United States
Environmental Protection
Agency

Solid Waste and Emergency
Response
(5305W)

EPA530-D-98-001A
July 1998
www.epa.gov/osw

EPA

Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities

Volume One

Peer Review Draft

Printed on paper that contains at least 20 percent postconsumer fiber

EPA 530-D-98-001A
July 1998

Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities

Volume One

U.S. EPA, OFFICE OF SOLID WASTE

U.S. ENVIRONMENTAL PROTECTION AGENCY

DISCLAIMER

This document provides guidance to U.S. EPA Regions and States on how best to implement RCRA and U.S. EPA's regulations to facilitate permitting decisions for hazardous waste combustion facilities. It also provides guidance to the public and to the regulated community on how U.S. EPA intends to exercise its discretion in implementing its regulations. The document does not substitute for U.S. EPA's regulations, nor is it a regulation itself. Thus, it cannot impose legally-binding requirements on U.S. EPA, States, or the regulated community. It may not apply to a particular situation based upon the circumstances. U.S. EPA may change this guidance in the future, as appropriate.

ACKNOWLEDGMENTS

Jeff Yurk (U.S. EPA Region 6) and David Weeks (formerly of U.S. EPA Region 6), the primary authors/editors of this document, would like to acknowledge that the development of this document could not have been accomplished without the support, input, and work of a multitude of U.S. EPA and support contractor personnel. The foundation for the procedures and methodologies outlined in this document were first developed by the Office of Research and Development (ORD) and the Office of Solid Waste (OSW) in previous versions of combustion risk assessment guidance. The State of North Carolinas' combustion risk assessment methodology was also evaluated in preparation of this document. This version of the guidance was originally initiated in response to the desire of the Region 6 Multimedia Planning and Permitting Division to implement an up-to-date and technically sound hazardous waste combustion permitting program. The decision to incorporate guidance on a full range of national combustion risk assessment issues into the document was encouraged and supported by the Director of the Office of Solid Waste.

The development of this document was significantly enhanced by a number of capable organizations and personnel within U.S. EPA. Karen Pollard, Alexander McBride and David Layland of the Economic Methods and Risk Analysis Division in conjunction with Rosemary Workman of the Permits and State Programs Division and Karen Kraus of the Office of General Council provided overall policy, technical and legal comment on this document. David Reisman, Glenn Rice, Eletha Brady Roberts and Matthew Lorber of the National Center for Environmental Assessment (NCEA), Office of Research and Development and Dr. Dorothy Canter, Science Advisor to the Assistant Administrator for the Office of Solid Waste and Emergency Response, provided key input on breaking scientific developments in the areas of mercury speciation, the dioxin reassessment, endocrine disruptors, toxicity factors, sulfur and brominated dioxin analogs, as well as technical comment on the overall methodologies presented in the document.

Contributions by Dr. Larry Johnson of the National Exposure Research Laboratory of ORD and Jeff Ryan of the National Risk Management Research Laboratory of ORD were significant in providing methodologies for conducting TO analysis and defining appropriate detection limits to be used in the risk assessment. Donna Schwede of the National Exposure Research Laboratory of ORD and Jawad Touma of the Office of Air Quality Planning and Standards provided technical review comments to strengthen the air modeling section of the document. Review and comment on the soil and water fate and transport models was provided by Robert Ambrose of EPA's Environmental Research Laboratory in Athens, GA.

All U.S. EPA Regional Offices contributed valuable comments which have significantly improved the usability of this document. In particular, Region 4 aided in making sure guidance for conducting trial burns was consistent with this document, and Region 10 provided significant input on the subject of acute risk assessment and PCB analysis. The authors would be remiss if they did not acknowledge significant contributions from the Texas Natural Resource and Conservation Commission through both comments and discussions of real-world applications of risk assessment methodologies. Additionally, useful comments were received from the States of Colorado, Utah, and Alabama. The Region 6 Superfund Division is to be commended for its valuable review of the early document. Region 6 apologizes and bears full responsibility for any mistakes made in the incorporation of comment and input from all reviewers into the document.

Finally, this work could not have been completed without the tireless efforts of support contractor personnel. Tetra Tech EM Inc. (Tetra Tech), performed the bulk of the background research. The Air Group, under subcontract to Tetra Tech, helped develop the chapter on air dispersion modeling. The work

of these contractors was performed under the technical direction of staff from the Region 6 Center for Combustion Science and Engineering, as well as key Agency project and contracting officers.

Region 6 looks forward to the insight and input yet to be provided by industry and other interested parties during the full external peer review of the document.

CONTENTS

CONTENTS (Continued)

Chapter **Page**

CONTENTS (Continued)

Chapter **Page**

CONTENTS (Continued)

CONTENTS (Continued)

Chapter **Page**

CONTENTS (Continued)

CONTENTS (Continued)

Chapter **Page**

CONTENTS (Continued)

CONTENTS (Continued)

Chapter **Page**

Appendix

FIGURES

TABLES

LIST OF ACRONYMS

μg	Microgram
μm	Micrometer
ACGIH	American Conference of Governmental Industrial Hygienists
ADD	Average daily dose
AEFA	Average Emission Factor Approach
Ah	Aryl hydrocarbon
AHH	Aryl hydrocarbon hydroxylase
AIEC	Acute inhalation exposure criteria
AIHA	American Industrial Hygiene Association
APCS	Air pollution control system
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
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
DEHP	Diethylhexylphthalate
dL	Decaliter
DNA	Dioxyribonucleic acid
DNOP	Di(n)octyl phthalate
DOE	Department of Energy
DRE	Destruction and removal efficiency

LIST OF ACRONYMS (Continued)

DW	Dry weight of soil or plant/animal tissue
EPACA	U.S. Environmental Protection Agency Correlation Approach
EQL	Estimated quantitation limit
ESP	Electrostatic precipitator
ExInter	Expert Interface Version 1.0
FW	Fresh weight (or whole/wet weight) of plant or animal tissue
g	Grams
GC	Gas chromatography
GEP	Good engineering practice
GRAV	Unspeciated gravimetric compounds
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 polystryene microspheres
IRIS	Integrated Risk Information System
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
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

LIST OF ACRONYMS (Continued)

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
NRC	Nuclear Regulatory Commission
NTP	National Toxicology Program
NWS	National Weather Service
OAQPS	Office of Air Quality Planning and Standards
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
PQL	Practicle 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

LIST OF ACRONYMS (Continued)

RME	Reasonable maximum exposure
RPF	Relative potency factor
RTDM	Rough terrain diffusion model
RTDMDEP	Rough terrain diffusion model deposition
s	Second
SAMSON	Solar and Meterological Surface Observational Network
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
TEF	Toxicity equivalent factor
TEQ	Toxicity equivalent quotient
TG	Terrain grid
TIC	Tentatively identified compound
TLV	Threshold limit value
TOC	Total organic carbon
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
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

LIST OF VARIABLES

β_0, β_1 = Regression constants (unitless)
γ = Empirical constant (unitless)
λ_z = Dimensionless viscous sublayer thickness (unitless)
μ_a = Viscosity of air (g/cm-s)
μ_w = Viscosity of water corresponding to water temperature (g/cm-s)
ρ_a = Density of air (g/cm^3 or g/m^3)
ρ_{forage} = Density of forage (g/cm^3)
ρ_s = Bed sediment density (kg/L)
ρ_w = Density of water corresponding to water temperature (g/cm^3)
θ = Temperature correction factor (unitless)
θ_{bs} = Bed sediment porosity (unitless)
θ_{sw} = Soil volumetric water content (mL water/cm^3 soil)

a = Empirical intercept coefficient (unitless)
A = Surface area of contaminated area (m^2)
A_{beef} = Concentration of COPC in beef (mg COPC/kg FW tissue)
$A_{chicken}$ = Concentration of COPC in chicken meat (mg COPC/kg FW tissue)
ADD = Average daily dose (mg COPC/kg BW-day)
ADD_{infant} = Average daily dose for infant exposed to contaminated breast milk (pg COPC/kg BW infant/day)
ADD_{mat} = Average daily dose (mother) (pg COPC/kg BW mother/day)
AEF = Applicable average emission factor for the equipment type (kg/hr-source)
A_{egg} = Concentration of COPC in eggs (mg COPC/kg FW tissue)
Ah = Area planted (m^2)
Ahi = Area planted to *i*th crop (m^2)
A_I = Impervious watershed area receiving COPC deposition (m^2)
A_L = Total watershed area receiving COPC deposition (m^2)
A_{milk} = Concentration of COPC in milk (mg COPC/kg FW tissue)
A_{pork} = Concentration of COPC in pork (mg COPC/kg FW tissue)
AT = Averaging time (days)
A_W = Water body surface area (m^2)

b = Empirical slope coefficient (unitless)
Ba_{beef} = Biotransfer factor for beef (day/kg FW tissue)
$Ba_{chicken}$ = Biotransfer factor for chicken (day/kg FW tissue)
Ba_{eggs} = Biotransfer factor for eggs (day/kg FW tissue)
BAF_{fish} = Bioaccumulation factor for fish (L/kg FW tissue)
Ba_{milk} = Biotransfer factor for milk (day/kg FW tissue)
Ba_{pork} = Biotransfer factor for pork (day/kg FW tissue)
BCF_{fish} = Bioconcentration factor for fish (mg COPC/kg FW tissue)/(mg COPC/kg dissolved water)—unitless
BD = Soil bulk density (g soil/cm^3 soil)
Br_{ag} = Plant-soil bioconcentration factor for aboveground produce
Br_{forage} = Plant-soil bioconcentration factor for forage (µg COPC/g DW plant)/(µg COPC/g soil)—unitless

LIST OF VARIABLES (Continued)

Br_{grain} = Plant-soil bioconcentration factor for COPC in grain (µg COPC/g DW plant)/(µg COPC/g soil)—unitless
$Br_{rootveg}$ = Plant-soil bioconcentration factor for COPC in belowground produce (µg COPC/g FW plant)/(µg COPC/g soil)—unitless
Bs = Soil bioavailability factor (unitless)
$BSAF$ = Biota-to-sediment accumulation factor (mg COPC/kg lipid tissue)/(mg COPC/kg sediment)—unitless
Bv_{ag} = COPC air-to-plant biotransfer factor for aboveground produce (µg COPC/g DW plant)/(µg COPC/g air)—unitless
$Bv_{forage/silage}$ = Air-to-plant biotransfer factor for forage and silage (µg COPC/g DW plant)/(µg COPC/g air)—unitless

C = USLE cover management factor (unitless)
C_a = Total COPC air concentration (µg/m^3)
C_{acute} = Acute air concentration (µg/m^3)
$Cancer\ Risk_i$ = Individual lifetime risk through indirect exposure to COPC carcinogen *i* (unitless)
$Cancer\ Risk_{inh(i)}$ = Individual lifetime cancer risk through direct inhalation of COPC carcinogen *i* (unitless)
C_{BS} = Bed sediment concentration (or sediment bulk density) (g sediment/cm^3 water)
C_{gen} = Generic chemical concentration (mg COPC/kg tissue or media) or (mg/L)
Ccn_k = Stack concentration of non-Table A-1 list *i*th carcinogenic COPCs (carbon basis) (mg COPC/m^3 stack emissions)
Ccp_i = Stack concentration of Table A-1 list *i*th carcinogenic COPCs (carbon basis) (mg COPC/m^3 stack emissions)
C_d = Drag coefficient (unitless)
C_{dw} = Dissolved phase water concentration (mg COPC/L water)
C_{fish} = Concentration of COPC in fish (mg COPC/kg FW tissue)
C_i = Stack concentration *i*th identified COPC (carbon basis) (mg/m^3)
Cn_j = Stack concentration of non-carcinogenic COPC *j* (carbon basis) (mg/m^3)
CR = Generic contact rate (kg/day or L/day)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
C_{sb} = Concentration sorbed to bed sediment (mg COPC/kg sediment)
Cs_{tD} = Soil concentration at time *tD* (mg COPC/kg soil)
C_{TOC} = Stack concentration of TOC, including speciated and unspeciated compounds (mg COPC/m^3 stack emissions)
C_{va} = Gas phase air concentration (µg COPC/m^3 air)
C_{VOC} = Total stack concentration of volatile speciated COPCs with boiling points less than 100°C (mg COPC/m^3 stack emissions)
$C_{VOC}(i)$ = Stack concentration of the *i*th volatile speciated COPC with a boiling point less than 100°C (carbon basis) (mg COPC/m^3 stack emissions)
C_{wctot} = Total COPC concentration in water column (mg COPC/L water column)
C_{wtot} = Total water body COPC concentration including water column and bed sediment (g COPC/m^3 water body) or (mg/L)

LIST OF VARIABLES (Continued)

Cyp = Unitized yearly average air concentration from particle phase (µg-s/g-m^3)
Cyv = Unitized yearly average air concentration from vapor phase (µg-s/g-m^3)
$Cywv$ = Unitized yearly (water body and watershed) average air concentration from vapor phase (µg-s/g-m^3)

D_a = Diffusivity of COPC in air (cm^2/s)
d_{bs} = Depth of upper benthic sediment layer (m)
D_{mean} = Mean particle size density for a particular filter cut size
Ds = Deposition term (mg COPC/kg soil-yr)
d_{wc} = Depth of water column (m)
D_w = Diffusivity of COPC in water (cm^2/s)
$Dydp$ = Unitized yearly average dry deposition from particle 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)
$Dywp$ = Unitized yearly average wet deposition from particle phase (s/m^2-yr)
$Dywv$ = Unitized yearly average wet deposition from vapor phase (s/m^2-yr)
$Dywwv$ = Unitized yearly (water body and watershed) average wet deposition from vapor phase (s/m^2-yr)
d_z = Total water body depth (m)

ED = Exposure duration (yr)
EF = Exposure frequency (days/yr)
ER = Soil enrichment ratio (unitless)
E_v = Average annual evapotranspiration (cm/yr)

f_{bs} = Fraction of total water body COPC concentration in benthic sediment (unitless)
Fd = Fraction of diet that is soil (unitless)
F_i = Fraction of plant type i grown on contaminated soil and eaten by the animal (unitless)
f_{lipid} = Fish lipid content (unitless)
Fw = Fraction of COPC wet deposition that adheres to plant surfaces (unitless)
f_{wc} = Fraction of total water body COPC concentration in the water column (unitless)
F_v = Fraction of COPC air concentration in vapor phase (unitless)

GEF = Applicable emission factor for sources with screening values >10,000 ppmv (kg/hr-source)

H = Henry's Law constant (atm-m^3/mol)
HI = Hazard index (unitless)
HI_j = Hazard index for exposure pathway j (unitless)
HQ = Hazard quotient (unitless)
HQ_i = Hazard quotient for COPC i (unitless)
$HQ_{inh(i)}$ = Hazard quotient for direct inhalation of COPC i (unitless)

LIST OF VARIABLES (Continued)

I = Average annual irrigation (cm/yr)
I_i = Daily intake of COPC (i) from animal tissue j (mg/day)

k = von Karman's constant (unitless)
K = USLE erodibility factor (ton/acre)
k_b = Benthic burial rate constant (yr^{-1})
Kd_{bs} = Bed sediment/sediment pore water partition coefficient (cm^3 water/g bottom sediment)
Kd_{ij} = Partition coefficient for COPC i associated with sorbing material j (unitless)
Kd_s = Soil-water partition coefficient (cm^3 water/g soil)
Kd_{sw} = Suspended sediments/surface water partition coefficient (L water/kg suspended sediment)
K_G = Gas phase transfer coefficient (m/yr)
K_L = Liquid phase transfer coefficient (m/yr)
K_{oc} = Soil organic carbon-water partition coefficient (mL water/g soil)
K_{ocj} = Sorbing material-independent organic carbon partition coefficient for COPC j
K_{ow} = Octanol-water partition coefficient (mg COPC/L octanol)/(mg COPC/L octanol)—unitless
kp = Plant surface loss coefficient (yr^{-1})
ks = COPC soil loss constant due to all processes (yr^{-1})
kse = COPC loss constant due to soil erosion (yr^{-1})
ksg = COPC loss constant due to biotic and abiotic degradation (yr^{-1})
ksl = COPC loss constant due to leaching (yr^{-1})
ksr = COPC loss constant due to surface runoff (yr^{-1})
ksv = COPC loss constant due to volatilization (yr^{-1})
k_v = Water column volatilization rate constant (yr^{-1})
K_v = Overall COPC transfer rate coefficient (m/yr)
k_{wt} = Overall total water body dissipation rate constant (yr^{-1})

L = Monin-Obukhov Length (m)
$LADD$ = Lifetime average daily dose (mg COPC/kg BW-day)
L_{DEP} = Total (wet and dry) particle phase and wet vapor phase COPC direct deposition load to water body (g/yr)
L_{dif} = Vapor phase COPC diffusion (dry deposition) load to water body (g/yr)
leak rate = Emission rate from the individual item of equipment (kg/hr)
L_E = Soil erosion load (g/yr)
LEF = Applicable emission factor for sources with screening values <10,000 ppmv (kg/hr-source)
L_R = Runoff load from pervious surfaces (g/yr)
L_{RI} = Runoff load from impervious surfaces (g/yr)
L_T = Total COPC load to the water body including deposition, runoff, and erosion (g/yr)
LS = USLE length-slope factor (unitless)

LIST OF VARIABLES (Continued)

M_{skin} = Mass of a thin (skin) layer of below ground vegetable (g)
$M_{vegetable}$ = Mass of the entire vegetable (g)
MF = Metabolism factor (unitless)

n = Number of items of equipment of the applicable type in the stream (unitless)
N_{ge} = Equipment count (specific equipment type) for sources with screening values >10,000 ppmv
N_{le} = Equipment count (specific equipment type) for sources with screening values <10,000 ppmv

OC_i = Organic carbon content of sorbing material *i* (unitless)
OC_{sed} = Fraction of organic carbon in bottom sediment (unitless)

p°_L = Liquid phase vapor pressure of chemical (atm)
p°_S = Solid phase vapor pressure of chemical (atm)
P = Average annual precipitation (cm/yr)
PF = USLE supporting practice factor (unitless)
Pd = Aboveground exposed produce concentration due to direct (wet and dry) deposition onto plant surfaces (mg COPC/kg DW)
P_i = Total COPC concentration in plant type *i* ingested by the animal (mg/kg DW)
Pr = Aboveground exposed and protected produce concentration due to root uptake (mg COPC/kg DW)
Pr_{bg} = Belowground produce concentration due to root uptake (mg COPC/kg DW)
Pv = Concentration of COPC in plant due to air-to-plant transfer (mg COPC/kg DW)

Q = COPC emission rate (g/s)
Q_i = Emission rate of COPC *(i)* (g/s)
$Q_{i(adj)}$ = Adjusted emission rate of COPC *(i)* (g/s)
$Qcp_{i(adj)}$ = Adjusted emission rate of Table A-1 carcinogenic COPC *(i)* (g/s)
Qcp_i = Emission rate of Table A-1 carcinogenic COPC *(i)* (g/s)
Q_f = Anthropogenic heat flux (W/m^2)
Qp_i = Quantity of plant type *i* ingested by the animal each day (kg DW/day)
Qs = Quantity of soil ingested by the animal each day (kg/day)
$Q_{VOCi,adj}$ = Adjusted emission rate of the *i*th volatile speciated COPC with a boiling point less than 100°C (g/s)
Q_{VOCi} = Emission rate of the *i*th volatile speciated COPC (g/s)
Q_* = Net radiation absorbed (W/m^2)

r = Interception fraction—the fraction of material in rain intercepted by vegetation and initially retained (unitless)
R = Universal gas constant (atm-m^3/mol-K)

LIST OF VARIABLES (Continued)

RCF = Root concentration factor (µg COPC/g DW plant)/(µg COPC/ml soil water)
RO = Average annual surface runoff from pervious surfaces (cm/yr)
REL = California EPA Air Toxics Hot Spots Program acute reference exposure levels
RF = USLE rainfall (or erosivity) factor (yr^{-1})
Rp = Interception fraction of the edible portion of plant (unitless)

$SBCF$ = Scale bias correction factor (unitless)
SD = Sediment delivery ratio (unitless)
ΔSf = Entropy of fusion [$\Delta S_f/R = 6.79$ (unitless)]
SF = Slope factor $(\text{mg/kg-day})^{-1}$
S_T = Whitby's average surface area of particulates (aerosols)
$= 3.5\times10^{-6}\ \text{cm}^2/\text{cm}^3$ air for background plus local sources
$= 1.1\times10^{-5}\ \text{cm}^2/\text{cm}^3$ air for urban sources
SV = Screening value (ppmv)

T_a = Ambient air temperature (K)
T_1 = Time period at the beginning of combustion (yr)
T_2 = Length of exposure duration (yr)
tD = Time period over which deposition occurs (time period of combustion) (yr)
T_m = Melting point of chemical (K)
TOC_{VOC} = Stack concentration of volatile TOC, including speciated and unspeciated compounds (mg/m^3)
TOC_{CSV} = Stack concentration of CSV TOC, including speciated and unspeciated compounds (mg/m^3)
TOC_{GRAV} = Stack concentration of GRAV TOC, including speciated and unspeciated compounds (mg/m^3)
Tp = Length of plant exposure to deposition per harvest of edible portion of plant (yr)
tp_i = Length of plant's exposure to deposition per harvest of the edible portion of the i th plant group (yr)
Total Cancer Risk = Individual lifetime cancer risk through indirect exposure to all COPC carcinogens (unitless)
Total Cancer Risk$_{inh}$ = Total individual lifetime cancer risk through direct inhalation of all COPC carcinogens
TSS = Total suspended solids concentration (mg/L)
T_{wk} = Water body temperature (K)
$t_{1/2}$ = Half-time of COPC (days)

u = Current velocity (m/s)

Vdv = Dry deposition velocity (cm/s)
Vf_x = Average volumetric flow rate through water body (m^3/yr)

LIST OF VARIABLES (Continued)

VG_{ag} = Empirical correction factor for aboveground produce (forage and silage)(unitless)
$VG_{rootveg}$ = Empirical correction factor for below ground produce (unitless)
VOC = Total VOC emission rate for an equipment type (kg/hr)
VOC_s = VOC emission rate from all equipment in the stream of a given equipment type (kg/hr)
Vp = Vapor pressure of COPC (atm)

W = Average annual wind speed (m/s)
w_b = Rate of burial (m/yr)
WF_{VOC} = Average weight fraction of VOC in the stream (unitless)

X_e = Unit soil loss (kg/m^2-yr)

Yh = Dry harvest yield = 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)
Yh_i = Harvest yield of *i*th crop (kg DW)
Yp = Yield or standing crop biomass of edible portion of plant (productivity) (kg DW/m^2)
Yp_i = Yield or standing crop biomass of the edible portion of the plant (productivity)
(kg DW/m^2)

Z_s = Soil mixing zone depth (cm)

0.01 = Units conversion factor (kg cm^2/mg-m^2)
10^{-6} = Units conversion factor (g/µg)
10^{-6} = Units conversion factor (kg/mg)
0.31536 = Units conversion factor (m-g-s/cm-µg-yr)
365 = Units conversion factor (days/yr)
907.18 = Units conversion factor (kg/ton)
0.1 = Units conversion factor (g-kg/cm^2-m^2)
0.001 = Units conversion factor (kg-cm^2/mg-m^2)
100 = Units conversion factor (mg-cm^2/kg-cm^2)
1000 = Units conversion factor (mg/g)
4047 = Units conversion factor (m^2/acre)
1×10^3 = Units conversion factor (g/kg)
3.1536×10^7 = Units conversion factor (s/yr)

Chapter 1
Introduction

What's Covered in Chapter 1:

- Objective and Purpose
- Related Trial Burn Issues
- Reference Documents
- Document Organization

Risk assessment is a science used to evaluate the carcinogenic risks and noncarcinogenic hazards to human health that are attributable to emissions from hazardous waste combustion units. These risk assessments include the evaluation of both direct and indirect risks. There is sufficient guidance available regarding the performance of direct inhalation risk assessments. On the other hand, indirect risk assessments are newer and more complex. As a result, this document describes the evaluation of direct inhalation risk, but primarily focuses on the procedures used to estimate risk resulting from indirect pathways. The following definitions as adopted from the National Academy of Sciences 1983, *Risk Assessment in the Federal Government*, for use throughout this guidance:

Risk Assessment	The scientific evaluation of potential health impacts that may result from exposure to a particular substance or mixture of substances under specified conditions.
Hazard	An impact to human health by 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	Defined as one oral exposure.
Exposure	Exposure to chemicals by relevant pathways to identified receptors.
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 include ingestion of above ground fruits and vegetables, beef and milk, freshwater fish and soil.

