated with this variable includes: Our recommended averaging time may not accurately represent site-specific time; specifically this single value may under- or overestimate the length of an average adult lifetime.

TABLE C-2-3

TOTAL INHALATION CANCER RISK: CARCINOGENS

(Page 1 of 1)

Description

Cancer risk to the individual via inhalation are added across all COPCs that are carcinogenic via the direct inhalation route of exposure.

Uncertainties associated with this equation include the following:

(1) *Total Cancer Risk* assumes that different carcinogens affect the same target organ to produce a cancer response, ignoring potential antagonistic or synergistic effects or disparate effects on different target organs. This assumption may overestimate *Total Cancer Risk*.

(2) The summation of cancer risks across multiple COPCs means that the uncertainties associated with estimating cancer risk for each COPC are also summed. This means *Total Cancer Risk*, as defined below, is unlikely to be overestimated.

Equation

$$Total\ Cancer\ Risk_{inh} = \sum_{i} Cancer\ Risk_{inh(i)}$$

Variable	Description	Units	Value
$Total\ Cancer\ Risk_{inh}$	Total individual lifetime cancer risk through direct inhalation of all COPC carcinogens	unitless	
$Cancer\ Risk_{inh(i)}$	Individual lifetime cancer risk through direct inhalation for COPC carcinogen *i*	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-2-1. Uncertainties associated with this variable include the following: (1) COPC-specific *URF* values are unlikely to underestimate, and may overestimate, the carcinogenic potential of COPCs because of the mathematical models and the use of uncertainty factors in the estimation of these values. (2) Most of the uncertainties associated with the variables used to calculate C_a, specifically *Q, Cyv*, and *Cyp*, are site-specific.

TABLE C-2-4

HAZARD INDEX FOR INHALATION: NONCARCINOGENS

(Page 1 of 1)

Description

For non-cancer health effects, *HQs* for inhalation exposures are added across COPCs when they target the same organ to obtain an *HI* for the target organ. See Appendix A-2 for target organs and Appendix A-3 for COPC-specific inhalation *RfC*s and for identification of COPCs that cause noncarcinogenic effects via the inhalation route of exposure and their associated target organs. Uncertainties associated with this equation include the following:

(1) The summation of noncarcinogenic hazards across multiple COPCs means that the uncertainties associated with estimating hazards for each COPC (see *HQ* below) are also summed. This means that the total noncarcinogenic hazard, as defined below, is unlikely to be overestimated.
(2) As defined below, the *HI* sums the *HQ*s for all COPCs to which a receptor is potentially exposed. Ideally, *HQs* should be summed only for COPCs that affect the same target organs and systems. To the extent that COPCs affect different target organs, summing their associated *HQs* will overestimate the actual *HI*.

Equation

$$HI_{inh} = \sum_{i} HQ_{i}$$

Variable	Description	Units	Value
$HI_{inh(j)}$	Hazard index for target organ effect *j* through direct inhalation of all COPCs	unitless	
$HQ_{inh(i)}$	Hazard quotient for direct inhalation of COPC *i*	unitless	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-2-2. Uncertainties associated with this variable include the following: (1) COPC-specific *RfCs* are unlikely to underestimate a COPC's potential for causing adverse health effects. (2) Most of the uncertainties associated with the variables used to calculate C_a, specifically *Q, Cyv*, and *Cyp*, are site-specific.

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 1 of 7)

Description

This equation calculates the concentration of dioxins and dioxin-like PCBs in milkfat of breast milk. Uncertainties associated with this equation include the following:

(1) The most significant uncertainties associated with this equation are those associated with the variable *m*. Because *m* is calculated as the sum of numerous potential intakes, estimates of *m* incorporate uncertainties associated with each exposure pathway. Therefore, *m* may be under- or overestimated. Every effort should be made to limit and characterize the uncertainties associated with this variable.

(2) This equation assumes that the concentration of dioxinand dioxin-like PCBs in breast milkfat is the same as in maternal fat. To the extent that this is not the case, uncertainty is introduced.