Direct Exposure Exposure via inhalation.

This Human Health Risk Assessment Protocol (HHRAP) has been developed as national guidance to consolidate information presented in other risk assessment guidance and methodology documents previously prepared by U.S. EPA and state environmental agencies. In addition, the HHRAP also addresses issues that have been identified while conducting risk assessments for existing hazardous waste combustion units. The overall purpose of this document is to explain how risk assessments should be performed at hazardous waste combustion facilities. This document is intended as (1) guidance for personnel conducting risk assessments, and (2) an information resource for permit writers, risk managers, and community relations personnel.

In the April 19, 1996, preamble to the proposed MACT rule, U.S. EPA recommended that site-specific risk assessments be conducted as part of the RCRA permitting process for hazardous waste combustors as necessary to protect human health and the environment. Often, the determination of whether or not a permit is sufficiently protective can be based on its conformance to the applicable technical standards specified in the regulations. Since the time that the current regulations for hazardous waste incinerators and boilers/industrial furnaces were issued (1981 and 1991, respectively), however, information has become available to suggest that these performance standards may not fully address potentially significant risks. Many recent studies (including the *Draft Health Reassessment of Dioxin-Like Compounds*, *Mercury Study Report to Congress*, and *Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Wastes: Background Information Document*) indicate that there can be significant risks from indirect exposure pathways (e.g., pathways other than direct inhalation). The food chain pathway appears to be particularly important for bioaccumulative pollutants which may be emitted from hazardous waste combustion units. In many cases, risks from indirect exposure may constitute the majority of the risk from a hazardous waste combustor. This key portion of the risk from hazardous waste combustor emissions was not directly taken into account when the hazardous waste combustion standards were developed. In addition, uncertainty remains regarding the types and quantities of non-dioxin products of incomplete combustion emitted from combustion units and the risks posed by these compounds.

The RCRA "omnibus" authority of §3005(c)(3) of RCRA, 42 U.S.C. §6925(c)(3) and 40 CFR. §270.32(b)(2) gives the Agency both the authority and the responsibility to establish permit conditions on a case-by-case basis as necessary to protect human health and the environment. Performance of a site-specific risk assessment can provide the information necessary to determine what, if any, additional permit conditions are necessary for each situation to ensure that operation of the combustion unit is protective of human health and the environment. Under 40 C.F.R. §270.10(k), U.S. EPA may require a permit applicant to submit additional information (e.g., a site-specific risk assessment) that the Agency needs to establish permit conditions under the omnibus authority. In certain cases, the Agency may also seek additional testing or data under the authority of RCRA §3013 (where the presence or release of a hazardous waste "may present a substantial hazard to human health or the environment") and may issue an order requiring the facility to conduct monitoring, testing, analysis, and reporting. Any decision to add permit conditions based on a site-specific risk assessment under this authority must be justified in the administrative record for each facility, and the implementing agency should explain the basis for the conditions.

The permitting agency should consider several factors in its evaluation of the need to perform a risk assessment (human health and ecological). These factors include:

- whether any proposed or final regulatory standards exist that U.S. EPA has shown to be protective for site-specific receptors
- whether the facility is exceeding any final technical standards
- the current level of hazardous constituents being emitted by a facility, particularly in comparison to proposed or final technical standards, and to levels at other facilities where risks have been estimated
- the scope of waste minimization efforts and the status of implementation of a facility waste minimization plan
- particular site-specific considerations related to the exposure setting (such as physical, land use, and sensitive subpopulation characteristics) and the impact of these characteristics on potential risks
- the hazardous constituents most likely to be found and those most likely to pose significant risk
- the volume and types of wastes being burned
- the level of public interest and community involvement attributable to the facility

This list is by no means exhaustive, but is meant only to suggest significant factors that have thus far been identified. Others may be equally or more important.

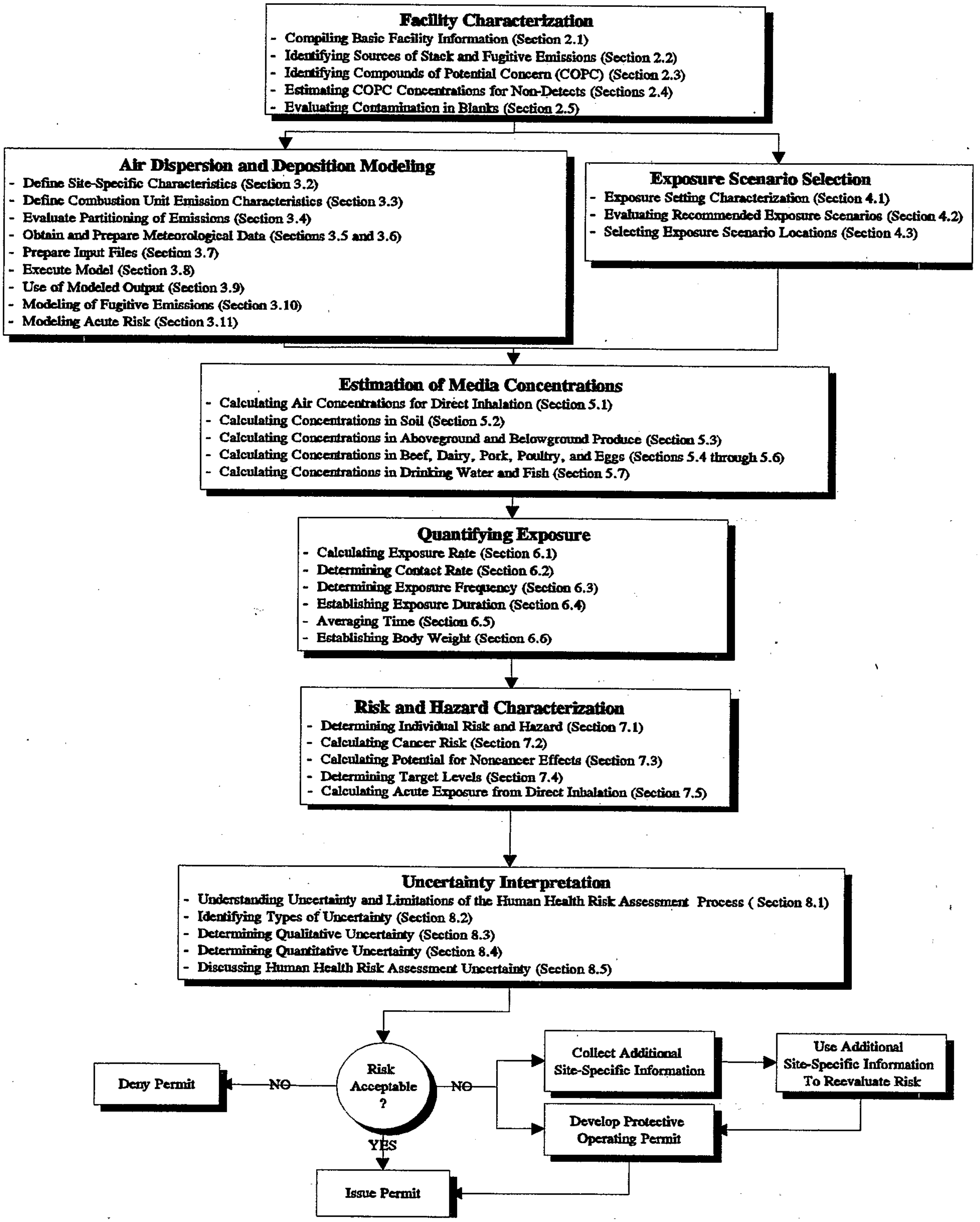
The companion document of the HHRAP is the Screening Level Ecological Risk Assessment Protocol (SLERAP). U.S. EPA OSW has prepared these guidance documents as a resource to be used by authorized agencies developing risk assessment reports to support permitting decisions for hazardous waste combustion units.

1.1 OBJECTIVE AND PURPOSE

U.S. EPA OSW's objective is to present a user-friendly set of procedures for performing risk assessments, including (1) a complete explanation of the basis of those procedures, and (2) a comprehensive source of data needed to complete those procedures. The first volume of this document provides the explanation (Chapters 1 through 9); and the second volume (Appendixes A-B) provides the data sources. Appendix A presents compound-specific information necessary to complete the risk assessment. Appendixes B and C present a user-friendly set of procedures for performing risk assessments. Figure 1-1 summarizes the tasks needed to complete a risk assessment and refers the reader to chapters in this guidance in which each task is described.

Implementation of this guidance will demonstrate that developing defensible estimates of compound emission rates is one of the most important elements of the risk assessment. As described in Chapter 2, traditional trial burns conducted to measure destruction and removal efficiency (DRE) do not sufficiently characterize organic products of incomplete combustion (PIC) and metal emissions for use in performing risk assessments. In some instances, a facility or regulatory agency may want to perform a pretrial burn risk assessment, following the procedures outlined in this document, to ensure that sample collection times during the trial burn or risk assessment burn are sufficient to collect the sample volumes needed to meet the detection limits needed for the risk assessment. The decision to perform such an assessment should consider regulatory permitting schedules and other site-specific factors.

FIGURE 1-1
HUMAN HEALTH RISK ASSESSMENT PROCESS



U.S. EPA OSW anticipates that risk assessments will be completed for new and existing facilities as part of the permit application process. The HHRAP recommends a process for evaluating *reasonable*—not theoretical worst-case maximum—potential risks to receptors posed by emissions from RCRA regulated units. The use of existing and site-specific information early in, and throughout, the risk assessment process is encouraged; conservative assumptions should be made only when needed to ensure that emissions from combustion units do not pose unacceptable risks. More conservative assumptions may be incorporated to make the process fit a classical "screening level" approach that is more conservative and may be easier to complete.

Regardless of whether theoretical worst case or more reasonable conservative assumptions are used in completing the risk assessment process, every risk assessment is limited by the quantity and quality of:

- site-specific environmental data
- emission rate information
- other assumptions made during the risk estimation process (for example, fate and transport variables, exposure assumptions, and receptor characteristics)

These limitations and uncertainties are described extensively throughout this document and the appendixes, and are summarized in Chapter 8.

Unacceptable risks or other significant issues identified by collecting preliminary site information and completing risk assessment calculations can be addressed by the permitting process or during an iteration of the risk assessment. After the initial risk assessment has been completed, it may be used by risk managers and permit writers in several ways:

- If the initial risk assessment indicates that estimated cancer risks and noncancer hazards are below regulatory levels of concern, risk managers and permit writers will likely proceed through the permitting process without adding any risk-based unit operating conditions to the permit.
- If the initial risk assessment indicates potentially unacceptable risks, additional site-specific information demonstrated to be more representative of the exposure setting may be collected and additional iterations of risk assessment calculations can then be performed.

- If the initial risk assessment or subsequent iterations indicate potentially unacceptable risks, risk managers and permit writers may use the results of the risk assessment to revise tentative permit conditions (for example, waste feed limitations, process operating conditions, and expanded environmental monitoring). To determine if the subject hazardous waste combustion unit can be operated in a manner that is protective of human health and the environment, an additional iteration of the risk assessment should be completed using the revised tentative operating conditions. If the revised conditions still indicate unacceptable risks, this process can be continued in an iterative fashion until acceptable levels are reached. In some situations, it may be possible to select target risk levels and back-calculate the risk assessment to determine the appropriate emission and waste feed rate levels. In any case, the acceptable waste feed rate and other appropriate conditions can then be incorporated as additional permit conditions.

- If the initial risk assessment, or subsequent iterations, indicate potentially unacceptable risks, risk managers and permit writers may also choose to deny the permit.

This process is also outlined in Figure 1-1. As stated earlier, in some instances, a facility or regulatory agency may want to perform a pretrial burn risk assessment—following the procedures outlined in this document—to ensure that sample collection times during the trial burn or risk assessment burn are sufficient to collect the sample volumes necessary to meet the appropriate detection limits for the risk assessment. This is expected to reduce the need for additional trial burn tests or iterations of the risk assessment due to problems caused when detection limits are not low enough to estimate risk with certainty sufficient for regulatory decision making. For example, if detection limits are too high then estimates of risk based on detection limits may be overly conservative.

1.2 RELATED TRIAL BURN ISSUES

In the course of developing this guidance and completing risk assessments across the country, U.S. EPA OSW has learned that developing defensible estimates of compound of potential concern (COPC) emission rates is one of the most important parts of the risk assessment process. As described in Chapter 2, traditional trial burns conducted to measure destruction and removal efficiency (DRE) *do not* sufficiently characterize organic products of incomplete combustion (PIC) and metal emissions for use in performing risk assessments.

U.S. EPA OSW considers the trial burn and risk assessment planning and implementation processes as interdependent aspects of the hazardous waste combustion unit permitting process. In addition, U.S. EPA

OSW advocates that facility planning, regulatory agency review, and completion of tasks needed for both processes be conducted simultaneously to eliminate redundancy or the need to repeat activities. U.S. EPA OSW expects that the following guidance documents will typically be used as the main sources of information for developing and conducting appropriate 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 (OSWER). EPA/625/6-89/021. June.
- U.S. EPA. 1992c. *Technical Implementation Document for EPA's Boiler and Industrial Furnace Regulations.* OSWER. EPA-530-R-92-011. March.
- U.S. EPA. 1994n. *Draft Revision of Guidance on Trial Burns. Attachment B, Draft Exposure Assessment Guidance for Resource Conservation and Recovery Act (RCRA) Hazardous Waste Combustion Facilities.* OSWER. June 2.
- Generic Trial Burn Plan and QAPPs developed by EPA regional offices or states.

1.3 REFERENCE DOCUMENTS

This section describes, in chronological order, the primary guidance documents used to prepare this HHRAP. Some of the guidance documents received a thorough review from EPA's Science Advisory Board, which mostly supported the work. Additional references used to prepare this HHRAP are listed in the References chapter of this document. These documents have been developed over a period of several years; in most cases, revisions to the original guidance documents address only the specific issues being revised rather than representing a complete revision of the original document. The following discussion lists and briefly describes each document. Overall, each of the guidance documents reflects a continual enhancing of the methodology. The most current risk assessment methodology frequently referenced in this guidance is the U.S. EPA NCEA guidance, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press).

References, such as "U.S. EPA 1990e," correspond to the citation for the document specified in the Reference section of this guidance.

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

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

This document outlined and explained a set of general procedures for conducting risk assessments. This document was subsequently revised by the following:

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

This document outlined recommended revisions to previous U.S. EPA guidance (1990e), which have been used by the risk assessment community since the release of the document; however, these recommended revisions were never formally incorporated into the original document. In 1994, U.S. EPA issued several additional risk assessment documents, including the following:

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

The actual substance of the 1994 U.S. EPA guidance (1994f) is included in the following series of attachments, all issued 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.
- U.S. EPA. 1998 (In Press). *"Guidance on Collection of Emissions Data to Support Site-Specific Risk Assessments at Hazardous Waste Combustion Facilities.* Internal Review Draft. Prepared by EPA Region 4 and the Office of Solid Waste.

Combined, these four documents present a revised procedure for completing a risk assessment. Because the original U.S. EPA guidance documents (1990e and 1993h) contained much of the background information necessary to complete the risk assessment process, this information was not repeated. In 1994, this new guidance was further revised by the following documents:

- 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.
- 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.

More recently, NC DEHNR developed the following guidance document for conducting risk assessments:

- 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 allows the regulatory agency or facility to choose the investment they want to make in conducting risk assessments. For instance, a small, on-site unit with limited waste stream variability is allowed the opportunity to conduct a Tier 1 assessment (more worst-case), whereas a larger facility with a diverse waste feed mixture may decide to complete a Tier 2 or 3 assessment (progressively more site-specific).

Finally, U.S. EPA OSW contracted for the development of *The Background Information Document to the Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion*

Units Burning Hazardous Wastes (Research Triangle Institute 1996) to support the proposed Hazardous Waste Combustion Rule. This document was reviewed and considered throughout the development of the HHRAP in order to ensure that the approach outlined is consistent with the most current OSW risk assessment policy.

Chapter 2
Facility Characterization

What's Covered in Chapter 2:

- Compiling Basic Facility Information
- Identifying Emission Sources
- Estimating Emission Rates
- Identifying Compounds of Potential Concern (COPCs)
- Estimating COPC Concentrations for Non-Detects
- Evaluating Contamination In Blanks

This chapter provides guidance on characterizing the nature and magnitude of emissions released from facility sources. The characterization 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.

2.1 COMPILING BASIC FACILITY INFORMATION

Basic facility information should be considered in conducting the risk evaluation, and provided to enable reviewers to establish a contextual sense of the facility regarding how it relates to other facilities and other hazardous waste combustion units. At a minimum, the basic facility information listed in the highlighted box at the end of this and other sections should be considered in the risk evaluation. The following sections and chapters describe the collection of this information in more detail; however, users may want to consult these discussions so that all site-specific information needed to complete the risk assessment can be collected simultaneously, when appropriate, for up front consideration.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Setting characterization
- Principal business and primary production processes
- Normal and maximum production rates
- Types of waste storage and treatment facilities
- Type and quantity of wastes stored and treated
- Process flow diagrams showing both mass and energy inputs and outputs

2.2 IDENTIFYING EMISSION SOURCES

Combustion of a hazardous waste generally results in combustion by-products being emitted from a stack. In addition to emissions from the combustion stack, additional types of emissions of concern that may be associated with the combustion of hazardous waste include (1) process upsets, (2) general RCRA fugitive emissions, (3) cement kiln dust (CKD) fugitive emissions, and (4) accidental releases. Each of these emission source types are defined below with regards to the context and scope of this guidance.

> ***Stack Emissions*** - Release of compounds or pollutants from a hazardous waste combustion unit into the ambient air while the unit is operated as intended by the facility and in compliance with a permit and/or regulation (for interim status).
>
> ***Process Upset Emissions*** - Release of compounds or pollutants from a hazardous waste combustion unit into the ambient air while the unit is not being operated as intended, or during periods of startup or shutdown. Upset emissions usually result from an upset in the hazardous waste combustion process and are often known as process upset emissions. Upset emissions are generally expected to be greater than stack emissions because the process upset results in incomplete destruction of the wastes or other physical or chemical conditions within the combustion system that promote the formation and/or release of hazardous compounds from combustion stacks. Upset emissions usually occur during events and times when the hazardous waste combustion unit is not operating within the limits specified in a permit or regulation.
>
> ***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 compounds or 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 compounds or pollutants into the ambient air caused by the handling, storage, and disposal of cement kiln dust.

Accidental Release - 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, or transportation accidents.

Consistent with previous U.S. EPA guidance (U.S. EPA 1994i), U.S. EPA OSW recommends that, with the exception of accidental releases, all of these emission source types be addressed in the risk assessment, as applicable. Accidental releases are not considered within the scope of this guidance, and should be evaluated as recommended in Section 112(r) of the CAA and current U.S. EPA guidance (U.S. EPA 1996f) or the *RMP Offsite Consequence Analysis Guidance*, dated May 24, 1996. A decision to consider accidental releases in risk assessments for hazardous waste combustion facilities should be made on a site specific basis by the relevant permitting authority.

The following subsections contain guidance for estimating emissions for the source types specified for inclusion 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

Stack emission rates (in grams per second) need to be determined for every compound of potential concern (COPC) identified using the procedures outlined in Section 2.3. U.S. EPA OSW expects that emission rates used to complete the risk assessment will be (1) long-term average emission rates adjusted for upsets, or (2) reasonable maximum emission rates measured during trial burn conditions in order to assure that risk assessments are conservative. Maximum emission rates measured during trial burn conditions (see Section 2.2.1.1) represent reasonable maximum emission rates. These emission rates can be controlled by hourly rolling average permit limits traditionally found in combustion unit operating permits, and are more conservative than emission estimates that are based on long-term average emission rates. Long-term average emission rates (see Section 2.2.1.2) are based on tests of the combustion unit burning worst-case wastes at operating conditions that are representative of normal operating conditions over a long-term period. If long-term average emission rate estimates are used in the risk assessment, the final permit will likely specify limitations in addition to any hourly rolling average limit typically used to regulate hazardous waste combustion facilities.

A permitting agency's decision to allow a facility to use emission rate data developed from either normal or maximum operating conditions will be made on a case-by-case basis. Some facilities may be required to use emission rate data developed from maximum operating conditions because the variability in waste feed and operating conditions is too great to make permit decisions based on emission data collected during normal operating conditions, or because the emissions from combustion of the waste feed material are anticipated to be highly toxic and only a conservative risk assessment can adequately ensure protection.

2.2.1.1 Estimates from Trial Burns

For existing facilities (such as those built and operational), emission rate information will generally be determined by direct stack measurements during pretrial burn or trial burn tests, because trial burn tests are required before a facility can be granted a permit to burn hazardous wastes. This policy is consistent with U.S. EPA 1998 *Guidance on Collection of Emissions Data to Support Site-Specific Risk Assessments at Hazardous Waste Combustion Facilities*, prepared by U.S. EPA Region 4 and OSW (U.S. EPA 1998). For new facilities (see Section 2.2.3), estimated emission rates used to complete pretrial burn risk assessments should be compared to the emission rates estimated from actual trial burns completed after the new facility receives a permit and is constructed. Trial burn tests are designed to produce emission rates higher than those anticipated under normal operating conditions. U.S. EPA OSW recommends that sampling be conducted, in accordance with U.S. EPA guidance on conducting trial burns, by using compound-specific stack sampling, analytical, and quality assurance/quality control (QA/QC) protocols and procedures approved by the permitting authority. An alternative to a trial burn test is the submittal of data "in lieu of" a trial burn. U.S. EPA OSW will consider this type of data for on-site units on a case-by-case basis. U.S. EPA OSW expects that this data to be based on recent stack test measurements from a similar type of combustion unit with similar waste feed, capacity, operating conditions, and air pollution control systems (APCSs) to ensure comparable emission rates and destruction and removal efficiencies (DREs).

U.S. EPA OSW expects that using data from a trial burn as a basis for estimating COPC emission rates will tend to overestimate risk. COPC emission rates measured during trial burns are expected to be greater than emission rates during normal unit operations, because a facility "challenges" its combustion unit during a trial burn to develop a wide range of conditions for automatic waste feed cutoff (AWFCO) systems. Trial burn 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 combustion unit to destroy principal organic hazardous constituents (POHCs) in the waste feed is challenged. The lessor of the 95th percentile or maximum stack gas concentration from the three trial burn runs should be used to develop the emission rate estimate used in the risk assessment.

High POHC feed rates and extreme operating conditions tested during the low-temperature trial burn test are usually expected to result in greater product of incomplete combustion (PIC) emission rates. However, this is not true in all cases. For example, the formation of PCDDs and PCDFs does not necessarily depend on "POHC incinerability" low temperature conditions. Polychlorinated dibenzo(p)dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) can be formed as a result of (1) catalytic formation in the low-temperature regions of the combustion unit or APCS during the low temperature test, or (2) catalytic formation that is dependent on high APCS temperatures typically experienced during the high temperature test.

Because the amount of testing required to develop estimates of COPC emission rates is so extensive and time consuming, U.S. EPA OSW places the responsibility for selecting the test conditions first on the facility and then on the permit writer. If a facility desires to receive a permit with no limits other than those traditionally based on hourly rolling average data gathered during a trial burn, then risk testing should be conducted during trial burn or "worst case" conditions. Whether the permit writer requires testing to be conducted at low, high, or both temperature conditions is a decision that must be made by the permit writer based on the characteristics of the facility and policy set forth by the senior management of the appropriate regulatory agency.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- All (current and historical) stack sampling information regarding rates of emissions from the combustion unit during normal or trial burn conditions

- Description of the waste feed streams burned during the stack sampling, including chemical composition and physical properties, which demonstrate that the waste feeds are representative of worst case site-specific "real" wastes

*** * * NOTICE * * ***

Although U.S. EPA OSW will not require a risk assessment for every possible metal or PIC from a combustion unit, this does not imply that U.S. EPA OSW will allow only targeted sampling for COPCs during trial burn tests. Based on regional permitting experience and discussions with regional analytical laboratories, U.S. EPA OSW maintains that complete target analyte list analyses conducted when using U.S. EPA standard sampling methods (e.g., 0010 or 0030), do not subject facilities to significant additional costs or burdens during the trial burn process. Facilities conducting stack emission sampling should strive to collect as much information as possible which characterizes the stack gases generated from the combustion of hazardous waste. Therefore, every trial burn or "risk burn" should include, at a minimum, the following tests: Method 0010, Method 0030 or 0031 (as appropriate), total organic compounds (using the Guidance for Total Organics, including Method 0040), Method 23A, and the multiple metals train. Other test methods may be approved by the permitting authority for use in the trial burn to address detection limit or other site-specific issues.

2.2.1.2 Normal Operation Emission Rate Data

Facilities with limited waste feed characteristic and operational variability may be allowed to conduct risk testing at normal operational conditions (U.S. EPA 1994j). The collection of COPC data during normal operating conditions is referred to as a "risk burn" throughout the remainder of this guidance. It is important to note, however, that a risk burn does not replace a traditional trial burn conducted to measure DRE. Instead, U.S. EPA OSW considers a risk burn as an additional operating condition of the trial burn during which data is collected for the purpose of completing a risk assessment.

Because operational data collected during the risk burn would not normally be extrapolated to hourly rolling average AWFCO limits specified in an operating permit; the regulatory agency permit writer should craft the permit with conditions designed to ensure that the facility does not operate at conditions in "excess" of the normal conditions over the long-term operation of the facility (for example, waste feed rate or stack gas flowrate). These additional permit limits are anticipated to take the form of quarterly or annual mass feed limitations on the waste feed, quarterly or annual average temperatures or stack gas flow rates, and other appropriate limitations.

It may also be necessary for the permit to contain appropriate reporting requirements to ensure that the regulatory agency can verify that the facility does not normally operate at conditions in excess of those tested during the risk burn. Monthly, quarterly, or annual reports which document long-term operations will likely be required of the facility. If a facility violates a long-term permit condition, the permit writer may also include language that requires the facility to cease waste burning immediately until a new test, risk assessment, and/or revised permit are completed. More detailed guidance on the development of permit limits can be found in U.S. EPA Region 6's *Hazardous Waste Combustion Permitting Manual*; which can be obtained from the U.S. EPA Region 6 web page (www.epa.gov/region06/).

One of the most important criteria which should be evaluated when considering the collection of data during a risk burn rather than a trial burn is the ability of the facility to document that the test is conducted with "worst case" waste. Worst case waste should be the waste feed material or combination of materials that are most likely to result in significant emissions of COPCs. The potential for both PIC and metal emissions should be considered in the selection of the worst case waste. For example, if a facility burns two types of waste—one waste with a high chlorine content and a significant concentration of aromatic organic compounds and a second with a low chlorine content and a significant concentration of alkanes—the former waste should be considered to be the "worst case" for PIC formation and should be used during the risk burn. A similar evaluation should be considered when selecting the worst case waste for metal emissions.

If a facility chooses to develop—and the appropriate regulatory agency allows the use of—emission rate estimates from a risk burn rather than a trial burn, the data set for each COPC should be the average of the COPC emission rate over all the acceptable test runs plus two standard deviations *or* the maximum COPC emission rate value from all acceptable test runs, whichever value is lower. U.S. EPA OSW does not believe that it is reasonable to perform a risk assessment with the 95th percentile emission rate if the maximum rate is less than this value. U.S. EPA OSW also recommends that, where possible, the COPC emission rate value from the trial burn test and the risk burn test be compared in the risk assessment report along with a comparison of the operational conditions at these two test conditions. For example, if the POHC used for the DRE test in the trial burn is a semivolatile organic compound (SVOC), the facility should analyze for all SVOCs (Method 0010) during the trial burn, and compare these values to those reported for the risk burn. The difference between the emission rates from the trial burn and risk burn should be evaluated in the uncertainty section of the risk assessment.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Sampling and analytical data for trial burn and risk burn (if the risk assessment is completed by using risk burn data) operating conditions
- Description of the operating conditions, under which each set of emission rate data being used was developed
- Complete evaluation of the differences between trial burn and risk burn operating conditions, with an explanation of the expected resultant risk differences

2.2.1.3 Estimates of the Total Organic Emission (TOE) Rate

Organic compounds that cannot be identified by laboratory analysis will not be treated as COPC's in the risk calculations. However, these compounds still may contribute significantly to the overall risk, and therefore, should be considered in the risk assessment (DeCicco 1995; U.S. EPA 1994i). U.S. EPA developed the total organic emissions (TOE) test to account for unidentified organic compounds because existing methods, such as total hydrocarbon analyzers, do not fully determine the total mass of organics present in stack gas emissions (Johnson 1996). U.S. EPA OSW anticipates that trial and risk burns will include sampling for TOE in order to provide permitting authorities with the information needed to address concerns about the unknown fraction organic emissions. The TOE can be used in conjunction with the identified organic compounds to calculate a TOE factor which can then be used to facilitate a evaluation of 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).

Use of the TOE data is dependent 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)** - TO_{VOC} is defined as the fraction of organic compounds having a boiling point 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)** - TO_{SVOC} is defined as the fraction of organic compounds having boiling points between 100°C and 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 to 300°C is determined by summing the total gas chromatorgraph/flame ionization detector results as described in the TO Guidance.
>
> ***Fraction 3:*** **Total Gravimetric Compounds (TO_{GRAV}) (referred to as Gravametric component in the TO Guidance)** - TO_{GRAV} is defined as the fraction of organic compounds having boiling points greater than 300°C. This fraction includes two types of compounds: (1) Identified SVOCs collected using U.S. EPA Method 0010 having boiling points greater than 300°C and (2) unidentified nonvolatile organics having 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 quantifies the mass, above this fractions boiling point, by measuring the total mass by evaporation and gravimetry (weighing) for nonvolatile total organics.

It should be noted that the TO total (TO_{TOTAL}) is the sum of the sums of each fraction. The sum of the TO fractions are 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)

The TOE data is used in conjunction with the identified data to compute a TOE factor. TOE factors have been computed which range from 2 to 40. The TOE factor is defined by this guidance as the ratio of the TO_{TOTAL} mass to the mass of identified organic compounds and calculated by the following equation:

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

where

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

One of the most critical components of the TOE factor is the identification of the organic compounds in the denominator of Equation 2-2. Although the permitting authority may not require a facility to analyze the organic compounds with all possible analytical methods, facilities should 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 be evolved 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, permitting authorities and facilities may choose to use additional test methods in the trial burn in order to speciate the maximum number of organic compounds.

U.S. EPA OSW also recommends that permitting authorities include tentatively identified compounds (TICs) in the denominator when computing the TOE factor to ensure that appropriate credit is given to defensible efforts at identifying the maximum number of organic compounds. Finally, U.S. EPA OSW recommends that non-detect compounds of potential concern be treated consistently between the risk assessment and TOE evaluation. That is, if a non-detected constituent is deleted as a compound of potential concern (See Section 2.3), then it would not be included in the identified fraction of the TOE equation. Compounds of potential concern identified as per Section 2.3, but not detected, should be included in the TOE factor equation at the reliable detection limit (non-isotope dilution methods) or the estimated detection limit (isotope dilution methods).

The results of the gravimetric fraction should also be carefully evaluated when using the TOE factor. Both regulated industry and U.S. EPA scientists have expressed some concern that the gravimetric fraction of TOE test may contain materials that are not organic. U.S. EPA Office of Research and Development National Risk Management Research Laboratory (NRMRL) recently completed a study conducted to identify products of incomplete combustion (U.S. EPA NRMRL 1997b). U.S. EPA NRMRL suggested in the study report that the gravimetric fraction of the TOE test may consist of organic and/or inorganic mass not directly attributable to organic incinerator emissions. U.S. EPA NRMRL theorized that these artifacts could consist of inorganic salts, super-fine particulate, or fractured XAD-2 resin. U.S. EPA NRMRL also concluded in this study report that the vast majority of the non-target semivolatile organic compounds detected, but not fully identified, were alkanes with more than 10 carbon atoms, esters of high molecular weight carboxylic acids, and phthlates. Most problems associated with accurately determining the gravimetric fraction attributable to incinerator emissions can be minimized; see the U.S. EPA 1998 *Guidance on Collection of Emissions Data to Support Site-Specific Risk Assessments at Hazardous Waste Combustion Facilities* (U.S. EPA 1998 In Press) for minimizing sample errors.

The TOE factor is used in the uncertainty section of the risk assessment report to evaluate the risks from the unknown fraction of organics. Permitting authorities can 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 may decide to do the following:

1. Describe in a narrative form what is known of the unknown portion of the emissions.
2. Attribute a risk to the unknown portion of the emissions. An example was presented as a preferred option in U.S. EPA (1994f) which assumed 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. Require 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.

Permitting authorities may use variations of the TOE factor to address site-specific concerns. For example, some permitting authorities may compute three separate TOE factors based on the apportioning provided by the TOE test (i.e., TO_{VOC}, TO_{SVOC}, and TO_{GRAV}). The unknowns associated with each separate fraction of unidentified organic compounds can then be evaluated separately.

2.2.2 Estimating Emission Rates for Facilities with Multiple Stacks

Emissions from all combustion units burning hazardous waste at a facility, not just the unit currently undergoing the permitting process, should be considered in the risk assessment. As discussed further in Chapter 3, air dispersion modeling for each combustion unit (source) should be conducted separate from the other combustion units, to allow evaluation of risk on a stack or source-specific basis. A case example is where a chemical manufacturing facility may operate both an on-site incinerator and several hazardous waste burning boilers. Whether it is the incinerator or the boilers undergoing the permitting process, the risk assessment should consider the emissions from all the combustion units in the estimate of facility risk. In addition to RCRA combustion units, emissions from other RCRA treatment, storage, or disposal units (e.g., open burning/open detonation and thermal desorption) may also be included in the risk evaluation in some cases.

2.2.3 Estimating Stack Emission Rates for Facilities Not Yet Operational

New hazardous waste combustion facilities should submit a Part B permit application, go through an extensive permitting process, and, if successful, receive a final permit to commencement of operation. The permitting process requires submittal of sufficiently detailed information for the regulatory authorities to evaluate compliance with existing regulations, guidance, and protectiveness. Stack (source) locations and dimensions, design flow and emission rate estimates, waste feed characteristics, surrounding building dimension data, facility plot plans, and terrain data should be reviewed and used in a pre-operation risk assessment. This will assist in decision-making and designing permit requirements.

The design emission rates, waste feed characteristics, and other design data should be reviewed along with supplementing documentation to assure 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 should be used to estimate COPC emission rates for new facilities that have not been constructed.

If the preferred option of using surrogate data from similar facilities is not available, some state environmental agencies enforce emission rate limits based on state laws. Since these limits cannot be exceeded, they can be used to develop emission rate estimates for the risk assessment. The facility will demonstrate that its emissions are less than the those considered in the permit and risk assessment during the trial or risk burn.

Other data which may cause problems when performing risk assessments for new facilities is particle size distribution. A default particle size distribution is presented in Chapter 3 for use if particle size distribution data from a similar type of facility are not available.

2.2.4 Estimating Stack Emission Rates for Facilities Previously Operated

Emissions from the historical operation of combustion units burning hazardous waste at a facility, not just the unit currently undergoing the permitting process, may also be considered in the risk assessment on a case-by-case basis as determined by the permitting authority. Such a case may be when the emissions from historical operation of a source or sources may have already resulted in potential risk concerns at or surrounding the facility. Emissions from historical operations could be taken into consideration by modeling as a separate source or, if applicable, in the fate and transport equations by adding the previous years of operation to the anticipated time period of combustion for a new or existing operating source. In addition to RCRA combustion units, historical emissions from other RCRA treatment, storage, or disposal units (e.g., open burning/open detonation and thermal desorption) at the facility under evaluation may also be included in the risk assessment in some cases.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

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

*** * * NOTICE * * ***

Facilities may use estimated emission rate data from other combustion units only to determine whether the construction of a new combustion unit should be completed. After a combustion unit has been constructed, U.S. EPA OSW will require an additional risk assessment using emission rates collected during actual trial burn conditions.

2.2.5 Emissions From Process Upsets

Uncombusted hazardous waste can 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 are not generally expected to significantly increase stack emissions over the lifetime of the facility.

Process upsets occur when the hazardous waste combustion unit is not being operated as intended, or during periods of startup or shutdown. Upset emissions are generally expected to be greater than stack emissions (over short periods of time) because the process upset results in incomplete destruction of the wastes or other physical or chemical conditions within the combustion system that promote the formation and/or release of hazardous compounds from combustion stacks. Upset emissions usually occur during events and times when the hazardous waste combustion unit is not operating within the limits specified in a permit or regulation.

To account for the increased emissions associated with process upsets, the stack emission rate estimated from trial burn data is multiplied by an upset factor. When available, facilities should use site specific emissions or process data to estimate the upset factor. The following types of data may be considered and evaluated to derive the upset factor:

- Data for continuous emissions monitoring systems that measure stack carbon monoxide, oxygen, total hydrocarbon (if required), or opacity (if appropriate)
- Data on combustion chamber, APCS, or stack gas temperature
- Frequency and causes of automatic waste feed cutoffs (AWFCO)
- Ratio of AWFCO frequency and duration to operating time
- APCS operating variables, such as baghouse pressure drop, liquid scrubber flow rate, or electrostatic precipitator voltage
- Stack test collected while the combustion unit was operated under upset conditions

This information may be analyzed with the objective of estimating the magnitude of the increase in emissions and the percentage of time on an annual basis that the unit operates at upset conditions.

When site specific data are not available or are inappropriate for deriving an upset factor, consistent with previous guidance (U.S. EPA 1993h), U.S. EPA OSW recommends that upset emissions be estimated by 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 does not operate under upset conditions continually, the factor must be adjusted to account for only the period of time, on an annual basis, that the units 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 combustion unit 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 resulting from non-routine events such as explosions, fires, and power failures are considered accidental releases and are not addressed by this guidance.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Historical operating data demonstrating the frequency and duration of process upsets
- A discussion on the potential cause 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 that should be evaluated in the risk assessment include waste storage tanks; process equipment ancillary to the combustion unit; 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 require evaluation in some cases.

This section contains guidance for quantitatively estimating fugitive emissions on the basis of procedures outlined by 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

Quantitative estimation of RCRA fugitive emissions includes (1) identifying equipment to be evaluated as fugitive emission source(s), (2) grouping equipment, as appropriate, into a combined source, and

(3) estimating compound specific emission rates for each source. Figure 2-1 is an example of a facility plot plan 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 for a hypothetical example facility. This figure, as well as Tables 2-1 and 2-2, have been provided as an example to facilitate understanding of each of the steps presented for estimating fugitive emissions.

Step 1: Identifying Fugitive Emission Sources - Generally, RCRA fugitive emission sources to be evaluated in the risk assessment should include waste storage tanks and process equipment that comes in contact with a RCRA hazardous waste such as equipment specified in Title 40, Code of Federal Regulations (40 CFR) Part 265, Subpart BB. Equipment covered under Subpart BB includes the following:

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

Each fugitive emission source should be identified 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).

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 per type at the source was multiplied by the equipment specific emission factor (Column 6) to obtain equipment specific emission rate for that respective type of equipment (Column 7).
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:

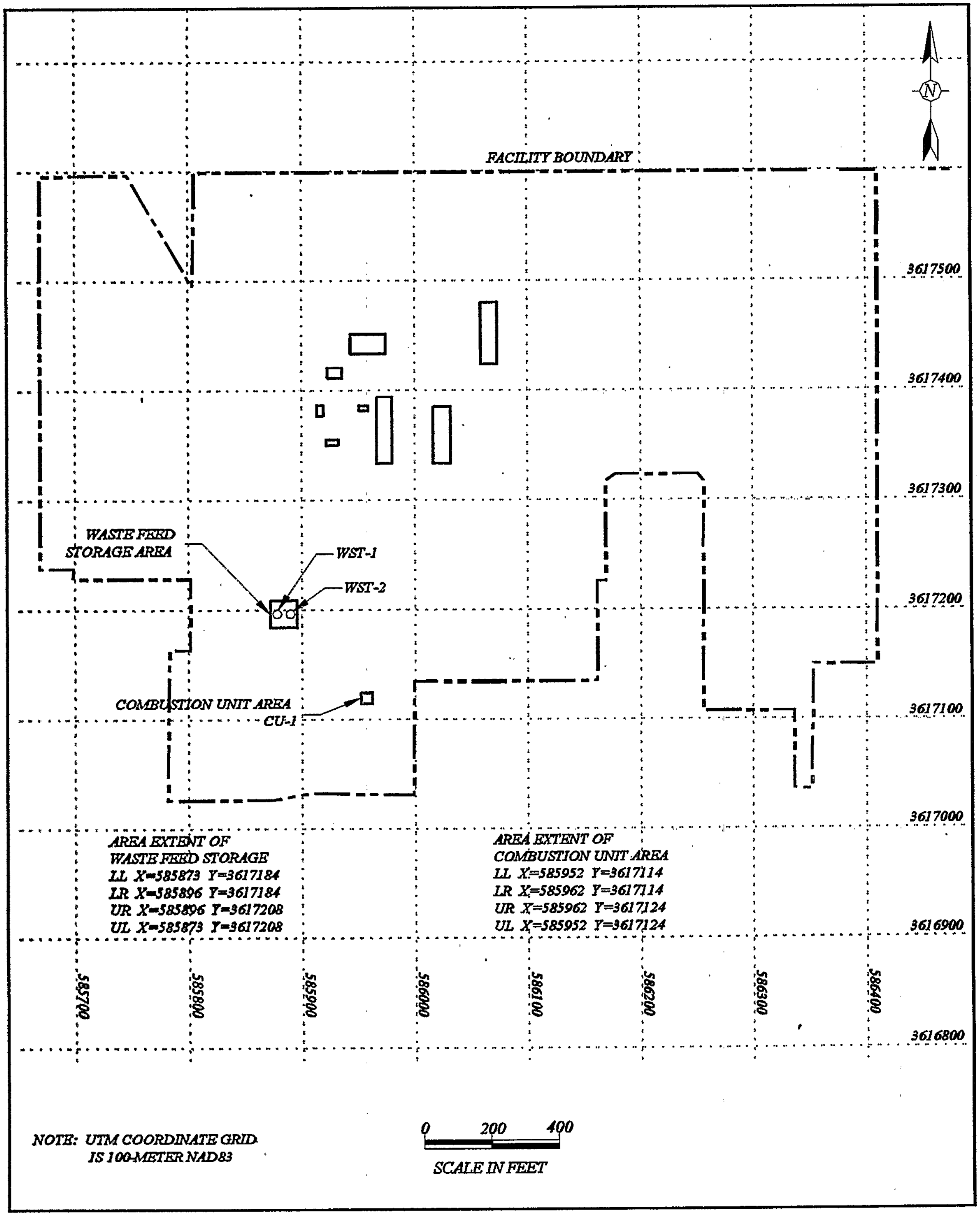
Column 1	Equipment in the Waste Feed Storage Area was identified and grouped as a combined RCRA fugitive emission source with an aerial extent defined by UTM coordinates (NAD83).
Column 2	The waste streams serviced by equipment in the Waste Feed Storage Area can be determined through review of the facility's 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.

Step 2: Grouping Equipment Into a Combined Source - To significantly reduce the effort required to complete air dispersion modeling and subsequent risk assessment, equipment in close proximity may be grouped and evaluated as a single combined source with the speciated emission rates for each piece of equipment summed. The area extent of the grouped or combined source, as defined by UTM coordinates (specify if NAD27 or NAD83), should be clearly denoted on a facility plot map. The area extent of the combined source should be defined by the actual locations of the equipment being grouped, without exaggeration to cover areas without fugitive sources. Consideration should also be made for how fugitive emission sources are to be defined when conducting the air dispersion modeling (see Chapter 3).