Equation

$$C_{milkfat} = \frac{m \cdot 1{\times}10^9 \cdot h \cdot f_1}{0.693 \cdot f_2}$$

The average maternal intake of dioxin for each adult exposure scenario (*m*) is equal to the total daily intake of dioxin and dioxin-like PCBs (*I*) from inhalation [calculated as the average daily COPC intake via inhalation (*ADI*) using the equation provided below] and all indirect pathways [which is calculated using the equation in Table C-1-6 and based on the highest annual average soil concentration (Cs_{td})] for each adult exposure scenario concentration.

$$ADI = \frac{C_a \cdot IR \cdot ET \cdot EF \cdot ED \cdot 0.001 \ mg/\mu g}{BW \cdot AT \cdot 365 \ day/yr}$$

Variable	Description	Units	Value
$C_{milkfat}$	Concentration of dioxin and dioxin-like PCBs in milk fat of breast milk for a specific exposure scenario	pg COPC/kg milk fat	

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 2 of 7)

Variable	Description	Units	Value
m	Average maternal intake of dioxin and dioxin-like PCBs for each adult exposure scenario	mg COPC/kg BW-day	**Varies** This variable is COPC- and site-specific and is equal to the total daily intake of dioxin and dioxin-like PCBs (*I*) from inhalation [calculated as the average daily COPC intake via inhalation (ADI) using the equation provided above] and all indirect pathways [which is calculated using the equation in Table C-1-6 and based on the highest annual average soil concentration (Cs_{td})] for each adult exposure scenario concentration. The following uncertainty is associated with this variable: (1) The uncertainty associated with this variable may be significant, because this uncertainty represents the sum of all uncertainties associated with each of the potential exposure pathways. To gauge the potential magnitude of the uncertainty associated with this variable, compare estimated *m* values to values reported in the literature.
1×10^9	Units conversion factor	pg/mg	
h	Half-life of dioxin in adults	days	**2,555** This variable is COPC- and site-specific. We recommend using this default value, consistent with U.S. EPA (1994a) and U.S. EPA (1994b). The following uncertainty is associated with this variable: As discussed in U.S. EPA (1994a), the half-life may vary from about 5 to 7 years for 2,3,7,8-TCDD. Using the upper end of the range is conservative. Based on the work of Schecter (1991), and Schlatter (1991), as discussed in U.S. EPA (1994a), the value of *h* may vary by almost one order of magnitude (1.1 to 50) for different dioxin and furan congeners around the value of 7 proposed for 2,3,7,8-TCDD. The differences are largely the result of differences in absorption. However, if the average material intake of dioxin is calculated in terms of *TEQ*s, we assume using a single *h* value based on 2,3,7,8-TCDD is reasonable.
f_1	Fraction of ingested dioxin and dioxin-like PCBs that is stored in fat	unitless	**0.9** This variable is COPC- and site-specific. We recommend using this default value, consistent with U.S. EPA (1994b). The source of this value is U.S. EPA (1994a).
f_2	Fraction of mother's weight that is fat	unitless	**0.3** This variable is COPC-specific. We recommend using this default value, consistent with U.S. EPA (1994a) and U.S. EPA (1994b). The source of this value is U.S. EPA (1994a). The following uncertainty is associated with this variable: Although this single value clearly does not adequately represent all potentially exposed women of childbearing age, we assume the average uncertainty associated with this value to be minimal.

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 3 of 7)

Variable	Description	Units	Value
ADI	Average daily COPC intake via inhalation	mg COPC/ kg-day	
C_a	Total COPC air concentration	μg/m^3	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table B-5-1. Uncertainty associated with this variable includes: Calculated assuming a default S_T value for background plus local sources, rather than a S_T value for urban sources. If a specific site is located in an urban area, using the latter S_T value may be more appropriate. Specifically, the S_T value for urban sources is about one order of magnitude greater than the S_T value for background plus local sources and would result in a lower calculated F_v value; however, the F_v value is likely to be only a few percent lower.
IR	Inhalation rate	m^3/hr	**adult: 0.83** We recommend using default values of 0.83 in the absence of site-specific information. The recommended adult value is consistent with values used in U.S. EPA (1991; 1994b; 1996a), and consistent with average of values provided in U.S. EPA (1997) when considering adult age brackets for both male and female inhalation rates for indoor and outdoor activities in a residential setting. Uncertainty associated with this variable includes: The recommended inhalation rates do not consider individual respiratory or activity differences. Therefore, based on the individual and the activities that individual is engaged in, the recommended inhalation rates may under-or overestimate the actual rates. However, we don't expect the degree of under-or overestimation to be significant.
ET	Exposure time	hrs/day	**24** This variable is site-specific. We recommend using this default value in the absence of site-specific data. Uncertainty associated with this variable includes: The recommended *ET* value assumes that an individual remains at a specific location 24 hours per day. In reality this is likely to be true only for a minority of the population including young children, their caregivers, and elderly or individuals who are sick. Therefore, the recommended value contributes to a degree of overestimation for much of the population. However, it must be noted that though an individual may not always be at a single location, that individual may continue to be exposed to emissions at an alternate location.