As shown in Figure 2-1, equipment in two areas at the hypothetical facility have been grouped into combined sources; these consist of the Waste Feed Storage Area and the RCRA Combustion Unit Area.

Step 3: Estimating Fugitive Emissions from Tanks - Fugitive emission rates for waste storage tanks can be obtained 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 the facility does not have such information 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 guidance document (1995a), *"Compilation of Air Pollution Emission Factors, January 1995."*

FIGURE 2-1
EXAMPLE FACILITY PLOT MAP



AREA EXTENT OF
WASTE FEED STORAGE
LL X=585873 Y=3617184
LR X=585896 Y=3617184
UR X=585896 Y=3617208
UL X=585873 Y=3617208

AREA EXTENT OF
COMBUSTION UNIT AREA
LL X=585952 Y=3617114
LR X=585962 Y=3617114
UR X=585962 Y=3617124
UL X=585952 Y=3617124

NOTE: UTM COORDINATE GRID
IS 100-METER NAD83

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)* *UTM zone no. 15* *Horizontal Coordinate 589100 m E* *Vertical coordinate 3616200 m N*

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

The information required for estimating fugitive emission rates from storage tanks includes, but is not limited to, the following:

- 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: Estimating Fugitive Emissions from Process Equipment - Based on guidelines provided in U.S. EPA (1995e), *"Protocol for Equipment Leak Emission Estimates, EPA-453/R-93-017,"* fugitive emissions for each equipment listed under 40 CFR Part 265, Subpart BB can be estimated by the following four approaches, 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 can be used at any facility to estimate fugitive emission rates of volatile organic compounds (VOCs) from equipment. Except for the AEFA method, all of the approaches require screening data collected by using a portable monitoring device (PMD). Because data on fugitive emissions at a facility is generally limited, the AEFA method will apply in most cases, and therefore, has been selected for use in the example demonstrated in Figure 2-1, and Tables 2-1 and 2-2. However, U.S. EPA OSW recommends that facilities use 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.

An Example Calculation Using the AEFA Method

Information required for estimating fugitive emission rates using the AEFA method is as follows:

- 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)

When this approach is used, equipment can be grouped by waste streams of similar characteristics and VOC composition (Columns 1 and 2, Table 2-1). However, the AEFA approach does not 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.

The average emission factors for synthetic organic chemicals manufacturing industry process units, refineries, and natural gas plants are presented in U.S. EPA (1995k) (Column 6, Table 2-1). 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.

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 (1993e)

The total VOC emissions rate for a specified equipment type can be calculated by multiplying 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).

The total VOC emission rates for each equipment type are summed to generate the total fugitive emission rate for the waste stream by (Column 10, Table 2-1). Speciated fugitive emissions can then be 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 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 Combustion Unit Leaks

Fugitive emissions that result from the construction, design, or operation of a combustion unit burning hazardous waste should be evaluated, as appropriate. Examples of fugitive emissions from combustion unit leaks include the following:

- Combustion units that operate 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 combustion unit 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, U.S. EPA OSW does not offer any specific quantitative guidance on how to estimate fugitive emissions from hazardous waste combustion units. However, risks associated with emissions from hazardous waste combustion unit leaks can be addressed in the uncertainty section of the risk assessment if no site specific quantitative methods are available. Specifically, the permitting authority can review facility specific data to determine whether or not the design addresses equipment leaks and whether the operational data indicates that equipment leaks may be a problem.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

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

2.2.7 RCRA Fugitive Ash Emissions

The combustion of hazardous waste materials may result in the production of flyash. Fugitive particle emissions may result from the subsequent 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 equipment, however, a fraction of the flyash may still escape into the atmosphere as fugitive emissions.

2.2.7.1 Quantitative Estimation of RCRA Fugitive Ash Emissions

Steps for the quantitative estimation of RCRA fugitive ash emissions include (1) determining an empirical emission factor, (2) estimating the flyash generation rate, and (3) accounting for air pollution control equipment, if applicable. As demonstrated in the example calculation below, the fugitive ash emission rate can then be estimated by multiplying the empirical emission factor by the flyash generation rate and the control deficiency of the air pollution control equipment, if applicable.

Step 1: Determining an Empirical Emission Factor - Particle emissions associated with flyash loading and unloading can be estimated using 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 the combustion of 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 combusted. 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 the combustion of coal (as in the study) was wetted. Flyash from the hazardous waste combustion facility may not be wetted depending on the facility.

Step 2: Estimating the Flyash Generation Rate - The flyash generation rate from the APCD can be obtained from the Part B Permit Application and 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 is a conservatively high estimate of the actual flyash generation rate.

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

Example Calculation

The fugitive ash emission rate is calculated by multiplying 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]. Accounting for the air pollution control equipment, the product of Steps 1 and 2 is multiplied times one minus the fabric filter efficiency (Step 3) to obtain 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 1993i), and (2) the more recent regulatory determination of CKD (60 FR 7366, February 7, 1995).

Most CKD constituents (for example, metals) are not volatile but could be released to air through fugitive dust emissions as a volatile or semivolatile organic that can be released 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) nonerodible elements, such as clumps of grass or stones on the pile, (3) 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 (1993i), 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 (1) infrequent application of water, and (2) 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. It should be noted that a crust does not always form for a variety of reasons such as weather and chemistry of the CKD.

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

U.S. EPA evaluated the potential direct and indirect risks resulting from on-site and off-site management of CKD (U.S. EPA 1993i; 1993j). U.S. EPA (1993i; 1993j) highlighted the limited amount of available information regarding the variation in 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 that is recycled back into the system and the "as-managed" CKD that is disposed on or offsite.

The air exposure pathway is 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 may 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 respirable concern by humans. Virtually all of the dust generated at the 15 facilities evaluated by U.S. EPA (1993i) 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 RISK ASSESSMENT REPORT

- Physical data, including particle size distribution and density
- Chemical data, including organic and inorganic analytical tests similar to those used for sampling combustion gases
- Plant net CKD generation rate (how much CKD per year that is available for disposal)
- Ambient air monitoring data
- CKD management, transportation, storage, and disposal methods
- Containment procedures, including fugitive dust prevention measures and the area of exposed CKD
- Meteorological data, including wind speed and precipitation

2.2.8.2 Estimation of CKD Fugitive Emissions

In general, this guidance does not address CKD risks in a quantitative fashion. However, risk assessments conducted for cement manufacturing facilities should, at a minimum, evaluate the fugitive emissions due to CKD on a qualitative basis. Readers are referred to the *Technical Background Document for the Report to Congress* (U.S. EPA 1993i), for methods to estimate the magnitude of fugitive emissions from the handling, storage, and disposal of CKD. A qualitative evaluation can be performed 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 (1993i) and the more recent regulatory determination of CKD (60 FR 7366, February 7, 1995). If the risks are equivalent, or the risks attributed to the CKD are greater than the risks estimated for the kiln stack emissions, the permitting authority may decide to evaluate the risk from CKD emissions in a more quantitative fashion. The permitting authority should ensure that any qualitative evaluation includes a comparison of the conditions at the facility to the conditions at the model facilities evaluated by U.S. EPA (1993i; 1993j). In addition, an analysis of a specific facility's compliance with other environmental statutes and regulations may be an appropriate method to qualitatively evaluate risks associated 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 one definition of a COPC, because a compound that is a COPC at one hazardous waste combustion unit may not be a COPC at another combustion unit. COPCs in the emissions from hazardous waste combustion units vary widely, depending on (1) the type of combustion unit, (2) the type of hazardous waste feed being burned, and (3) the type of APCS used. COPCs are metals and/or PICs. PICs are any organic compounds emitted from a stack, such as (1) compounds initially present in the hazardous waste feed stream and not completely destroyed in the combustion process, and (2) compounds that are formed during the combustion process. Because PICs may be formed by trace toxic organic compounds in the waste feed stream, these compounds should be evaluated as PIC precursors, in addition to those compounds that constitute most of the hazardous waste feed.

PICs should not be confused with principal organic hazardous constituents (POHC), which are compounds in the waste feed stream used to measure DRE of the combustion unit during a trial burn test. Unburned POHCs and partially destroyed or reacted POHCs are PICs, but PICs are not necessarily related to POHCs. COPCs have typically been identified by U.S. EPA in seven different general categories (U.S. EPA 1994g; 1994i; 1994j; 1994n):

- Polychlorinated dibenzo(p)dioxins (PCDD) and polychlorinated dibenzofurans (PCDF)
- Polynuclear aromatic hydrocarbons (PAH)
- Polychlorinated biphenyls (PCB)

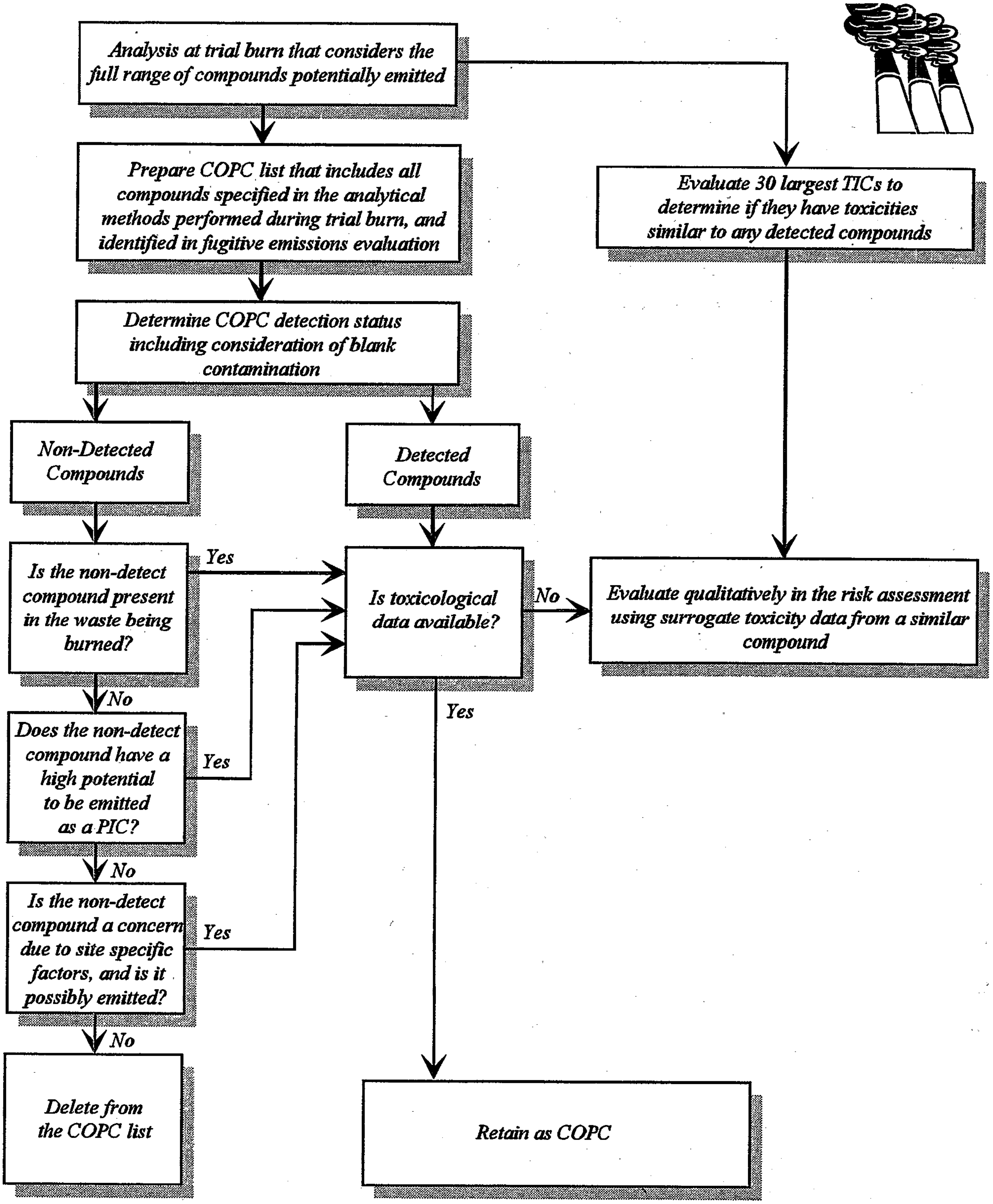
- Nitroaromatics
- Phthalates
- Other organics
- Metals

Table A-1 (Appendix A) presents a comprehensive list of compounds typically identified (1) in hazardous waste, and (2) in hazardous waste combustion stack gas emissions. For each compound, Table A-1 identifies the Chemical Abstracts Service (CAS) number 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 (1) U.S. EPA and state risk assessment reference documents, (2) emission test results that have identified the compound in the emissions from hazardous waste combustion facilities, or (3) other literature that suggests that the risks from the compound may be significant. Table A-1 has been provided in this guidance in order to help risk assessors ensure that the trial burn considers the full range of compounds potentially emitted from a combustion unit and the appropriate analytical method. Once the trial burn stack tests are completed, the COPC selection process is initiated based on the universe of stack test data, not Table A-1. The purpose of a risk assessment is not to evaluate every potential metal and PIC listed in Table A-1.

COPCs are identified from the trial burn data based on their potential to pose increased risk or hazard via one or more of the direct or indirect exposure pathways. This identification process should focus on compounds that (1) are likely to be emitted, based on the presence of the compound or its precursors in the waste feed, (2) are potentially toxic to humans, and/or (3) have a definite propensity for bioaccumulating or bioconcentrating in human and ecological food chains. Appendix A discusses further carcinogenic and noncarcinogenic toxicity of specific 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, permitting authorities should consult with the most current version of EPA's Integrated Risk Information System (IRIS) and *Health Effects Assessment Summary Tables* before completing a risk assessment to ensure that the toxicity data used in the risk assessment is based upon the most current Agency consensus.

As illustrated in Figure 2-3, seven steps should be followed to identify the COPCs that will be evaluated for each facility (U.S. EPA 1993h; 1994i). For each of the following steps, a sample table—based on

FIGURE 2-3
COPC IDENTIFICATION



data from an existing facility—has been included in Appendix A1.15 as an example to illustrate the completion of each step.

Step 1: Evaluate analytical data from the stack tests performed during the trial burn and compounds associated with fugitive emissions (see Section 2.2.5). Prepare a list which includes all the compounds specified in the analytical methods performed in the trial burn, and fugitive emission evaluation. Describe whether the compound was detected or not detected.

A detection in any one of the sample components (e.g., front half rinse, XAD resin, condensate, Tenax tube) in any run constitutes a detection for that specific compound. Evaluation of blank contamination results, included in the quality assurance (QA) data section of the trial burn report, should be considered when determining the non-detect status of the compounds (see Section 2.5).

See Table A1.9-1 which was developed from an actual trial burn using Methods 0030, 0010, and 23. Metals results were based on waste feed sampling.

Step 2: Evaluate the type of hazardous waste burned in the combustion unit—including all wastes that the unit will be permitted to burn—to determine whether any of the non-detect compounds should be retained for evaluation as COPCs because they are present in the waste.

For example, if a facility is permitted to burn explosives which characteristically include nitroaromatic compounds, yet the stack test showed non-detect status for all nitroaromatic compounds, nitroaromatic compounds should still be evaluated in the risk assessment. This evaluation should also consider other materials fed to the combustion unit (for example, raw materials or coal in a cement kiln). Regardless of the type of hazardous waste being burned in the combustion unit, every risk assessment should include PCDD/PCDFs and PAHs (the rationale for including these compounds is discussed in greater detail in Sections 2.3.1 and 2.3.2).

Table A1.9-2 lists the compounds that an example facility expects to burn based on process knowledge. Waste feed analytical data was not used to develop Table A1.9-2 because the detection limits for the waste feed were 100 mg/kg for the semivolatile organics, and 50 to 250 mg/l for the volatile organics.

Table A1.9-3 identifies in bold lettering the compounds from Table A1.9-1 that are in the waste feed. In this example, these compounds include toluene, 2-butanone, 4-methyl-2-pentanone, and the metals cadmium and nickel.

Step 3: Delete from the list of COPCs those compounds that are non-detect, are not components of any combustion unit feed stream, and do not have toxicological data. From compounds that are detected but have no toxicological data, evaluate using surrogate toxicity data from a similar compound and retain on the COPC list.

However, COPCs that are evaluated on the basis of surrogate toxicity should not be quantitatively evaluated in the risk assessment. These COPCs should be evaluated qualitatively in the uncertainty section of the report.

Table A1.9-4 is an example of non-detected compounds with no toxicological data that have been deleted from the COPC list, as indicated by striking-out these compounds. For this example, toxicological data is available for all of the detected compounds.

Step 4: Delete from the list of COPCs those compounds that are non-detect, are not components of any combustion unit feed stream, and do not have a high potential to be emitted as PICs.

Although some compounds (nitroaromatics, pthalates, hexachlorobenzene, and petachlorphenol) have traditionally been automatically identified as PICs in previous U.S. EPA guidance, inclusion of these compounds should be based on consideration of potential to be emitted and waste feed composition (e.g., nitrogenated wastes, plastics, or highly chlorinated organic waste streams) (see Sections 2.3.4 through 2.3.6).

Table A-1.9-5 of the example shows additional compounds deleted following Step 4. Note that 2,4-Dinitrotoluene was not deleted because the example facility burns toluene and wastes with significant amounts of nitrogen.

Step 5: Evaluate the 30 largest 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, consider surrogate toxicity data, as recommended in Step 3.

Step 6: Evaluate any compound that may be of concern due to other site-specific factors (e.g., community and regulatory concern, high background concentrations). Include as COPCs those compounds that (1) are a concern due to site-specific factors, and (2) may be emitted by the combustion unit.

If the compound in question does not have a reasonable potential of being present in the stack emissions, the risk assessment report should justify this assertion. This information will provide the risk manager with the information necessary to conclude that the facility has not overlooked a serious risk.

No additional COPCs were identified for the example facility based on evaluation of the 20 largest TIC peaks (Step 5) or on site-specific factors (Step 6). Therefore, Table A-1.9-5 depicts the final list of COPCs for the example facility following the COPC identification process.

Previous U.S. EPA guidance (1989e; 1993h; 1994j; 1994n) has recommended that for indirect exposure analysis, the COPC list consist of those constituents considered to present the most significant risks. These constituents were selected on the basis of (1) the quantity of the hazardous waste to be burned, (2) the toxicity of the hazardous waste to be burned, and (3) the potential for the hazardous waste to bioaccumulate. For direct exposure analysis, however, it was recommended that all constituents for which stack emission data and inhalation health benchmarks exist should be included. U.S. EPA OSW is now recommending that one COPC list be developed which applies to both indirect and direct exposure analysis. U.S. EPA OSW believes that risk assessors can complete spreadsheet-based risk calculations for all COPCs listed as a result of the identification process, thereby providing for an efficient use of facility and regulatory resources. This approach should help minimize public concern over the exclusion of some COPCs and should reduce confusion for those interested in reviewing the results of the risk assessment.

The following subsections provide specific information and guidance on identifying COPCs for each facility—with specific discussions provided for classes of compounds that U.S. EPA guidance documents have recommended for automatic inclusion in all risk assessments. Because U.S. EPA's boiler and industrial furnace (BIF) regulations also regulate emission rates of PM and hydrochloric acid/chlorine gas, the risks associated with these compounds are also discussed. There is also a discussion of the emerging issues surrounding the class of compounds referred to as "endocrine disruptors."

The following subsections also focus on compounds that can drive risk assessments as indicated by past experience. These compounds include polychlorinated dibenzo(p)dioxins and dibenzofurans, polynuclear aromatic hydrocarbons, polychlorinated biphenyls, nitroaromatics, phthalates, hexachlorobenzene and pentachlorophenol, and metals. Volatile organic compounds are also discussed. Specific issues that affect the COPC identification process and evaluation of these compounds in the risk assessment are discussed.

U.S. EPA OSW recognizes that, for many compounds, only limited information is available regarding potential health effects. In addition, for those chemicals for which health effects have been identified, the relationship between dose and response may be poorly understood. U.S. EPA OSW advocates 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, permitting authorities should consult with the most current version of EPA's Integrated Risk Information System (IRIS) (www.EPA.gov/IRIS) before completing a risk assessment to ensure that the toxicity data used in the risk assessment is based upon the most current Agency consensus.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

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

2.3.1 Polychlorinated Dibenzo(p)dioxins and Dibenzofurans

Consistent with previous U.S. EPA guidance (U.S. EPA 1993h, 1994i, 1994j, 1994n, and 1994r), PCDDs and PCDFs should be included in every risk assessment. One mode in which PCDDs and PCDFs form in dry APCSs is fly ash catalyzed reactions between halogens and undestroyed organic material from the furnace. 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. In recent years, as chemical analytical methods have become more sensitive, additional sources of PCDDs and PCDFs have been identified, including (1) effluent from paper mills that use chlorine bleaches, and (2) combustion sources, including forest fires, municipal waste and medical incinerators, and hazardous waste combustion units. Duarte-Davidson et al. (1997) noted that the combustion of 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.

PCDDs and PCDFs are formed at these combustion sources from the reaction of chlorine-containing chemicals and organic matter. Predicting the production 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 Robert 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 organic compound 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. 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 combustion units 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 needed to enable completion of a more refined risk assessment at each combustion unit.

In evaluating fate-and-transport pathways, 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 of the PCDDs and PCDFs. Evaluation of ambient air monitoring studies, in which researchers evaluated the partitioning of dioxin-like compounds between the vapor and particle phases, suggests that the higher chlorinated congeners (the hexa through hepta congeners) were principally sorbed to airborne particulates, whereas the Tetra and penta congeners were significantly, if not predominantly, partitioned to the vapor phase (U.S. EPA 1994k). This finding is consistent with vapor/particle partitioning as theoretically modeled in Bidleman (1988). Dioxin-like compounds exhibit little potential for significant leaching or volatilization after they have been sorbed to particulate matter (U.S. EPA 1994k).

Information in U.S. EPA (1994a) 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. The following subsections clarify the procedures for estimating risks associated with PCDDs and PCDFs, to be used in conjunction with the procedures described in Chapter 7. U.S. EPA OSW is also aware of growing concern regarding the risks resulting from (1) fluorine- and bromine-substituted dioxins and furans, and (2) sulfur analogs of PCDDs and PCDFs. Although research regarding these compounds is ongoing, the following subsections also discuss U.S. EPA guidance on how to evaluate these compounds as potential COPCs.

2.3.1.1 PCDD/PCDF Cancer Risks

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

Dioxin Congener	TEF (unitless)	Furan Congener	TEF (unitless)
2,3,7,8-Tetrachlorodibenzo(p)dioxin	1.000	2,3,7,8-Tetrachlorodibenzofuran	0.100
1,2,3,7,8-Pentachlorodibenzo(p)dioxin	0.500	1,2,3,7,8-Pentachlorodibenzofuran	0.050

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

Source: U.S. EPA (1994l)

The combined risk resulting from exposure to a mixture of the 17 dioxin-like congeners can be computed using the TEFs and assuming that the risks are additive. To estimate the exposure media concentration, U.S. EPA OSW recommends that a risk assessment for PCDDs and PCDFs be completed using the congener-specific emission rates from the stack and fate and transport properties in the media concentration equations (see Appendix B). The exposure media concentrations of the individual congeners should then be converted to a 2,3,7,8-TCDD Toxicity Equivalent (TEQ) by multiplying by the congener specific TEFs, as previously discussed. The lifetime average daily dose (LADD) can then be estimated for a 2,3,7,8-TCDD TEQ, according to the procedures described in Chapter 7. Cancer risk on a 2,3,7,8-TCDD TEQ basis can be assessed using the cancer slope factor for 2,3,7,8-TCDD, in combination with the 2,3,7,8-TCDD TEQ-based LADD.

2.3.1.2 PCDD/PCDF Noncancer Hazards

U.S. EPA typically evaluates noncancer effects of chemicals by comparing exposure levels to health based reference doses, which are levels the Agency considers in evaluating potential impacts on human health. However, for reasons discussed in the Agency's Draft Dioxin Reassessment (U.S. EPA 1994a), U.S. EPA has not developed reference doses for any of the PCDD or PCDF congeners.

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 exposures estimated to result from a facility's emissions with national average background exposure levels for these compounds (1 to 3 pg TEQ/Kg/day for adults and 60 pg TEQ/Kg/day for nursing infants). If exposures due to the facility's emissions during the exposure duration of concern are low compared to background exposures, then the emissions are not expected to cause noncancer effects.

U.S. EPA OSW recommends that risk assessments include a comparison of exposures to PCDDs and PCDFs from a facility's emissions over the exposure duration of concern with national average background exposure levels, using 1 pg/kg/day for adults. In the future, the Agency may develop alternative approaches to evaluate noncancer effects from exposures to PCDDs and PCDFs; in that case, these approaches should be included in future risk assessments.

2.3.1.3 Fluorine, Bromine, and Sulfur PCDD/PCDF Analogs

U.S. EPA (U.S. EPA 1996h; 1996m) 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. Available information indicates that fluorinated dioxins and furans are not likely to be formed as PICs; however, the presence of free fluorine in the combustion gases may increase the formation of chlorinated dioxins (U.S. EPA 1996h). U.S. EPA OSW is not aware of any studies conducted to evaluate this relationship. Available information indicates the potential for the formation of brominated or chlorobrominated dioxins (U.S. EPA 1996i). Although the likelihood of the formation or toxicity of these compounds is not currently well understood, U.S. EPA OSW recommends that the potential formation of these compounds be evaluated in the risk assessment uncertainty analysis (see Chapter 8). U.S. EPA 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 to indicate 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, which contain two carbon-sulfur bonds in the central ring of the structure (U.S. EPA 1996h). The likelihood of the formation or associated toxicity of these compounds is not currently well understood. Therefore, a quantitative toxicity assessment of fluorine, bromine, and sulfur analogs is not required for inclusion in the risk assessment report. Instead, the uncertainty section (Chapter 8) of the risk assessment report should discuss the potential for the formation of these analogs. It should be noted that there is currently no U.S. EPA approved method for the sampling or analysis of these dioxin analogs. The use of the method for total organics (see Section 2.2.1.3) is currently recommended to account for the potential presence of these compounds.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Description of any combustion unit-specific operating conditions that may contribute to the formation of dioxins
- Any facility specific sampling information regarding PCDD and PCDF concentrations in air, soil, water, or biota
- Information regarding the concentration of sulfur, fluorine, and bromine in the combustion unit feed materials

2.3.2 Polynuclear Aromatic Hydrocarbons

Consistent with previous U.S. EPA guidance (U.S. EPA 1993h, 1994i, 1994j, 1994r), PAHs should be evaluated as COPCs in the risk assessment. 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. U.S. EPA considers all of these compounds to be carcinogenic; all except chrysene are known to be animal carcinogens. However, an oral cancer slope factor is only available for one PAH, benzo(a)pyrene.

PAHs are readily formed in combustion units by either (1) dechlorination of other PAHs present in the waste feed or emissions stream (such as dioxins), 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. Therefore, based on the toxicity and combustion chemistry of PAHs, the absence of these compounds from stack emissions should always be confirmed via stack gas testing.

At present, BaP is the most studied PAH and the only one that has been subjected to oral carcinogenesis bioassays to approximate the National Toxicology Program (NTP) standard (U.S. EPA 1991c). The only other whole animal studies conducted on other class B2 carcinogen PAHs use injection or dermal (skin-painting) dosing. Multiple animal studies in rodent and nonrodent species demonstrate BaP to be carcinogenic following administration by oral, intratracheal, inhalation, and dermal routes; BaP has also produced positive results in several in vitro bacterial and mammalian genetic toxicity assays, in addition to numerous in vivo tests for dioxyribonucleic acid (DNA) damage. BaP is metabolized to reactive electrophiles that are capable of binding to DNA (U.S. EPA 1990h). Therefore, U.S. EPA (1993f) used various nonbioassay results to determine relative potency factors (RPFs) for the class B2 carcinogen PAHs. RPFs for these seven PAHs are as follows:

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

Determining the *SF* for an individual chemical by using a standard carcinogenesis bioassay through the NTP requires the following: (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, as an alternative to the full carcinogenesis bioassay, in vitro studies can be used to compare various PAHs, as have been conducted by Knebel, Aufderheide, and Emura (1994) and many other groups; these studies require a few milligrams of each chemical, a few weeks, and about $1,000 per chemical. Because of this difference in cost-effectiveness, it is not likely that many full carcinogenesis bioassays of PAHs will be carried out in the near future.

Consistent with previous guidance (U.S. EPA 1994g), U.S. EPA OSW recommends utilizing the BaP-RPF method for evaluation of PAHs in the risk assessment. The BaP-RPF method requires that the user adjust the concentrations of the individual PAHs and sum them to obtain an equivalent total concentration of BaP. This summed concentration, the BaP cancer *SF*, and BaP fate-and-transport properties are used to estimate total risk from all carcinogenic PAHs.

It should also be noted that noncarcinogenic health effects, in addition to carcinogenic effects, may be associated with exposure to PAHs. However, RPFs for noncarcinogenic effects of PAHs similar to those developed for potentially carcinogenic PAHs have not been developed. The uncertainties associated with attempting to quantify the potential noncarcinogenic effects of PAHs without RfDs or RfCs is 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 may be emitted, the potential to underestimate the noncarcinogenic health effects associated with exposure to PAHs should be discussed in the uncertainty analysis section of the site-specific risk assessment report.

2.3.3 Polychlorinated Biphenyls

The use 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). 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 manipulation of the reaction conditions, including temperature, pressure, and the ratio of the reactants (Erickson 1992; Grayson 1985).

The most commercially useful property of PCBs is that they are chemically stable in relatively adverse conditions, such as a temperature of several hundred degrees in an oxygen-containing atmosphere; the more chlorinated congeners are more resistant to reaction. Therefore, destruction of PCBs by combustion generally requires conditions of high temperatures (at least 1,200°C) and an extended contact time (more than 2 seconds) in that temperature with adequate oxygen (Erickson 1992).

Limited data and studies, including laboratory and field, show that PCBs may be formed from the combustion of 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 (see definition in Section 2.3.3.1) 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 the combustion of 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 the combustion of highly chlorinated (60 percent or greater chlorine) wastes can produce PCBs.

PCBs should automatically be included as COPCs for combustion units that burn PCB-contaminated wastes or waste oils, highly variable waste streams such as municipal and commercial wastes for which PCB contamination is reasonable, and highly chlorinated waste streams.

Due to the toxicity and uncertainties associated with combustion chemistries the permitting authority may choose to confirm that the absence of these compounds from stack emissions via stack gas testing for units burning hazardous wastes.

2.3.3.1 Carcinogenic Risks

Earlier U.S. EPA guidance has (1994g; 1994i; 1994j; 1994r) recommended that all PCB congeners (209 different chemicals) be treated in a risk assessment as a mixture having a single carcinogenic potency. This recommendation was based on the U.S. EPA drinking water criteria for PCBs (U.S. EPA 1988a), which used available toxicological information with the following limitations:

- The only PCB for which a cancer *SF* had been developed was Aroclor 1260; 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 is 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.

Since the compilation of U.S. EPA (1988a), additional research on PCBs has been reported. The most important result of this research is the demonstration that some of the moderately chlorinated PCB congeners can have dioxin-like effects (U.S. EPA 1992e; U.S. EPA 1994a; ATSDR 1995d). These dioxin-like congeners include (U.S. EPA 1996q):

PCB Congener	TEF (unitless)	PCB Congener	TEF (unitless)
3,3',4,4'-tetrachlorobiphenyl	0.0005	2,3,3',4,4',5-hexachlorobiphenyl	0.0005
2,3,3',4,4'-pentachlorobiphenyl	0.0001	2,3,3',4,4',5'-hexachlorobiphenyl	0.0005
2,3,4,4',5-pentachlorobiphenyl	0.0005	2,3',4,4',5,5'-hexachlorobiphenyl	0.00001
2,3',4,4',5-pentachlorobiphenyl	0.0001	3,3',4,4',5,5'-hexachlorobiphenyl	0.01
2',3,4,4',5-pentachlorobiphenyl	0.0001	2,2',3,3',4,4',5-heptachlorobiphenyl	0.0001
3,3',4,4',5-pentachlorobiphenyl	0.1	2,2',3,4,4',5,5'-heptachlorobiphenyl	0.00001
		2,3,3',4,4',5,5'-heptachlorobiphenyl	0.0001

Source: U.S. EPA (1996q)

The listed congeners have four or more chlorine atoms with 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 then react with the aryl hydrocarbon receptor; this same reaction is believed to initiate the adverse effects of PCDDs and PCDFs. As reported by U.S. EPA (1996q), the World Health Organization has used various test results to derive interim toxicity equivalency factors (TEFs; ranging from 0.1 to 0.00001) for the above-listed PCB congeners. Additional congeners are suspected of producing similar reactions, but there is not yet enough data to derive TEF values for them.

U.S. EPA OSW recommends that permitting authorities estimate 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 (e.g., draft Method 1668), available at most commercial laboratories with dioxin/furan analytical capabilities, should be used to identify the specific concentration of individual coplanar PCBs in stack gas.

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

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
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 per molecule comprise less than 0.5 percent of the total PCBs

An *SF* of 2 (milligrams per kilogram-day)$^{-1}$ will typically be used in most circumstances when conducting a risk assessment. A *SF* if 0.07 (milligram per kilogram-day)$^{-1}$ can be used for adult exposures, when supporting congener specific analyses of emissions by high-resolution gas chromatography/mass spectroscopy or similar means for total PCB concentrations for each mono- through deca-isomer group have demonstrated 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). U.S. EPA OSW does not expect that the 0.4 SF will be widely used in combustion risk assessments, however, the SF of 2 will be used in most risk assessments because the PCB mixture will usually contain 0.5 percent or more PCB congeners with greater than 4 chlorines.

When assessing risks from the coplanar PCB congeners, and when evaluating PCB mixtures containing isomers with more than 4 chlorines in quantities greater than 0.5 percent of the total PCBs, U.S. EPA OSW recommends that the fate and transport properties for Aroclor 1254 be used in the modeling. This approach is reasonable because approximately 77 percent of Aroclor 1254 is composed of PCB congeners with more than 4 chlorines (Hutzinger et al. 1974). When assessing risks from PCB mixtures which contain less than 0.5 percent of PCB congeners with more than 4 chlorines, U.S. EPA OSW recommends that the fate and transport properties of Aroclor 1016 be used in the modeling. This approach is reasonable because approximately 99 percent of Aroclor 1016 is comprised of PCB congeners with 4 or less chlorines (Hutzinger et al. 1974).

2.3.3.2 Potential Non-Cancer Effects

In addition to the *SF*s and associated carcinogenic risk, Aroclor 1254 and Aroclor 1016 have *RfD*s specified in IRIS (U.S. EPA 1996a). When conducting a risk assessment that includes PCBs as a COPC, in addition to carcinogenic risk associated with all PCBs, noncarcinogenic risk should be determined for those Aroclors having *RfD*s. The evaluation of noncarcinogenic risk is similar to the evaluation of carcinogenic risk in that if the PCB mixture contains isomers with more than 4 chlorines in quantities greater than 0.5 percent of the total PCBs, U.S. EPA OSW recommends that the fate and transport properties for Aroclor 1254 be used in the modeling. This approach is reasonable because approximately 77 percent of Aroclor 1254 is composed of PCB congeners with more than 4 chlorines (Hutzinger et al. 1974). When assessing risks from PCB mixtures which contain less than 0.5 percent of PCB congeners with more than 4 chlorines, U.S. EPA OSW recommends that the fate and transport properties of Aroclor 1016 be used in the modeling. This approach is reasonable because approximately 99 percent of Aroclor 1016 is comprised of PCB congeners with 4 or less chlorines (Hutzinger et al. 1974).

The *RfD* for Aroclor 1254 of $2x10^{-5}$ milligrams per kilogram-day will typically be used in most circumstances when conducting a risk assessment. The *RfD* for Aroclor 1016 of $7x10^{-5}$ milligram per kilogram-day can be used when supporting analysis of emissions for each homologue group having demonstrated 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).

2.3.4 Nitroaromatics

Careful consideration should be made before the automatic inclusion of nitroaromatic organic compounds, including 1,3-dinitrobenzene; 2,4-dinitrotoluene; 2,6-dinitrotoluene; nitrobenzene; and pentachloronitrobenzene, in risk assessments for combustion units. These compounds 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, which 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 will not be formed as PICs if they are not 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 be reacted with the nitronium ion, resulting in the attachment of the nitronium ion to the ring. This reaction process is not likely to occur in a hazardous waste combustion unit 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 are not usually formed in a combustion unit environment (if they are, a further thermodynamically favorable reaction will occur, thereby eliminating the nitronium ion) (Hoggett, Moodie, Preston, and Schofield 1971; Schofield 1980; March 1985).

Nitroaromatics should be included as COPCs if the hazardous waste feed streams include nitroaromatic compounds or close relatives (TDA and TDI). Also, combustion of feed streams containing unusually high amounts of fuel-bound nitrogen (greater than 5 percent) may lead to increased levels of nitrogenated PICs (U.S. EPA 1994j). Examples of waste feeds identified include heavy distillation fractions and bottoms streams from the production of coal tars and petroleum distillation. 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). Sampling for hydrogen cyanide is also recommended (U.S. EPA 1994j).

It should also be noted, that earlier U.S. EPA guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) has recommended that risk assessments always include nitroaromatic organic compounds, including 1,3-dinitrobenzene; 2,4-dinitrotoluene; 2,6-dinitrotoluene; nitrobenzene; and pentachloronitrobenzene. However, U.S. EPA OSW no longer recommends automatic inclusion of nitroaromatic organic compounds in risk assessments. Rather, this guidance recommends that careful consideration should be given to including nitroaromatics as COPCs based on the information presented above.

2.3.5 Phthalates

Careful consideration should be made before the automatic inclusion of phthalates, including bis(2-ethylhexyl)phthalate (BEHP) and di(n)octyl phthalate (DNOP), in risk assessments for combustion units. 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, which is 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 micrograms (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 are synthesized by reacting alcohol with phthalic anhydride in the presence of an acidic catalyst in a nonaqueous solvent (ATSDR 1993; ATSDR 1995b). Phthalates and their predecessors are readily combusted compounds, 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 the combustion of other chemical compounds.

Therefore, phthalates are very unlikely to be emissions from a combustion unit, 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.

The evaluation of phthalate plasticizers in risk assessments should not be automatically discounted due to the toxicity and biaccumulative potential of these compounds. Moreover, the uncertainties associated with combustion chemistry suggest that the absence of these compounds from stack emissions should always be confirmed via stack gas testing rather than process knowledge or waste feed characterization data. Based on the findings of liver tumors in long-term animal carcinogenicity studies, U.S. EPA has classified BEHP as a "probable human carcinogen" (class B2) (NTP 1982). BEHP has been presumed to have a high tendency to bioaccumulate (based solely on the log of the octanol-water coefficient (log K_{ow}) value (Mackay, Shiu, and Ma 1992; Karickoff and Long 1995). Based on its ubiquity, its B2 classification, and its high tendency to bioaccumulate, BEHP has been placed on most of the U.S. EPA 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). It should also be noted that earlier U.S. EPA guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) has recommended that BEHP and DNOP always be included in every risk assessment. However, U.S. EPA OSW no longer recommends automatic inclusion of phthalates in risk assessments. Rather, this guidance recommends that careful consideration should be given to including phthalates as COPCs based on the information presented above.

In evaluating BEHP in the risk assessment, consistent with U.S. EPA (1995h), U.S. EPA OSW recommends a metabolism factor (*MF*) of 0.01 for bis(2-ethylhexyl)phthalate (BEHP), and 1.0 for all other COPCs. The *MF* represents the estimated amount of COPC that remains in fat and muscle. Based on a study by Ikeda et al. (1980), U.S. EPA (1995h) utilized 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). Considering the recommended values for this variable, *MF* has no quantitative effect on A_{beef} (with the exception of BEHP).

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 in this guidance are intake driven, the use of a metabolism factor applies only to food sources used in evaluating indirect human exposure, including ingestion of beef, milk, and pork. In summary, use of a *MF* does not apply for direct exposures to air, soil, or water, or to ingestion of produce, chicken, or fish. The use of a *MF* is further discussed in Section 5.4.4.7 and Appendix B, Table B-3-10.

2.3.6 Hexachlorobenzene and Pentachlorophenol

Careful consideration should be made before the automatic inclusion of hexachlorobenzene and pentachlorophenol in risk assessments for combustion units. Hexachlorobenzene and pentachlorophenol, like all chlorinated aromatics, are synthesized by the reaction of elemental chlorine with the 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 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). 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). The combustion properties of these chlorinated compounds indicate that they are not likely to be formed as PICs if they are not present in the waste feed stream.

Hexachlorobenzene and pentachlorophenol should be included as COPCs for units that burn waste feeds containing hexachlorobenzene and pentachlorophenol, wood preservatives, pesticides, and highly variable waste streams such as municipal solid waste. However, precluding these compounds from analytical testing during the trial burn based on process knowledge and waste feed characterization is not recommended. Because PCDDs and PCDFs can be formed from fly ash-catalyzed reactions between halogens and undestroyed organic material from the furnace, U.S. EPA guidance (U.S. EPA 1993h; 1994i) has recommended that potential precursor compounds be included in the risk assessment and trial burn (see Section 2.3). These precursor compounds may include chlorinated phenols (such as pentachlorophenol) and chlorinated aromatics (such as hexachlorobenzene). Furthermore, the toxicity and uncertainties associated with combustion chemistry suggest that the absence of these compounds from stack emissions should always be confirmed via stack gas testing.

It should also be noted, that earlier U.S. EPA guidance (U.S. EPA 1994g; 1994i; 1994j; 1994r) has recommended that risk assessments always include hexachlorobenzene and pentachlorophenol. However, U.S. EPA OSW no longer recommends automatic inclusion of hexachlorobenzene and pentachlorophenol in risk assessments. Rather, this guidance recommends that careful consideration should be given to including hexachlorobenzene and pentachlorophenol as COPCs based on the information presented above.

2.3.7 Volatile Organic Compounds

U.S. EPA (1990e) reported that volatile organics (based on Freeman 1988 and 1989) listed as probable PICs produced by the combustion of 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 for excluding these COPCs from evaluation is that there is no empirical evidence that VOC emissions pose a hazard via indirect pathways. U.S. EPA OSW agrees that it is not aware of any such evidence; however, U.S. EPA OSW is similarly unaware of a lack of evidence to the contrary.

Another argument for excluding these COPCs 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. U.S. EPA OSW disagrees with both aspects of this argument. First, U.S. EPA OSW is not aware of any information or research documenting the fate-and-transport of volatile organic COPCs from hazardous waste combustion units. Second, although U.S. EPA OSW agrees that the toxicity of the parent COPC is irrelevant following transformation, this argument ignores the potential toxicity of the reaction products. U.S. EPA OSW is not aware of any available quantitation methods that may be used to predict atmospheric chemical reactions of this nature, and believes that evaluation of the fate-and-transport of the parent COPC is currently the best available method for conservatively accounting for the potential reaction products to which receptors are ultimately indirectly exposed.

Finally, another basis for excluding these COPCs 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-3). Although U.S. EPA OSW agrees with the basic premise of this issue, it is similarly unaware of any other method for evaluating the potential indirect exposure to volatile organic COPCs or their atmospheric reaction products (empirical data are not available).

To summarize, U.S. EPA OSW agrees in principle that the science regarding the fate-and-transport of volatile organic COPCs in the environment is poorly understood. However, because the potential risk associated with indirect exposure to these COPCs is also poorly understood, U.S. EPA OSW believes that the evaluation of volatile organic COPCs via the indirect exposure pathways—with the proper explanation of the uncertainties associated with this process—provides the most reasonable (based on current science) and conservative estimate of these potential risks. U.S. EPA OSW also believes that the risk equations are set up to address this issue because a calculation cannot be completed unless there are sufficient fate and transport properties for each COPC. If these properties are available, then it is scientifically reasonable to hypothesize that the COPC may be of concern to receptors exposed via various exposure scenarios (see Chapter 4), and that the COPC should be evaluated in the interest of protecting human health and the environment. If the necessary fate and transport properties are not available, then it seems reasonable to exclude that COPC from the risk assessment process. For example, a volatile organic COPC will be present primarily in the vapor phase, therefore particulate deposition and subsequent soil concentration should be minimal or negligible. However, as volatility decreases, the potential for significant particulate deposition and subsequent soil concentration increases. Additionally, if there is no COPC biotransfer value for milk (see Chapter 5), then it can be assumed that, based on current information, the COPC is not of interest for the ingestion of milk in the subsistence farmer exposure pathway (however, the lack of fate and transport data does not automatically equate to an absence of potential exposure and risk). This principle holds true for other variables as well. U.S. EPA OSW recommends that as long as there are sufficient fate and transport properties available, the calculations for each exposure pathway should be completed, and any uncertainties introduced into the risk assessment described in the uncertainty discussion provided in the risk assessment report (see Chapter 8).