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 4 of 7)

Variable	Description	Units	Value
EF	Exposure frequency	days/yr	**350** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This value is based on U.S. EPA (1991) and is consistent with U.S. EPA (1994b). Uncertainties associated with this variable include: This exposure frequency is a single value that represents the most frequent exposure that is reasonably expected to occur at a site with two weeks of vacation. This recommended value may overestimate *EF* for individuals who are away from their home for more than two weeks each year. On the other hand, some individuals (such as farmers) may remain at their home (or farm) for more than 350 days per year. In either case, we don't expect the degree of over- or underestimation to be significant in most cases.
ED	Exposure duration	yr	**30, or 40** This variable is site-specific. Consistent with U.S. EPA (1994b), we recommend using the following default values: Exposure Scenario — *ED* Farmer — 40 (U.S. EPA 1994a) Adult Resident — 30 (U.S. EPA 1989) Fisher — 30 (U.S. EPA 1994a) Uncertainties associated with this variable include: These exposure durations are single values that represent the highest exposure that is reasonably expected to occur at a site. These values may overestimate *ED* for some individuals.
BW	Body weight	kg	**70** This variable is site-specific. We recommend using default value of 70 (adults) in the absence of site-specific information. These default values are consistent with U.S. EPA (1991; 1994b). Uncertainties associated with this variable include: This body weight represents the average weight of an adult. However, depending on the site, the body weights may be higher or lower. This default value may overestimate or underestimate actual body weights. However, we don't expect the degree of under- or overestimation to be significant.

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 5 of 7)

Variable	Description	Units	Value
AT	Averaging time	yr	**70** This variable is site-specific. We recommend using this default value in the absence of site-specific data. This default value is consistent with U.S. EPA (1989), U.S. EPA (1991), and U.S. EPA (1994b). Uncertainties associated with this variable include: The recommended averaging time may not accurately represent site-specific time; specifically this single value may under- or overestimate the length of an average adult lifetime.
0.001	Units conversion factor	mg/µg	
365	Units conversion factor	days/yr	

REFERENCES AND DISCUSSION

Schecter, A. 1991. "Dioxins and Related Chemicals in Humans and in the Environment." In: *Biological Basis for Risk Assessment of Dioxins and Related Compounds*: Gallo, M.; Schenplein, R; Van Der Heijden, K. Eds; Banbury Report 35, Cold Spring Harbor Laboratory Press.

This document is cited by U.S. EPA (1994a) as the source of information related to the metabolism of dioxin and related compounds, in addition to concentrations of various congeners in adipose tissue.

Schlatter, C., 1991. "Data on Kinetics of PCDDs and PCDFs as a Prerequisite for Human Risk Assessment." In: *Biological Basis for Risk Assessment of Dioxins and Related Compounds*; Gallo, M; Schenplein, R; Van Der Heijder, K., eds. Banbury Report 35, Cold Spring Harbor Laboratory press.

This document is cited by U.S. EPA (1994a) as a source of a method of estimating the half-life of dioxin-related compounds, based on uptake data relative to 2,3,7,8-TCDD. U.S. EPA (1994a) proposed the following equation, based on this document:

$$C_{TCDD} = \frac{D_{TCDD} \cdot t_{1/2},\ TCDD \cdot V}{\ln\ 2}$$

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

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where
C_{TCDD} = Concentration of TCDD in body
D_{TCDD} = Daily intake of TCDD
$t_{1/2}, TCDD$ = Half-life of TCDD in body
V = Volume of body compartment

Smith, A.H. 1987. "Infant Exposure Assessment for Breast Milk Dioxins and Furans Derived from Waste Incineration Emissions." *Risk Analysis.* 7(3) 347-353.

This document is cited by U.S. EPA (1994a) as the source of the equation in Table C-3-1 and the recommended values for h (2,555 days), f_1 (0.9), and f_2 (0.3). This document assumes that the concentration of dioxins in breast milkfat is the same as in maternal fat.

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

This document is cited as the reference source document of the exposure duration for adult residents. This document is also cited as reference source document for the averaging time for carcinogens.

U.S. EPA. 1991. *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.* Office of Solid Waste and Emergency Response. OSWER Directive 9285.6-03. Washington, D.C. March 21.

This document is cited as the reference source document of the exposure frequency, body weight, and adult inhalation rate variables.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Volume II: Properties, Sources, Occurrence, and Background Exposures. Review Draft.* Office of Research and Development. EPA/600/6-88/0055Cb. Washington, D.C. June.

This document cites Smith (1987) as the source for half of the recommended values for the life of dioxin for adults (h), proportion of ingested dioxin that is stored in fat (f_1), and proportion of mother's milk that is fat (f_2).