Finally, a risk assessment may also account for other organic compounds on the basis of the total organic emissions (TOE) from the hazardous waste combustion unit. The TOE rate is used to account for the unidentified mass of organic compounds in stack emissions because current sampling and analytical methods are not always able to positively identify each organic compound in stack emissions. The methodology for using TOE in a risk assessment is discussed further in Section 2.2.1.3.

2.3.8 Metals

Previous guidance (U.S. EPA 1993h; 1994g; 1994i; NC DEHNR 1997) has recommended 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). In the case of nickel and selenium, U.S. EPA has recommended that these metals be evaluated, to determine whether additional terms and conditions should be incorporated into the permit, by using U.S. EPA's omnibus authority provided under 40 CFR Part 270.32(b)(2) (U.S. EPA 1992a). Facilities may also apply the BIF regulation Tier I assumptions, that assume all metals in the waste feed pass through the combustion unit and APCS and are passed through to the emission stream (U.S. EPA 1992c).

It should be noted, that the presence of metals in the combustion unit's feed stream is not required for inclusion in the risk assessment. Although metals cannot be formed as PICs, U.S. EPA OSW is aware of combustion units with metal emissions resulting from waste feed leaching of stainless steel feed piping.

RECOMMENDED INFORMATION FOR 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 regarding U.S. EPA-recommended procedures for evaluating four metals—chromium, lead, mercury, and nickel—that may be specifically altered during the combustion process.

2.3.8.1 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 1996a). Trivalent chromium (Cr^{+3}), a commonly found less oxidized form of chromium, has not been shown to be carcinogenic in either humans or laboratory animals (U.S. EPA 1996a). U.S. EPA (1990a; 1990b) has indicated that chromium emitted from a combustion unit 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. Therefore, unless site-sampling or process-specific information is provided, the worst-case assumption—that 100 percent of the facility chromium emissions are in the hexavalent form—should be used.

Because medium-specific chromium speciation information is often difficult to obtain, risk assessments should be prepared following the conservative initial assumption that all exposure is to hexavalent chromium. However, U.S. EPA OSW recognizes that chromium may exist partially or in some cases entirely as trivalent chromium in various media. For example, as stated in *Casarett and Doull's Toxicology* (Amdur et al. 1991), ". . . hexavalent chromium readily crosses cell membranes and is reduced intracellularly to trivalent chromium" and, therefore, ". . . chromium in biological materials is probably always trivalent."

Therefore, in the event risks or hazards associated with chromium exceed target levels based on the initial conservative assumption that exposure is entirely to hexavalent chromium, risks and hazards may be recalculated assuming potential receptors are exposed through indirect exposure pathways (e.g., ingestion of fish, beef, pork, chicken, dairy products, and produce) to trivalent chromium. These additional risks estimates may then be presented in the report with the hexavalent chromium estimates, and discussed in the uncertainty section of the risk assessment report.

The assumption that receptors are exposed through direct exposure pathways (e.g., inhalation of air) to hexavalent chromium should be maintained in the absence of site specific data. However, permitting authorities may prepare supplemental calculations (that is, in addition to the calculations described above) considering chromium speciation at the points of potential exposure.

2.3.8.2 Lead

U.S. EPA does not currently list an RfD or RfC for lead, because a threshold level for exposure to lead has not been established. Based on findings that neurobehavioral effects have been observed in children with blood lead levels below those that have caused carcinogenic effects in laboratory animals, a cancer *SF* has not been derived by U.S. EPA. U.S. EPA has relied on the neurological effects observed in children as the sensitive endpoint for evaluating lead toxicity. Consequently, U.S. EPA has developed the integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children. This model evaluates potential risks based on predicted blood lead levels associated with exposure to lead (U.S. EPA 1994e), and was developed through the efforts of U.S. EPA (1990c) and Kneip, Mallon, and Harley (1983). The IEUBK model integrates several assumptions about the complex exposure pattern 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 IEUBK model has been reviewed and recommended by the U.S. EPA Science Advisory Board (U.S. EPA 1992b) and by U.S. EPA's Technical Review Workgroup for Lead.

U.S. EPA 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 1994f). The IEUBK computer model cannot predict potential blood lead levels in adults. U.S. EPA 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, U.S. EPA'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).

Based on this information, U.S. EPA OSW recommends that risk assessments evaluating lead as a COPC use the IEUBK model instead of evaluating carcinogenic risks or noncarcinogenic hazards. When run with standard recommended default values (these generally represent national averages, or "typical" values), U.S. EPA's IEUBK model predicts that no more than 5 percent of children exposed to a lead concentration in soil of 400 mg/kg will have lead concentrations in blood exceeding 10 μg/dL (U.S. EPA 1994e and 1994o).

2.3.8.3 Mercury

Consistent with earlier guidance (U.S. EPA 1993h, 1994i, 1994j, 1994r), U.S. EPA OSW recommends that mercury be evaluated as COPCs in the risk assessment. Air emissions of mercury contribute to local, regional, and global deposition. The U.S. Congress explicitly found this to be the case and required U.S. EPA to prioritize maximum achievable control technology (MACT) controls for mercury (U.S. Congress 1989).

Anthropogenic mercury releases are thought to be dominated on the national scale by industrial processes and combustion sources that release mercury into the atmosphere (U.S. EPA 1997d). Stack emissions containing mercury include both vapor and particulate forms. Vapor mercury emissions are thought to include both elemental (Hg^0) and oxidized (e.g., Hg^{+2}) chemical species, while 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 1997d). While coal combustion is responsible for more than half of all emissions of mercury in the U.S. anthropogenic sources, the fraction of coal combustion emissions in oxidized form is thought to be less that from waste incineration and combustion (U.S. EPA 1997d).

The analytical methods for mercury speciation of exit vapors and emission plumes are being refined, and there is still controversy in this field. Chemical reactions occurring in the emission plume are also possible. The speciation of mercury emissions is thought to depend on the fuel used, flue gas cleaning, and operating temperatures. The exit stream is thought to range from almost all divalent mercury to nearly all elemental mercury; with true speciation of mercury emissions from the various source types still uncertain and thought to vary, not only among source types, but also for individual plants as feed stock and operating conditions change (U.S. EPA 1997d). Most of the total mercury emitted at the stack outlet is found in the vapor phase; although exit streams containing soot or particulate can bind up some fraction of the mercury (U.S. EPA 1997d). Total mercury exiting the stack is assumed to consist of elemental and divalent species, with no emissions of methylmercury assumed. The divalent fraction is split between vapor and particle-bound phases (Lindqvist et al. 1991). Much of the divalent mercury is thought to be mercuric chloride ($HgCl_2$) (U.S. EPA 1997d); this is particularly the case for the combustion of wastes containing chlorine.

It should be noted that data on mercury speciation in emissions exiting the stack is very limited, as well as, the behavior of mercury emissions close to the point of release has not been extensively studied. This results in a significant degree of uncertainty implicit in modeling of 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. Discussions of uncertainty and sensitivity analyses of several of the assumptions used in the modeling of mercury emissions are presented in the *Mercury Study Report to Congress* (U.S. EPA 1997d).

Phase Allocation and Speciation of Mercury Exiting the Stack

As discussed above, stack emissions are thought to include both vapor and particle-bound forms; and speciated as both divalent and elemental mercury. Based on review of mercury emissions data presented for combustion sources in U.S. EPA (1997d) 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, at this time U.S. EPA OSW recommends a conservative approach that assumes phase allocation of mercury emissions from hazardous waste combustion of 80 percent of the total mercury in the vapor phase and 20 percent of total mercury in the particle-bound phase. 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 conservative, 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.

As indicated in the global cycle mass percentages in Figure 2-4, mercury exits the stack in both the elemental and divalent vapor forms. Based on U.S. EPA (1997d), a vast majority of mercury exiting the stack does not readily deposit and is transported outside of the U.S. or vertically diffused to the free atmosphere to become part of the global cycle (see Figure 2-4). The divalent form emitted, either in the vapor phase or particle-bound, are thought to be subject to much faster atmospheric removal than elemental mercury (Lindberg et al. 1992; Peterson et al. 1995; and Shannon and Voldner 1994). In addition, vapor phase divalent mercury is thought to be more rapidly and effectively removed by both dry and wet deposition than particle-bound divalent mercury, as a result of the reactivity and water solubility of vapor divalent mercury (Lindberg et al. 1992; Peterson et al. 1995; and Shannon and Voldner 1994).

Vapor Phase Mercury

As illustrated in Figure 2-4, of the 80 percent total mercury in the vapor phase, 20 percent of the total mercury is in the elemental vapor form and 60 percent of the total mercury is in the divalent vapor form (Peterson et al. 1995). A vast majority (assumed to be 99 percent) of the 20 percent vapor phase elemental mercury does not readily deposit and is transported outside of the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle (U.S. EPA 1997d). Only a small fraction (assumed to be one percent) of vapor-phase elemental mercury either is adsorbed to particulates in the air and is deposited or converted to the divalent form to be deposited (assumed to be deposited as elemental mercury, see Figure 2-4). Of the 60 percent vapor phase divalent mercury, about 68 percent is deposited and about 32 percent is transported outside of the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle (U.S. EPA 1997d).

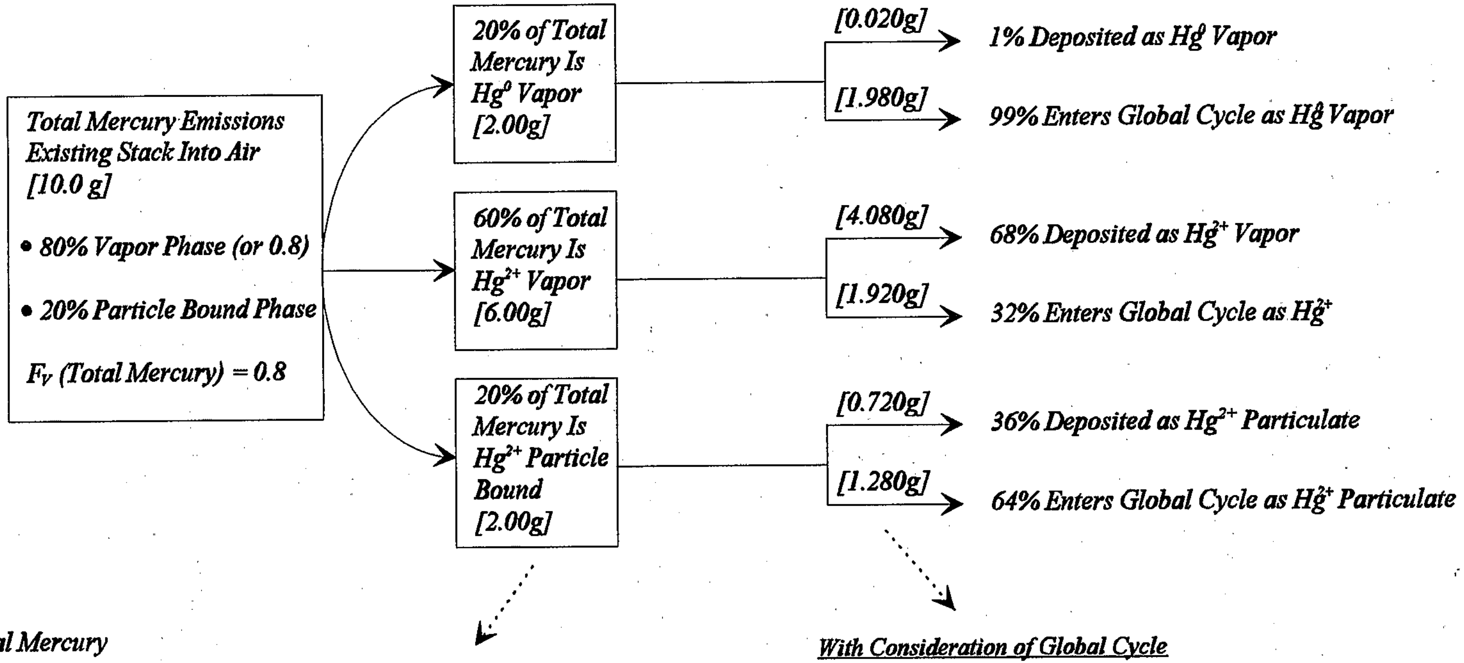
Particle-bound Mercury

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. U.S. EPA (1997d) indicates that only 36 percent of the particle-bound divalent mercury is deposited, and the rest is either transported outside of the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

Deposition and Modeling of Mercury

Consistent with U.S. EPA (1997d) and as shown in Figure 2-4, it is assumed that deposition to the various environmental media is entirely divalent mercury in either the vapor or particle-bound form. Without consideration of the global cycle, mercury speciations will result in 80 percent of the total mercury emitted being deposited as divalent mercury and the remaining 20 percent being deposited as elemental mercury.

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



LEGEND

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

Without Consideration of Global Cycle

- *80% of Total Mercury Emitted is Deposited as Hg^{2+} [(6g + 2g) / 10g]*
- *20% of Total Mercury Emitted is Deposited as Hg^0 [2g / 10g]*

Calculated F_V

- *F_V (Hg^{2+}) = [6g / (6g +2g)] = 0.75*
- *F_V (Hg^0) = [2g / 2g] = 1.0*

With Consideration of Global Cycle

- *48% of Total Mercury Emitted is Deposited as Hg^{2+} [(4.08g + 0.72g) / 10g]*
- *0.2% of Total Mercury Emitted is Deposited as Hg^0 [0.02g / 10g]*

Calculated F_V

- *F_V (Hg^{2+}) = [4.08g / (4.08g + 0.72g)] = 0.85*
- *F_V (Hg^0) = [0.02g / (0.02g + 0g)] = 1.0*

Compound Specific Emission Rate Q

- *Actual Q (Hg^{2+}) = 48% * Q (Total Mercury)*
- *Actual Q (Hg^0) = 0.2% * Q (Total Mercury*

U.S. EPA recommends utilizing the percentages provided in U.S. EPA (1997d) to account for the global cycle, the percentage of total mercury deposited is 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-3, these speciation splits result in fraction in vapor phase (*Fv*) values of 0.85 (40.8/48.2) 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. Media equations assume pseudo steady-state conditions.

Consistent with U.S. EPA (1997d) and as shown in Figure 2-4, it is assumed that deposition to the various environmental media is entirely divalent mercury in either the vapor or particle-bound form. Therefore, divalent mercury is considered for both the indirect exposure and inhalation pathways. A small fraction (assumed to be one percent) of elemental mercury is in the vapor phase and is assumed to be deposited in its original form. Therefore, elemental mercury will only be considered in the inhalation pathway and not the indirect pathways of the risk assessment. Based on these assumptions, human exposure to (1) elemental mercury occurs only through direct inhalation of the vapor phase elemental form, and (2) divalent mercury occurs through both indirect exposure and direct inhalation of the vapor and particle-bound mercuric chloride.

Inhalation of elemental mercury should be assessed using the reference concentration (RfC) for elemental mercury. Exposure to divalent mercury should be assessed using the RfD for mercuric chloride (divalent mercury). Inhalation of divalent mercury should be assessed using the RfC for elemental mercury due to lack of available toxicity data. Appendix A-3 provides the parameter values specific to the various forms of mercury, and Appendix B provides media concentration equations for modeling mercury through the exposure pathways assuming steady-state conditions.

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 1997d). Consistent with U.S. EPA (1997d), a fraction of the divalent mercury that is deposited is assumed to speciate to organic mercury (methyl mercury) in soil. In soil, 98 percent of total mercury is assumed to be divalent mercury and the remaining mass as methyl mercury (U.S. EPA 1997d). 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 be exported to nearby water bodies and potentially bioaccumulated in the aquatic food chain (U.S. EPA 1997d). Therefore, the percentage of methyl mercury in wetland soils is assumed to be higher than the 2 percent assumed for non-wetland soils. However, wetlands soils are not specifically considered in evaluation of 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 1997d). There appears to be a great deal of variability in the processing of mercury among water bodies. This variability is primarily a result of the characteristically wide range of chemical and physical properties of water bodies that influence the levels of methylated mercury. Some of the mercury entering the water body is methylated predominately through biotic processes (U.S. EPA 1997d). In the absence of modeling site-specific water body properties and biotic conditions, consistent with U.S. EPA (1997d), U.S. EPA OSW recommends 85 percent of total mercury in surface water is assumed to be divalent mercury and the remaining mass as methyl mercury.

For most environmental systems, the literature suggests that various physical and chemical conditions may influence the methylation of mercury. Consideration of these conditions, and the magnitude of their potential impact, may be required in some cases to assess the potential for over or under predicting mercury methylation in media and subsequent biotransfer up the food chain. Due to the extreme variance between environmental systems modeled, and at times disagreement, identified in literature reviewed regarding the quantitative influence of specific conditions on methylation, U.S. EPA OSW recommends that extensive research of literature, specific to the conditions prevalent at the site, be conducted before application and deviation from the conservative assumptions recommended above. The following table summarizes the qualitative effect some of the physical and chemical conditions, as reported in literature, may have on methylating:

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

To account for methylation of mercury in the media and its subsequent biotransfer assuming steady-state conditions, the deposition and media concentration equations (presented in Appendix B) have been modified specifically for modeling methyl mercury. Appendix A-3 provides the parameter values specific for methylmercury, and 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 modeling of mercury methylation. To expand on the qualitative information presented in the above table, and better understand conditions that may influence mercury methylation specific to a site, U.S. EPA OSW recommends review of information on this subject presented in the *Mercury Study Report to Congress* (U.S. EPA 1997d).

Conclusion

In the event risks associated with mercury exceed target levels based on modeling with equations and initial conservative assumptions presented in this guidance, the permitting authority may approve use of more complex models that utilize more extensive site-specific data to predict transformation of chemical forms and biotransfer of mercury for evaluation at points of potential exposure. For example, the draft version of the ISCST3 dry gas algorithm for estimating dry gas deposition may be utilized. This draft model can be found on the SCRAM bulletin (see Chapter 3); and specific default parameter values for mercury are presented in U.S. EPA (1997d). While this guidance does not address what models should be used or how data to support such models should be collected, the decision to use site-specific mercury models in a risk assessment is not precluded just because it is different; nor does this guidance automatically approve the use of such models. A permitting authority that chooses to use complex mercury models should carefully identify and evaluate their associated limitations and clearly document this discussion in the uncertainty section of the risk assessment report.

U.S. EPA OSW encourages 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 proposed MACT standards for hazardous waste combustors, or the MACT standards for municipal waste combustors). U.S. EPA OSW acknowledges that site-specific risk assessments as currently conducted may not identify the entire potential risk from mercury emissions. Mercury that does not deposit locally will ultimately enter the global mercury cycle for potential deposition elsewhere.

2.3.8.4 Nickel

U.S. EPA OSW recommends that permitting authorities evaluate nickel as an inhalation carcinogen because some forms of nickel—including nickel carbonyl, nickel subsulfide, and nickel refinery dust—are considered to be carcinogens (U.S. EPA 1996a). This is contrary to U.S. EPA's previous analysis of the toxicity of nickel emissions from hazardous waste combustion units. These forms of nickel were not considered in the development of 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).

Nickel oxide is a major component of nickel refinery dust (other major components include nickel subsulfide and nickel sulfate) which is identified as a potential human inhalation carcinogen (U.S. EPA 1996a). The components responsible for the carcinogenicity of nickel refinery dust have not been conclusively established (U.S. EPA 1996a). Therefore, because nickel oxide is a major component of nickel refinery dust, and because the reason component of nickel refinery dust causing it to be carcinogenic has not been established, nickel emissions should be evaluated as a potential carcinogen via the inhalation pathway. In addition, nickel oxides can be reduced to nickel sulfates (which are carcinogenic) in the presence of sulfuric acid (Weast 1986). Hazardous waste combustion units which burn wet wastes containing significant amounts of nickel and sulfur may need to be especially concerned with nickel emissions.

OSW recommends that nickel be evaluated as an inhalation carcinogen using the inhalation unit risk factor for nickel refinery dust. However, if the permitting authority has information at points of potential inhalation exposure that demonstrate the absence of nickel refinery dust components or the presence of noncarcinogenic nickel species such as soluble salts, this information may be used as the basis for supplemental noncarcinogenic calculations. For exposure pathways other than inhalation, nickel has not been shown to be carcinogenic (U.S. EPA 1996a) and should be evaluated as a noncarcinogen using the oral RfD for nickel soluble salts, the only available nickel related RfD (see Appendix A-3).

2.3.9 Particulate Matter

Particulate matter (PM) can be classified as aerosols, fogs, fumes, mists, smogs, or smokes, depending on its physical state and origin. PM10 is defined as all condensed material suspended in air that has a mean aerodynamic diameter of 10 micrometers or less; PM2.5 is the part of PM10 that is 2.5 micrometers or less in diameter. Although coarse particles (PM10) come primarily from physical processes (dust, grinding operations, and so on) and fine particles (PM2.5) from chemical processes (combustion facilities, including power plants and diesel-powered vehicles), their biological effects are indistinguishable (U.S. EPA 1997a). Larger particles are filtered out or settle in the nose and throat; some of these are swallowed, with doses so small that their effects are negligible (Dinman 1978). The smaller particles include those that reach the trachea and lungs and can have adverse effects. Research has shown that about 20 percent of 10-micrometer particles and 40 percent of 2.5-micrometer particles survive the body's defenses and reach the lung (Dinman 1978). In contrast, almost all particles of 1 micrometer or less penetrate that far. The recently revised National Ambient Air Quality Standard (NAAQS) (U.S. EPA 1997a) retains the PM10 standard of 150 micrograms per cubic meter ($\mu g/m^3$) for a 24-hour average and 50 $\mu g/m^3$ for an annual average. The new NAAQS for PM2.5, effective September 16,1997, is 65 $\mu g/m^3$ for a 24-hour average and 15 $\mu g/m^3$ for an annual average.

The adverse effects of particulates, however they are defined in terms of size, depend on the chemical nature of the matter (Dinman 1978; Wright 1978). These effects can range from potentially fatal to negligible. This range is exemplified by the different forms of silica (silicon dioxide), as evaluated in the industrial hygiene literature (Stokinger 1981). Cristobalite, also known as calcined diatomaceous earth from its usual preparation method, is one of the three major crystalline forms of silica.

The American Conference of Governmental Industrial Hygienists (ACGIH) (1991) used the results of animal and human studies to recommend a threshold limit value (TLV) of 0.05 mg/m^3 of "respirable dust," which is defined almost identically to PM, for cristobalite. This concentration should not produce adverse effects (in this case pneumoconiosis, or fibers leading to scarring of the lung tissue) in healthy workers exposed 8 hours per day, 5 days per week. These and other studies led to a recommended TLV of 10 mg/m^3 for uncalcined diatomaceous earth, which is an amorphous form of silica. This TLV of 10 mg/m^3 is the generic TLV for "nuisance dusts." Even if particulate material, such as cellulose and flour, have no adverse effects they are assigned the default TLV to minimize visual disturbances and general annoyance. No higher TLVs are assigned to any solid matter, regardless of how innocuous the material is.