This document is cited by U.S. EPA (1994b) as the same document for the recommended default exposure duration (*ED*) values for the farmer and fisher. The *ED* value of 40 years recommended for both the farmer and the fisher is based on the assumption that "farmers live in one location longer than the general population".

TABLE C-3-1

CONCENTRATION OF DIOXINS AND DIOXIN-LIKE PCBs IN BREAST MILK

(Page 7 of 7)

U.S. EPA. 1994b. *Revised Draft Guidance for Performing Screening Level Risk Analyses at Combustion Facilities Burning Hazardous Waste. Attachment C, Draft Exposure Assessment Guidance for RCRA Hazardous Waste Combustion Facilities*. Office of Emergency and Remedial Response. Office of Solid Waste. December 14.

This document recommends using the equation in Table C-3-1 and values for the variables in this equation: h (2,555 days), f_1 (0.9), and f_2 (0.3).

This document also recommends the following:

- An adult inhalation rate of 20 m^3/day (0.83 m^3/hr)
- An exposure frequency of 350 days per year
- Receptor-specific exposure duration values as presented in U.S. EPA (1994a)—fisher (40 years) and farmer (40 years) and U.S. EPA (1989)—adult resident (30 years)
- Adult body weight of 70 kg
- An averaging time, AT, of 70 years

U.S. EPA. 1997. *Exposure Factors Handbook.* Office of Research and Development. EPA/600/P-95/002F. August.

This document recommends an adult inhalation rate value of 20.5 m^3/day (0.85 m^3/hr) if considering indoor and outdoor activities.

TABLE C-3-2

AVERAGE DAILY DOSE TO THE EXPOSED INFANT

(Page 1 of 3)

Description

This equation calculates the average daily dose for an infant exposed to contaminated breast milk. Uncertainty associated with this equation includes the following:

> The most significant uncertainty associated with this equation is the selection of a value for averaging time (*AT*). As stated in U.S. EPA (1994a), "Little agreement exists regarding the appropriate choice of an averaging time for less than lifetime exposures. This is especially true for cases where exposure is occurring in a particularly sensitive developmental period."
>
> An averaging time (*AT*) of 1 year is appropriate for assessing noncarcinogenic effects. However, using this value may overestimate a lifetime average appropriate for assessing carcinogenic risk by almost two orders of magnitude (70/1).

Equation

$$ADD_{infant} = \frac{C_{milkfat} \cdot f_3 \cdot f_4 \cdot IR_{milk} \cdot ED}{BW_{infant} \cdot AT}$$

Variable	Description	Units	Value
ADD_{infant}	Average daily dose for infant exposed to contaminated breast milk	pg COPC/kg BW-day	
$C_{milkfat}$	Concentration of COPC in milk fat of breast milk for a specific exposure scenario	pg COPC/kg milkfat	**Varies** This variable is COPC- and site-specific, and is calculated using the equation in Table C-3-1. The following uncertainty is associated with this variable: The most significant uncertainties associated with the calculation of this variable are those associated with the variable *m* and the estimate of $C_{milkfat}$. Uncertainties associated with *m* represent a sum of the various uncertainties associated with each of the potential exposure pathways (see the equation in Table C-1-6).
f_3	Fraction of mother's breast milk that is fat	unitless	**0.04** This variable is COPC- and site-specific. We recommend using this default value, consistent with U.S. EPA (1994a; 1994b; 2002). As cited in U.S. EPA (1994a), the source of this variable value is Smith (1987). We assume the uncertainty associated with this value is minimal.

TABLE C-3-2

AVERAGE DAILY DOSE TO THE EXPOSED INFANT

(Page 2 of 3)

Variable	Description	Units	Value
f_4	Fraction of ingested COPC that is absorbed	unitless	**0.9** This variable is COPC- and site-specific. We recommend using this default value, as do U.S. EPA (1994a; 1994b). As cited in U.S. EPA (1994a), the source of this variable value is Smith (1987). We assume the uncertainty associated with this value is minimal.
IR_{milk}	Ingestion rate of breast milk by the infant	kg/day	**0.688** This variable is COPC- and site-specific. We recommend using this default value, from Table 2-10 in U.S. EPA (2002). The following uncertainty is associated with this variable: U.S. EPA (2002) reports a 12-month time weighted average of 688 mL/day. Assuming a density of breast milk of slightly more than 1.0, the recommended value is converted from milliliters per day to kilograms per day. We don’t anticipate significant site-specific variation from this value.
ED	Exposure duration	yr	**1.0** This variable is COPC- and site-specific. We recommend using this default value, as do U.S. EPA (1994a; 1994b). The following uncertainty is associated with this variable: Some infants may nurse for more or less than the recommended 1 year. However, we don’t expect the average uncertainty associated with this variable value to be large.
BW_{infant}	Body weight of infant	kg	**9.4** As cited in U.S. EPA (2002), this value is estimated using using data from NHANES III, which was conducted from 1988 to 1994. The following uncertainty is associated with this variable: As reported in U.S. EPA (2002), NHANES III collected body weight data for approximately 15,000 children between the ages of 2 months and 17 years. U.S. EPA (2002) reported a range of body weights for 7-12 month-old infants of 9.1 to 9.7 kilograms, with a mean body weight of 9.4 kilograms. Based on this information and an assumed 1-year *ED*, we expect the uncertainty associated with this variable value to be minimal.