Because of this wide range of toxicity, PM concentrations cannot be used to estimate the toxicity of emissions. U.S. EPA OSW does not recommend that PM be evaluated as a separate COPC in the risk assessment. However, PM is quite useful as an indicator variable, because it can be measured in real time and is sensitive to changes in combustion conditions.

2.3.10 Hydrogen Chloride/Chlorine Gas

Hydrogen chloride (which becomes hydrochloric acid when dissolved in water) and chlorine are two of the 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 the two chemicals are similar but not identical (Stokinger 1981; ACGIH 1991).

Hydrogen chloride, as all other strong acids and bases, is an irritant on contact; adverse effects are seen only in the upper respiratory tract (including the nose, mouth, and throat). High concentrations can become corrosive and destroy tissues, producing chemical burns. Repeated exposures can cause chronic irritation (such as pain on shaving), dissolve the teeth, and cause similar local problems. However, these problems generally do not worsen when exposure ceases. Unless it is highly concentrated, ingested hydrochloric acid has only minimal adverse effects. In fact, the stomach secretes about 0.1 normal hydrochloric acid.

Single doses of chlorine are even more irritating than similar doses of hydrogen chloride and affect the upper and lower (trachea and lungs) respiratory tracts. In particular, single large doses (such as seen in gas exposures in World War I soldiers) and repeated doses (as seen in workers in the chlor-alkali industry) can both lead to fibrosis of the lung and impaired respiration.

2.3.11 Criteria Pollutants

The most widely effective regulations issued under the CAA are the NAAQS in 40 CFR 50. These cover the "criteria pollutants", namely: sulfur dioxide, particulate matter, carbon monoxide, ozone, nitrogen dioxide, and lead. Lead has already been covered in Section 2.3.7.2, above, and particulate matter in Section 2.3.8. The remaining criteria pollutants are discussed below in order of their biological class.

Sulfur dioxide, nitrogen dioxide, and ozone are biologically very similar (Amdur et al. 1991), although sulfur dioxide is a reducing agent and the other two are oxidizing agents. Acute doses of all three are irritants to the respiratory system. If concentrations are high, people already in poor health (including the elderly and those with preexisting cardiac or respiratory diseases) will be most impacted due to the additional stress put on their systems by the pollutants. Chronic exposure to lower concentrations will eventually produce adverse effects on the respiratory system, including decreases in lung capacity, inflammation, and aggravation of other problems such as asthma; chronic exposures also impair the immune system (Amdur et al.1991). These effects are generally seen in non-specific ways, such as increases in outpatient visits, hospital admissions, and deaths (Amdur et al. 1991). Nitrogen dioxide and ozone, the oxidizing agents, have an often more serious indirect effect (Amdur et al. 1991). When present in the air with hydrocarbons (which come from natural sources, such as the terpenes from trees, and manmade sources, such as alkanes from gasoline) during daylight, the oxidizers will react to form extremely irritating aldehydes and other components of "photochemical smog". The effects of smog are essentially the same as those from the parent oxidizers, but the reaction products that comprise the smog are more potent than the original ingredients (Amdur et al. 1991).

Carbon monoxide has a definite, well-known chemical mechanism of toxicity. It reacts with hemoglobin to form carboxyhemoglobin, which cannot carry oxygen (Beard 1982). This reaction is irreversible. Therefore either a short-term exposure to a relatively high concentration of carbon monoxide or a long-term exposure to a much lower concentration will produce similar effects (Beard 1982). If a person's blood carboxyhemoglobin reaches 2 percent of the total hemoglobin, the central nervous system is adversely affected (Beard 1982). Higher levels of carboxyhemoglobin produce adverse effects on other systems.

A decision to include the risk from criteria pollutants in the overall quantitative risk assessment for hazardous waste combustion facilities is one that should be made by each individual permitting authority. 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).

2.3.12 Endocrine Disruptors

Endocrine disruptors are chemical compounds that interfere with the endocrine system's normal function and homeostasis in cells, tissues, and organisms. It has been hypothesized by U.S. EPA that endocrine disruptors adversely affect the reproductive system by interfering with the production, release, transport, receptor binding action, or elimination of natural blood-borne hormones and ligands.

Several studies have been conducted and serve as the basis for further experimentation to determine whether the hypothesis is correct. These studies include (1) wildlife reproduction (feminization of birds, alligators, and certain terrestrial mammals); (2) wildlife population ecology (population decline); (3) human reproductive physiology (decreased sperm count in males in industrialized nations); (4) molecular biology (data on receptor-mediated mode of action); and (5) endocrinology (increased understanding of mechanisms of hormone regulation and impacts of perturbations).

Some have attempted to classify chemical compounds as endocrine disruptors; however, several problems have been encountered. Only limited empirical data are available to support the designation of specific chemicals as endocrine disruptors, and some of the data are conflicting. There has been a lack of clear structure-activity relationship among the diverse groups of chemicals considered as endocrine disruptors. There is a lack of unifying dose-response relationship among the diverse group of chemicals. Also, there are multiple modes of action for chemicals that are currently considered as 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 risk resulting from potential endocrine disruptors (U.S. EPA 1996i). Currently, no quantitative U.S. EPA methods exist to specifically address the effects of endocrine disrupters on the human endocrine system in a risk assessment. Because the methods for addressing endocrine disrupters are developing at a rapid pace, permits writers and risk assessors should contact the Economics, Methods and Risk Analysis Division (EMRAD) of the Office of Solid Waste for the latest policy on how to deal with endocrine disrupters in site specific risk assessments. Additional information can also be obtained from review of available publications (e.g., EPA Special Report on Endocrine Disruption) at the web site "www.epa.gov/ORD/WebPubs/endocrine/).

2.3.13 Radionuclides

Radionuclides exist in (1) naturally occurring radioactive materials such as coal and other rocks, as (2) radioactive by-products of industrial processes. This risk assessment guidance does not consider the naturally occurring radioactive materials such as uranium and thorium (and their decay elements) based on U.S. EPA doctrine 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 is radioactive waste and hazardous waste defined by U.S. EPA. Radioactive and mixed waste 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.

U.S. EPA OSW recommends that the combustion of mixed waste and radioactive material should be evaluated in the risk assessment. Cancer risks from exposure to radionuclides should be estimated using the slope factor methodology presented in U.S. EPA's *Estimating Radiogenic Cancer Risks* (U.S. EPA 1997a). Limitations of the slope factor methodology include:

- It assumes a single chemical form which is not necessarily site-specific or most conservative
- Ground surface exposure slope factors are only provided for soil contaminated to an infinite thickness which will over estimate exposure for radionuclides which do not move rapidly through soil
- Slope factors are not available for the submersion in water exposure pathway
- Slope factors include decay chains for a limited number (18) of parent radionuclides, however, these are the most significant decay chains

Direct radiation (e.g., radiation from sealed sources such as instruments that are not released to the environment) does not need to be evaluated in the risk assessment. However, some radioactive materials, such as uranium, also present a noncarcinogenic hazard that should be evaluated. Cancer risk from both radiological and non-radiological contaminants should be presented along side each other in a risk summary table. Results should include a discussion of additivity and the uncertainties of additivity when combining risks from radiological and non-radiological contaminants. A radionuclide should be included as a COPC if it is in the combustion unit's waste feed, and has an available toxicity value (e.g., slope factor). Slope factors, developed by U.S. EPA for over 300 radionuclides, can be obtained from 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 obtain dose and predict cancer risk.

Radionuclide exposure pathways generally evaluated in a human health risk assessment 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 air submersion exposure pathway may not need to be quantitatively evaluated if it can be demonstrated that the risk from this pathway is negligible relative to other exposure pathways. 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 from *Federal Guidance Report #11* (U.S. EPA 1988c) 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)

U.S. EPA OSW recommends using the ISCST3 air dispersion model, utilizing the exponential decay option to calculate air concentrations and ground deposition rates. Intake should 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, to ISCST3, 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).

A special consideration in integrating radioactive materials into risk calculations is related to decay and ingrowth of radionuclides, especially the few decay processes that involve a change of state. Decay should always be considered, both over the air transport time and the surface exposure duration. Ingrowth may be important, and special care must be taken in the use of radionuclide slope factors that include contributions from daughters (‘+D” slope factors). Ingrowth involving change of physical states is another situation that will require special handling 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.

Equations for fate and transport of radionuclides in soil and water should be consistent with those presented for non-radionuclides factoring in decay (and ingrowth if applicable). Food chain biotransfer parameters necessary 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 1994).

USER NOTE

Prescriptive methodology for calculating risk from combustion facilities burning mixed waste is beyond the scope of the current document. The above information is provided to outline the methodology recommended by U.S. EPA OSW.

2.4 ESTIMATES OF COPC CONCENTRATIONS FOR NON-DETECTS

The lowest level of an analyte that can be detected using an analytical method is generally termed the "detection limit." One particularly difficult issue is the treatment of data in the risk assessment that are reported as below the "detection limit." The following subsections (1) define commonly reported "detection limits," (2) describe use in the risk assessment of data reported as non-detect, (3) describe statistical distribution techniques applied to address this issue, (4) summarize U.S. EPA OSW recommendations regarding quantification of non-detect issues in preparation of a risk assessment, and (5) clarify data flagged as estimated maximum possible concentration (EMPC) in the risk assessment.

2.4.1 Definitions of Commonly Reported Detection Limits

U.S. EPA's commonly-used definition for the detection limit for non-isotope dilution methods has been 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 termed the "quantitation limit" or "quantitation level." In practice, numerous terms have been created to describe detection and quantitation levels. The significance and applicability of the more widely reported of these detection and quantitation levels by analytical laboratories are summarized below. These levels—listed generally from the lowest limit to the highest limit—include the following:

- Instrument Detection Limit (IDL) is 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.
- Method Detection Limit (MDL) is 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, and 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, the MDLs are determined on analytical reagents (e.g., water) and not on the matrix of concern. 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).
>
> It should be noted that 40 CFR Part 136 is specific to the Clean Water Act, and therefore, 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 U.S. EPA analytical methods for the determination of several hundred analytes.

- Reliable Detection Level (RDL) is 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). The RDL is a total of 8 standard deviations above the MDL developmental test data (3.14 times 2.623).

- Estimated Detection Limit (EDL) is 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 such as SW-846 Method 8270, even though an EDL is not defined by the method.

  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_{β} = 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_{σ} = The mean analyte relative response factor from the initial calibration

- Practical Quantitation Limit (PQL) is 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).

- Target Detection Limit (TDL) is a quantitation level constructed similar to the PQL.
- Reporting Limit (RL) is a quantitiation level constructed similar to the PQL.
- Estimated Quantitation Limit (EQL) is a quantitiation level constructed similar to the PQL.
- Sample Quantitation Limit (SQL) is 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) is a quantitation pre-set by contract, which may incorporate U.S. EPA (1986d) 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 Use In the Risk Assessment of Data Reported As Non-Detect

In collecting data for use in risk assessments or in setting regulatory compliance levels, the 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. Because measurements made below analytical detection and quantitation levels are associated with increased measurement uncertainty, an understanding of these levels is important to the comprehension of the impact they may have when they are applied.

As a result 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. For the second question, the answer is "somewhere between true zero and the quantitation level applied." For samples obtained during the trial burn that report compounds at below the detection limit, earlier U.S. EPA (1994i) guidance has recommended that emission rates for non-detects be developed using one-half of the "detection limit" and applied in conducting the risk assessment. However, which detection limit should be used has not been explicitly defined or presented in quantitative terms.

To increase consistency and reproducibility in dealing with non-detects, U.S. EPA OSW recommends application of the MDL-derived RDL to quantify non-detects fcr COPCs analyzed with non-isotope dilution methods, and application of the method-defined EDL to quantify non-detects for COPCs analyzed with isotope dilution methods. Procedures for these applications are as follows:

> ***Non-isotope Dilution Methods:*** Non-detects for COPCs analyzed with non-isotope dilution methods should be quantified for use in the risk assessment using an MDL-derived RDL. Commonly used non-isotope dilution methods include SW-846 Method 8260 (volatiles), SW-846 Method 8270 (semivolatiles),
>
> 1. Require the laboratory to report the actual MDL for every non-detect compound analyzed, in addition to the commonly used reporting limit, such as an EDL, EQL, or PQL. The MDL should be derived in a manner consistent with 40 CFR Part 136 Appendix B. This would also apply for 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 for quantitative application in the risk assessment. This would be obtained by multiplying the MDL, as reported by the laboratory, times 2.623 (interim factor) (U.S. EPA 1995i).
>
> 3. Adjust the RDL, as appropriate, to account for sample-specific volumetric treatments (e.g., splits and dilutions) that differ from those utilized in the Part 136 MDL determinations.
>
> ***Isotope Dilution Methods (SW-846 Methods 8290,1624, 1625; and CARB 429, etc.):*** Non-detects for COPCs analyzed with isotope dilution methods should be quantified for use in the risk assessment using the EDL as defined by the analytical method without the use of empirical factors or other mathematical manipulations specific to the laboratory (e.g., EDL as defined in SW-846). Commonly used isotope dilution methods include SW-846 Methods 8290, 1624, and 1625.

It should be noted that the MDL definition used in 40 CFR Part 136 (see Section 2.4.1) addresses errors of the first type, false negatives. The 99 percent confidence limit stating that the MDL has only a 1 percent chance the detects will be misidentified as negative, when the compound of concern was present. Errors of the second type, false positives are not addressed. By not addressing false positives, or errors of the second type, the statistically defined default value become 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 conservative approach, and biased toward not missing any compounds of potential concern that may be present. The use of 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): (1) the uniform fill-in (UFI) method, in which each LOD value is replaced with a randomly generated data point by using a uniform distribution; (2) the log fill-in LFI method which is the same as UFI, except for using a logarithmic distribution; (3) the normal fill-in (NFI) method which is the same as UFI, except for using a log-normal distribution; and (4) the maximum likelihood estimation (MLE) techniques. If determined to be applicable by the permitting authority, a Monte Carlo simulation may also be used to determine a "statistical" value for each non-detect concentration.

2.4.4 U.S. EPA OSW-Recommendations on Quantifying Non-Detects

Use of non-detects in risk assessments is dependent on the analytical method used to produce the data. In most cases, U.S. EPA will estimate 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 the method-defined EDL for isotope dilution methods. U.S. EPA OSW believes that these methods are reasonable and conservative, and that they represent a scientifically sound approach that allows 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. It is also recognized 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 state permitting authorities have expressed the desire to obtain and use non-routine data (e.g., uncensored data) of defensible quality in the risk assessment as a way to deal with non-detect issues. While this guidance does not address what forms or how such data may be used, the decision to use non-routine data in a risk assessment is not precluded just because it is different; nor does this guidance automatically approve the use of non-routine data. A permitting authority that chooses to use non-routine data should carefully identify and evaluate the limitations associated with non-routine data and clearly document this discussion in the uncertainty section of the risk assessment report.

For collection of data to be used in a risk assessment, U.S. EPA OSW recommends comprehensive sampling using typical sampling and analytical methods for VOCs, SVOCs, metals, PCDDs, PCDFs, total organics, and other appropriate constituents as necessary based on the type of waste that will be burned by the unit. A pretrial burn risk assessment can help to ensure that the desired quantitation limit (and, therefore, DREs and COPC stack gas emission rates) will be achieved during the trial burn test.

2.4.5 Estimated Maximum Possible Concentration (EMPC)

The EMPC, as defined in SW-846 Methods 8280A and 8290, is in most cases only used with the isotope dilution methods as stated. An EMPC is calculated for dioxin isomers that are characterized by 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 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. U.S. EPA OSW recommends that EMPC values be used as detections without any further manipulation (e.g., dividing by 2). However, because EMPCs are worst case estimates of stack gas concentrations, permitting authorities and facilities should consider techniques to minimize EMPCs when reporting trial and risk burn results, especially when the EMPC values result in risk estimates above regulatory levels of concern. Some techniques that may be applied 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.

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Actual MDLs for all non-detect results
- Description of the method applied to quantify the concentration of non-detects

2.5 CONCENTRATIONS DETECTED IN BLANKS

Blank samples are intended to provide a measure of any contamination that may have been introduced into a sample either in the field while the samples were being collected, in transport to the laboratory, or in the laboratory during sample preparation or analysis. Blank samples are analyzed in the same manner as the site samples from the trail burn. In order to prevent the inclusion of non-site related compounds in the risk assessment, the concentrations of compounds detected in blanks should be compared 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 of 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 should 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 of 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., degestions 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 should be compared with the results with which the blanks are associated. However, if the association between blanks and data can not be made, blank data should be compared to the results from the entire sample data set.

U.S. EPA (1989e) makes a division in comparison 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 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 treated as a non-detect in that particular sample.

In some limited cases, it may be appropriate to consider blanks which contain compounds that are not considered by U.S. EPA to be common laboratory contaminants as identified above. In these limited cases, sample results are not 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 is treated as a non-detect in that particular sample.

Permitting authorities should carefully consider the evaluation of blank data in the overall context of the risk assessment and permitting process. U.S. EPA OSW expects that issues related to non-laboratory contaminant blanks to be minimal because data collection and analysis efforts in support of trial and risk burns are expected to be of high quality in strict conformance to QA/QC plans and SOPs. The trial and risk burn data should be carefully evaluated to ensure that the level of contamination present in the blanks does not compromise the integrity of the data for purposes of risk assessment, or result in retesting in order to properly address data quality issues.

When considering blank contamination in the COPC selection process, permitting authorities should 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 and 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 reports 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:

- U.S. EPA-Recommended Air Dispersion and Deposition Model
- Air Model Development
- Site-Specific Characteristics Required for Air Modeling
- Use of Unit Emission Rate
- Partitioning of Emissions
- Meteorological Data Required for Air Modeling
- Meteorological Preprocessors and Interface Programs
- ISCST3 Model Input Files
- ISCST3 Model Execution
- Use of Modeled Output
- Modeling Fugitive Emissions
- Modeling Acute Risk

Combustion of materials produces residual amounts of pollution that may be released to the environment. Estimation of potential human health risks associated with these releases requires knowledge of atmospheric pollutant concentrations and annual deposition rates in the areas around the combustion facility at actual and reasonable future exposure scenario locations. Air concentrations and deposition rates are usually estimated by using air dispersion models. Air dispersion models are mathematical constructs that approximate the physical processes occurring in the atmosphere that directly influence the dispersion of gaseous and particulate emissions from the stack of a combustion unit. These mathematical constructs are coded into computer programs to facilitate the computational process.

This chapter provides guidance on the development and use of the standard U.S. EPA air dispersion model that U.S. EPA expects to be used in most situations—the Industrial Source Complex Short-Term

Model (ISCST3). ISCST3 requires the use of the following information for input into the model, and consideration of output file development:

- Site-specific characteristics required for air modeling (Section 3.2)
    - Surrounding terrain (Section 3.2.1)
    - Surrounding land use (Section 3.2.2)
    - Facility building characteristics (Section 3.2.4)
- Unit emission rate (Section 3.3)
- Partitioning of emissions (Section 3.4)
- Meteorological data (Section 3.5)
- Source Characteristics (Section 3.7)

ISCST3 also requires the use of several preprocessing computer programs that prepare and organize data for use in the model. Section 3.6 describes these programs. Section 3.7 describes the structure and format of the input files. Section 3.8 describes limitations to be considered in executing ISCST3. Section 3.9 describes how to use the model output in the risk assessment computations. Section 3.10 discusses air modeling of fugitive emissions.

If applicable, readers are encouraged to consult the air dispersion modeling chapter (Chapter 3) of the 1998 U.S. EPA *Protocol for Screening Level Ecological Risk Assessment*, before beginning the air modeling process to ensure the consideration of specific issues related to ecological risk assessment. Additionally, the *Guideline on Air Quality Models* (GAQM) is a primary reference for all US EPA and state agencies on the use of air models for regulatory purposes. The GAQM is incorporated in 40 CFR Part 51 as Appendix W. 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, user's guides, and model applicability on the Support Center for Regulatory Air Models (SCRAM) web site at address "http://www.epa.gov/scram001/index.htm". This web site should also be periodically reviewed by the user to check for updates and changes to the ISCST3 model. General questions regarding air modeling or information on the web site should be addressed to "atkinson.dennis@epamail.epa.gov". Specific questions on the use of this guidance should be addressed to the appropriate permitting authority.

3.1 DEVELOPMENT OF AIR MODELS

This section (1) briefly describes the history of air model development, (2) introduces some data preprocessing programs developed to aid in preparing air model input files (these preprocessing programs are described in more detail in Sections 3.2.4 and 3.6, and (3) introduces ExInter Version 1.0, a preprocessor to ISCST3.

3.1.1 History of HHRAP Air Dispersion Models

Before 1990, several air dispersion models were used by U.S. EPA and the regulated community. These models were inappropriate for use in risk assessments because they considered only concentration, and not the deposition of contaminants to land. The original U.S. EPA guidance (1990e) on completing risk assessments 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

COMPDEP was updated to include building wake effects from a version of the ISCST model in use at the time. Subsequent U.S. EPA guidance (1993h and 1994g) recommended the use of COMPDEP for air deposition modeling. U.S. EPA (1993h) 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 the use of 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.

The use of the COMPDEP, RTDMDEP, and ISCST models is described in more detail in the following user's manuals; however, all models except the current version of ISCST3 are 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.

Users of this document are advised that a draft version of ISCST3 that includes algorithms for estimating the dry gas deposition (currently referred to as the "Draft Dry Gas Deposition Model: GDISCDFT, Version 96248") is available on the SCRAM web site. Use of this version to support site specific air modeling applications is not required, because many of the parameters needed to execute the model are not available in guidance or the technical literature. Therefore, until the draft version is reviewed and approved, and the data is provided by U.S. EPA or in the technical literature, U.S. EPA OSW recommends that the current version of ISCST3, in conjunction with the procedure presented in this guidance

(Appendix B) for estimating dry gas deposition using deposition velocity and gas concentration, should be used for risk assessments.

3.1.2 Preprocessing Programs

ISCST3 requires the use of additional computer programs, referred to as "preprocessing" programs. These programs manipulate available information regarding surrounding buildings and meteorological data into a format that can be used by ISCST3. Currently, these programs include the following:

- PCRAMMET (Personal Computer Version of the Meteorological Preprocessor for the old RAM program) prepares meteorological data for use in ISCST3. The program organizes data—such as precipitation, wind speed, and wind direction—into rows and columns of information that are read by ISCST3. The PCRAMMET User's Guide contains detailed information for preparing the required meteorological input file for the ISCST3 model (U.S. EPA 1995g).