TABLE C-3-2

AVERAGE DAILY DOSE TO THE EXPOSED INFANT

(Page 3 of 3)

Variable	Description	Units	Value
AT	Averaging time	yr	**1** This variable is COPC- and site-specific. We recommend using this default value, as do U.S. EPA (1994a; 1994b). The following uncertainty is associated with this variable: The uncertainty associated with this variable value is significant, as stated in U.S. EPA (1994a): "Little agreement exists regarding the appropriate choice of an averaging time for less than lifetime exposures. This is especially true for cases where exposure is occurring in a particularly sensitive developmental period." An averaging time of 1 year is appropriate for assessing noncarcinogenic effects. However, using this value may overestimate a lifetime average, appropriate for assessing carcinogenic risk, by almost two orders of magnitude (70/1).

REFERENCES AND DISCUSSION

National Center for Health Statistics. 1987.

Cited in U.S. EPA (1994a) as the source of the recommended BW_{infant} value of 10 kilograms. However, that document does not provide a complete reference for this document.

Smith., A.H. 1987. "Infant Exposure Assessment for Breast Milk Dioxins and Furans Derived from Waste Incineration Emissions." *Risk Analysis*. 7(3) 347-353.

This document is cited by U.S. EPA (1994a) as the source of the recommended values for the variables in the equation in Table C-3-2.

U.S. EPA. 1994a. *Estimating Exposure to Dioxin-Like Compounds. Review Draft.* Office of Research and Development. EPA/600/6-88/0055Cc. Washington ,D.C. June.

This document is cited as the original source of the fraction of fat in breast milk, fraction of ingested COPC that is absorbed, and exposure duration.

U.S. EPA. 1994b. *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.

This document recommends using the equation in Table C-3-2 and values for the variables in this equation: f_3 (0.04), f_4 (0.9), ED (1 year), and AT (1 year).

U.S. EPA. 2002. *Child-Specific Exposure Factors Handbook*. EPA-600-P-00-002B. National Center for Environmental Assessment (Washington Office) and U.S. EPA ORD. September.

This document recommends values for the variables f_3 (0.04), IR_{milk} (0.688 kg/day), and BW_{infant} (9.4 kg).

TABLE C-4-1

ACUTE HAZARD QUOTIENT

(Page 1 of 1)

Description

This equation calculates the acute hazard quotient *AHQ* for short term inhalation exposures to COPCs. Uncertainties associated with this equation include the following:

(1) Uncertainties may be associated with development components of COPC-specific acute inhalation exposure criteria (AIECs), including exposure group protected, exposure duration, and toxicity endpoint. Uncertainties are specific to each COPC's AIEC, and may under or overestimate the potential for causing adverse health effects.

(2) Most of the uncertainties associated with the variables in the equation in Table B-6-1 (used to calculate C_{acute}), specifically those associated with the variables *Q, Chv*, and *Chp*, are site-specific.

Equation

$$AHQ_{inh(i)} = \frac{C_{acute} \cdot 0.001}{AIEC}$$

Variable	Description	Units	Value
$AHQ_{inh(i)}$	Acute hazard quotient for inhalation of COPCs	unitless	
C_{acute}	Acute air concentration	µg/m^3	**Varies** This variable is COPC- and site-specific, and is calculated by using the equation in Table B-6-1.
AIEC	COPC acute inhalation exposure criteria	mg/m^3	**Varies** This variable is COPC-specific (see the companion database to the HHRAP) and determined following a hierarchal approach as discussed in Chapter 7 of the HHRAP. The following uncertainty is associated with this variable: Uncertainties may be associated with development components of COPC-specific acute inhalation exposure criteria (AIECs), including exposure group protected, exposure duration, and toxicity endpoint. Uncertainties are specific to each COPC's AIEC, and may under or overestimate the potential for causing adverse health effects.
0.001	Conversion factor	mg/µg	