- Building Profile Input Program (BPIP) calculates the maximum crosswind widths of buildings, which ISCST3 then uses to estimate the effects on air dispersion. This effect on dispersion by surrounding buildings is typically known as building downwash or wake effects. The BPIP User's Guide contains detailed information for preparing the required 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 use in the ISCST3 by using on-site meteorological data rather than data from government sources (National Weather Service [NWS] or the Solar And Meteorological Surface Observational Network [SAMSON]). MPRM merges on-site measurements of precipitation, wind speed, and wind direction with off-site data from government sources into rows and columns of information that are read by ISCST3. The MPRM User's Guide contains information for preparing the required meteorological input file for the ISCST3 model (U.S. EPA 1996j).

Most air dispersion modeling performed to support risk assessments will use PCRAMMET and BPIP. MPRM will generally not be used unless on-site meteorological information is available. However, only MPRM is currently scheduled to be updated to include the meteorological parameters (solar radiation and leaf area index) required to execute the dry deposition of vapor algorithms included in the new version of ISCST3. The draft version of MPRM is available for review and comment on the SCRAM web site as GDMPRDFT (dated 96248).

3.1.3 Expert Interface (ExInter Version 1.0)

ExInter is an expert interface system enhanced by U.S. EPA Region 6 for the ISCST3 model. By enhancing ExInter, the goal of U.S. EPA Region 6 was to support the in-house performance of air dispersion modeling by regional U.S. EPA and state agency personnel at hazardous waste combustion units necessary to support risk assessments conducted at these facilities. ExInter enables the user to build input files and run ISCST3 and its preprocessor programs in a Windows-based environment. Specific procedures for developing input files are stored in an available knowledge database. The underlying premise of the ExInter system is that the knowledge of an "expert" modeler is available to "nonexpert" modeling personnel at all times. However, some air modeling experience is required to use ExInter and its components as recommended in this HHRAP. The ExInter program has been written in Microsoft Visual C++ in a Microsoft Windows environment.

ExInter allows for a generic source category that comprises point, area, and volume sources. For each source type, the program queries the relevant variables for the user. In addition to asking about the inputs regarding the source types, ExInter also asks about control options, receptors, meteorology, and output formats. ExInter then creates an input file, as required by the ISCST3 dispersion model. ExInter also allows the user to run the ISCST3 model and browse the results file.

Version 1.0 of ExInter provides for input parameters to model dry gas deposition included in a draft version of ISCST3. However, the data required for dry gas deposition requires a literature search and prior regulatory approval. The procedure presented in the HHRAP for estimating dry gas deposition using deposition velocity and gas concentration is appropriate without prior approval. More detailed information on how to use ExInter can be found in the following:

- U.S. EPA. 1996l. *User's Guide for ExInter 1.0. Draft Version.* U.S. EPA Region 6 Multimedia Planning and Permitting Division. Center for Combustion Science and Engineering. Dallas, Texas. EPA/R6-096-0004. October.

ExInter is available on the SCRAM web site at "http://www.epa.gov/scram001/index.htm" under the Modeling Support section "Topics for Review". Six self-extracting compressed files contain all components for installation and use. The user's guide is accessed interactively using the help command.

Individual user's guides to ISCST3, BPIP, PCRAMMET, and MPRM also provide good references for using ExInter components. ExInter requires a minimum of 15 megabytes of free hard disk space, Windows 3.1, 8 megabytes of system memory, and a 486 processor.

3.2 SITE-SPECIFIC INFORMATION REQUIRED TO SUPPORT AIR MODELING

Site-specific information for the facility and surrounding area required to support air dispersion modeling includes (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.

Often, site-specific information required to support air dispersion modeling can be obtained from review of available maps and other graphical data on the area surrounding the facility. The first step in the air modeling process is a review of available maps and other graphical data on the surrounding area. 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, should be obtained to identify site location, nearby terrain features, waterbodies and watersheds, ecosystems, nearby residences, and land use. Aerial photographs are frequently available for supplemental 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 should be provided in Universal Transverse Mercator (UTM) grid coordinates in meters east and north in both USGS reference systems.

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). Special consideration should be given not to mix source data obtained from USGS maps based on NAD 27 with digital terrain elevation data based on NAD 83. Emission source information should be obtained in the original units from the facility data, and converted to metric units for air modeling, if necessary. Digital terrain data should be acquired from USGS or another source.

The specific information that must be collected is described in the following subsections. Entry of this information into the ISCST3 input files is described in Section 3.7.

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

3.2.1 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 from 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 the modeler includes terrain elevation for each receptor grid node and specifies 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. U.S. EPA OSW recommends that most 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 its historical effect on air modeling results has been shown to be minimal.

In addition to maps which are used to orient and facilitate air modeling decisions, the digital terrain data used to extract receptor grid node elevations should be provided in electronic form. One method of obtaining receptor grid node elevations is using digital terrain data available from the USGS on the Internet at web site "http://www.usgs.gov". 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. USGS 30-meter data is available for a fee. Either 90-meter or 30-meter data is sufficient for most risk assessments which utilize 100 meter or greater grid spacing. Digital terrain data may also be purchased from a variety of commercial vendors which may require vendor-provided programs to extract the data. The elevations may also be extracted manually at 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

3.2.2 Surrounding Land Use Information

Land use information in the risk assessment is used for purposes of air dispersion modeling and the identification or selection of exposure scenario locations (see Chapter 4) in the risk assessment. Land use analysis for purposes of selecting exposure scenario locations usually occurs out to a radius of 50 kilometers from the centroid of the stacks to ensure identification of all receptors that may be impacted. However, in most cases, air modeling performed out to a radius of 10 kilometers allows adequate characterization for the evaluation of exposure scenario locations. If a facility with multiple stacks or emission sources is being evaluated, the radius should be extended from the centroid of a polygon drawn from the various stack coordinates.

Land use information is also important to air dispersion modeling, but at a radius closer (3 kilometers) to the emission source(s). Certain land uses, as defined by air modeling guidance, effect the selection of air dispersion modeling variables. These variables are known as dispersion coefficients and surface roughness. USGS 7.5-minute topographic maps, aerial photographs, or visual surveys of the area typically are used to define the air dispersion modeling land uses.

3.2.2.1 Land Use for Dispersion Coefficients

The Auer method specified in the Guideline on Air Quality Models (40 CFR Part 51, Appendix W) is used to define land use for purposes of specifying the appropriate dispersion coefficients built into ISCST3. Land use categories of "rural" or "urban" are taken from the methods of Auer (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:

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 definition.

Step 3 Classify smaller areas within the radius as either rural or urban, based on Auer's definition. (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 total 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

3.2.2.2 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 causes particles to be "thrown" 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.

U.S. EPA (1995g) recommends that land use within 5 kilometers of the stack be used to define the average surface roughness height. For consistency with the HHRAP method for determining land use for dispersion coefficients (Section 3.2.2.1), the land use within 3 kilometers generally is acceptable for determination of surface roughness. Surface roughness height values for various land use types are as follows:

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, Wesley, and Hicks (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). A specific methodology for determining average surface roughness height has not been proposed in prior guidance documents. For facilities using National Weather Service surface meteorological data, the surface roughness height for the measurement site may be set to 0.10 meters (grassland, summer) without prior approval. If a different value is proposed for the measurement site, the value should be determined applying the following procedure to land use at the measurement site. For the application site, the following method should be used to determine surface roughness height:

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 areas within the radius according to the PCRAMMET categories (for example water, grassland, cultivated land, and forest); a site visit may be necessary to verify some classifications.

Step 3 Calculate the wind rose directions from the 5 years of meteorological data to be used for the study (see Section 3.4.1.1); a wind rose can be prepared and plotted by using the U.S. EPA WRPLOT program from the U.S. EPA's Support Center for Regulatory Air Models bulletin board system (SCRAM BBS).

Step 4 Divide the circular area into 16 sectors of 22.5 degrees, corresponding to the wind rose directions (for example, north, north-northeast, northeast, and east-northeast) to be used for the study.

Step 5 Identify a representative surface roughness height for each sector, based on an area-weighted average of the land use within the sector, by using the land use categories identified above.

Step 6 Calculate the site surface roughness height by computing an average surface roughness height weighted with the frequency of wind direction occurrence for each sector.

Alternative methods of determining surface roughness height may be proposed for agency approval prior to use in an air modeling analysis.

3.2.3 Information on Facility Building Characteristics

Building wake effects 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. Therefore, the ISCST3 model contains algorithms for evaluating this phenomenon, which is also referred to as "building downwash." The downwash analysis should 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]). Building dimensions and locations are used with stack heights and locations in BPIP to identify the potential for building downwash. BPIP and the BPIP user's guide can be downloaded from the SCRAM web site and should be referred to when addressing 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. The following procedure should be used to identify buildings for input to BPIP:

Step 1 Lay out facility plot plan, with buildings and stack locations clearly identified (building heights must be 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 required to be included in the BPIP analysis by comparing building heights to stack heights. The building height test requires 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 (0.40 multiplied by 50 feet) tall will affect air flow at stack top. Any buildings shorter than 20 feet should 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 required 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.

As an example, if a 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," U.S. EPA OSW requires that all inputs to the air modeling be prepared using UTM coordinates (meters) for consistency. UTM coordinates are rectilinear, oriented to true north, and in metric units required for ISCST3 modeling. Almost all air modeling will require the use of USGS topographic data (digital and maps) for receptor elevations, terrain grid files, location of plant property, and identification of surrounding site features. Therefore, using an absolute coordinate system will enable the modeler 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 of multiple years of meteorological data requiring many hours (up to 40 hours for one deposition run with depletion), verification of locations at each step of preparing model inputs will prevent the need to remodel.

Several precautions and guidelines should be observed in preparing input files for BPIP:

- Before BPIP is run, the correct locations should be graphically confirmed. One method is to plot the buildings and stack locations by using a graphics program. Several commercial programs incorporating BPIP provide graphic displays of BPIP inputs.

- U.S. EPA recommends, in addition to using UTM coordinates for stack locations and building corners, using meters as the units for height.

- Carefully include the stack base elevation and building base elevations by using the 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).

- The stack identifications (up to eight characters) in BPIP must be 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. The five blank spaces preceding "SO" in the BPIP output file must be deleted 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 or determine stack height increases required to avoid the 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.

3.3 USE OF UNIT EMISSION RATE

The ISCST3 model is usually run with a unit emission rate of 1.0 g/s in order to preclude having to run the model for each specific COPC. The unitized concentration and deposition output from ISCST3, using a unit emission rate, are adjusted to the COPC-specific air concentrations and deposition rates in the estimating media concentration equations (see Chapter 5) by using COPC-specific emission rates obtained during the trial burn (see Chapter 2). Concentration and deposition are directly proportional to a unit emission rate used in the ISCST3 modeling.

For facilities with multiple stacks or emission sources, each source must be modeled separately. The key to not allowing more than one stack in a single run is the inability to estimate stack-specific risks, which limits the ability of a permitting agency to evaluate which stack is responsible for the resulting risks. Such ambiguity would make it impossible for the agency to specify protective, combustion unit-specific permit limits. If a facility has two or more stacks with identical characteristics (emissions, stack parameters, and nearby locations), agency approval may be requested to represent the stacks with a single set of model runs.

3.4 PARTITIONING OF EMISSIONS

COPC emissions to the environment occur in either vapor or particle phase. In general, most metals and organic COPCs with very low volatility (refer to fraction of COPC in vapor phase [*Fv*] less than 0.05, as presented in Appendix A-3) are assumed to occur only in the particle phase. Organic COPCs occur as either only vapor phase (refer to *Fv* of 1.0, as presented in Appendix A-3) or with a portion of the vapor condensed onto the surface of particulates (e.g., particle-bound). COPCs released only as particulates are modeled with different mass fractions allocated to each particle size than the mass fractions for the organics released in both the vapor and particle-bound phases. Due to the limitations of the ISCST3 model, estimates of vapor phase COPCs, particle phase COPCs, and particle-bound COPCs cannot be provided in a single pass (run) of the model. Multiple runs are required. An example of this requirement 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 .

3.4.1 Vapor Phase Modeling

ISCST3 output for vapor phase air modeling runs are vapor phase ambient air concentration and wet vapor deposition at receptor grid nodes based on the unit emission rate. Vapor phase runs do not require a particle size distribution in the ISCST3 input file. One vapor phase run is required for each receptor grid that is modeled (see Section 3.7).

3.4.2 Particle Phase Modeling (Mass Weighting)

ISCST3 uses algorithms to compute the rate at which dry and wet removal processes deposit particulate-phase COPCs emitted from a combustion unit stack to 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, with very small particles remaining 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. An ISCST3 modeling analysis of particle phase emissions for deposition rate requires an initial estimate of the particle size distribution, distinguished on the basis of particle diameter.

The diameters of small particulates contained in stack emissions are usually measured in micrometers. The distribution of particulate by particle diameter will differ from one combustion process to another, and is greatly dependent on (1) the type of furnace, (2) the design of the combustion chamber, (3) the composition of the feed fuel, (4) the particulate removal efficiency, (5) the design of the APCS, (6) the amount of air, in excess of stoichiometric amounts, that is used to sustain combustion, and (7) the temperature of combustion. However, based on these variables, 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).

U.S. EPA OSW recommends that existing facilities perform stack tests to identify particle size distribution. These data should 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 should be prepared using stack test data in the format in Table 3-1.

U.S. EPA OSW expects that stack test data will be different from the values presented in Table 3-1 because of the use of particle "cut size" for the different cascade impactor filters (or Coulter counter-based distributions) used during actual stack sampling. The test method will drive the range of particle sizes that are presented in the results of the stack test. However, because ISCST3 requires 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, stack test data must be converted into a mean particle diameter which approximates the diameter of all the particles within a defined range. Consistent with U.S. EPA 1993h, the mean particle diameter is calculated by using the following equation:

$$D_{mean} = [0.25 \cdot (D_1^3 + D_1^2 D_2 + D_1 D_2^2 + D_2^3)]^{0.33} \qquad \text{Equation 3-1}$$

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:

$$D_{mean} = \left[0.25\ (5.0^3\ +\ (5.0)^2(6.15)\ +\ (5.0)(6.15)^2\ +\ 6.15^3)\right]^{0.33} = 5.5\ \mu m$$

TABLE 3-1

GENERALIZED PARTICLE SIZE DISTRIBUTION, AND PROPORTION OF AVAILABLE SURFACE AREA, TO BE USED AS A DEFAULT IN DEPOSITION MODELING IF SITE-SPECIFIC DATA ARE UNAVAILABLE

1 Mean Particle Diameter[a] (μm)	2 Particle Radius (μm)	3 Surface Area/ Volume (μm^{-1})	4 Fraction of Total Mass[b]	5 Proportion Available Surface Area	6 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

Notes:

a Geometric mean diameter in a distribution from U.S. EPA (1980a), as presented in U.S. EPA (1993h)
b The terms mass and weight are used interchangeably when using stack test data

---

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 and should be used in the air modeling. A minimum of three particle size categories (> 10 microns, 2-10 microns, and < 2 microns) detected during stack testing are required for air modeling. 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 microns should be used to represent all mass (e.g., particle diameter of 1.0 microns 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.

After calculating the mean particle diameter (Column 1), the fraction of total mass (Column 4) per mean particle size diameter must be computed from the stack test results. For each mean particle diameter, the stack test data provides an associated mass of particulate. The fraction of total mass for each mean particle diameter is calculated by dividing the associated mass of particulate for that diameter by the total mass of particulate in the sample. In many cases, the fractions of total mass will not sum to 1.0 due to rounding errors. In these instances, U.S. EPA OSW advocates that the remaining mass fraction be added into the largest mean particle diameter mass fraction to force the total mass to 1.0.

Direct measurements of particle-size distributions at a proposed new facility may be unavailable, so it will be necessary to provide assumed particle distributions for use in ISCST3. In such instances, a representative distribution may be used. The unit on which the representative distribution is based should be as similar as practicable to the proposed unit. For example, the default distribution provided in Table 3-1 is not appropriate for a hazardous waste burning boiler with no APCD or a wet scrubber, because it is based on data from different type of unit. However, the generalized particle size (diameter) distribution in Table 3-1 may be used as a default for some combustion facilities equipped with either ESPs or fabric filters, because the distribution is relatively typical of particle size arrays that have been measured at the outlet to advanced equipment designs (Buonicore and Davis 1992; U.S. EPA 1986a; U.S. EPA 1987a).

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

3.4.3 Particle-Bound Modeling (Surface Area Weighting)

A surface area weighting, instead of mass weighting, of the particles is used in separate particle runs of ISCST3. Surface area weighting approximates the situation where a semivolatile organic contaminant that has been volatilized in the high temperature environment of a combustion system and then condensed to the surface of particles entrained in the combustion gas after it cools in the stack. Thus, the apportionment of emissions by particle diameter becomes a function of the surface area of the particle that is available for chemical adsorption (U.S. EPA 1993h).

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/m^3), the proportion of available surface area of aerodynamic spherical particles is the ratio of surface area (*S*) to volume (*V*), as follows:

- Assume aerodynamic 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 uses the particle size distribution in Table 3-1 as an example of apportioning the emission rate of the particle-bound portion of the COPC based on surface area. This procedure can be followed 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 0.400 (S/V = 3/7.5), 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. Multiplication of Column 3 by Column 4 yields a 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. The sum of 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, and 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.

After developing the particulate size distribution based on surface area, this distribution is used in ISCST3 to apportion mass of particle-bound COPCs (most organics) based on particle size. Column 6 of Table 3-1 (as developed from actual stack test data) is used in the ISCST3 input file to perform a particulate run for the particle-bound COPCs apportioned based on 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

3.5 METEOROLOGICAL DATA

To model air concentration and deposition, the ISCST3 model requires a variety of meteorological information:

1. Air concentration
    a. Hourly values
        (1) Wind direction (degrees from true north)
        (2) Wind speed (m/s)
        (3) Dry bulb (ambient air) temperature (K)
        (4) Opaque cloud cover (tenths)
        (5) Cloud ceiling height (m)
    b. Daily values
        (1) Morning mixing height (m)
        (2) Afternoon mixing height (m)

2. Deposition

   a. Dry particle deposition—hourly values for surface pressure (millibars)

   b. Wet particle deposition—hourly values
      (1) Precipitation amount (inches)
      (2) Precipitation type (liquid or frozen)

   c. Dry vapor deposition (when available)—hourly values for solar radiation (watts/m^2)

As shown in Figure 3-1, these data are available from several different sources. For most air modeling, five years of data from a representative National Weather Service station is recommended. However, in some instances where the closest NWS data is clearly not representative of site specific meteorlogical conditions, and there is insufficient time to collect 5 years of onsite data, 1 year of onsite meteorological data (consistent with GAQM) may be used to complete the risk assessment. The permitting authority should approve the representative meteorological data prior to performing air modeling.

The following subsections describe how to select the surface and upper air data that will be used in conjunction with the ISCST3 model. Section 3.7 describes the computer programs used to process the meteorological data for input to the ISCST3 model.

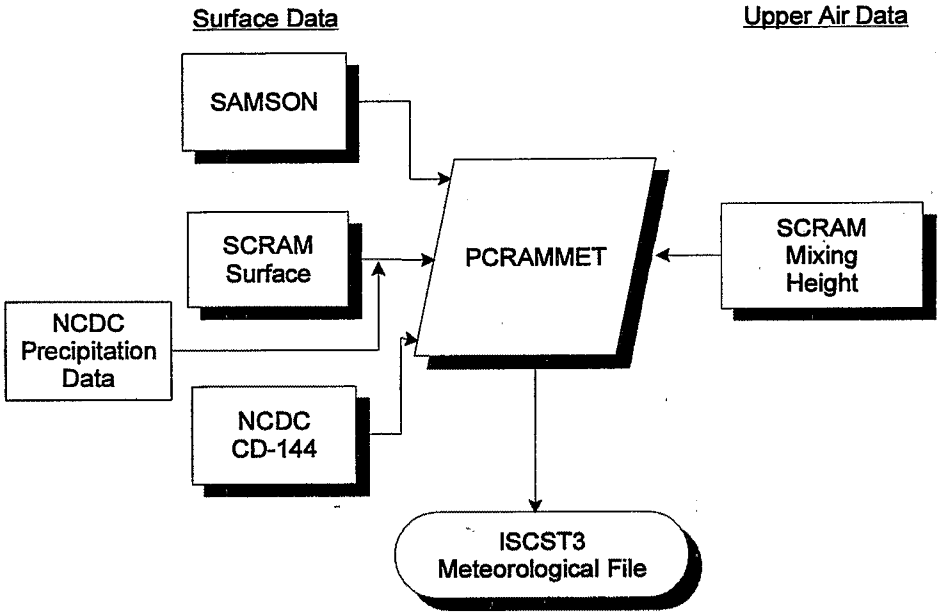
RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Identification of all sources of meteorological data

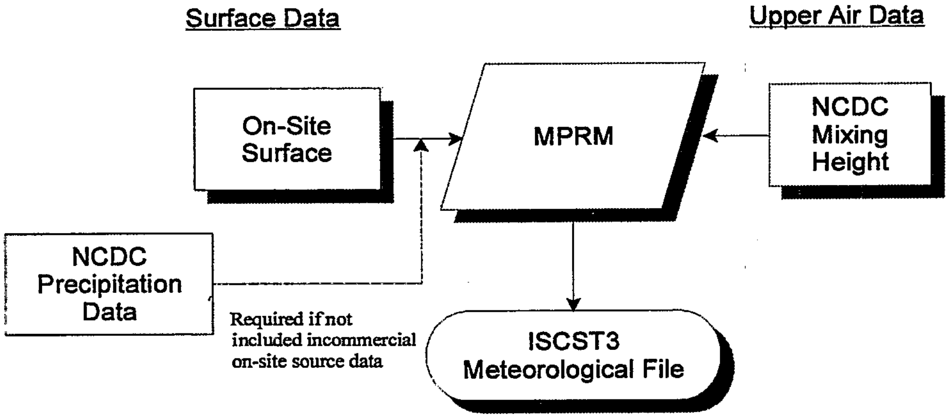
FIGURE 3-1

SOURCES OF METEOROLOGICAL DATA

Meteorological Data Processing - Government Sources



Meteorological Data Processing - Commercial Sources



Required if not included incommercial on-site source data

3.5.1 Surface Data

Surface data can be obtained from SAMSON in CD-ROM format. SAMSON data are available for 239 airports across the U.S. for the period of 1961 through 1990. The National Climate Data Center (NCDC) recently released the update to SAMSON through 1995 surface data. However, since the upper air (mixing height) data available from the U.S. EPA SCRAM web site has not been updated to cover this recent data period, it is acceptable to select the representative 5 years of meteorological data from the period up through 1990. SAMSON data contain all of the required input parameters for concentration, dry and wet particle deposition, and wet vapor deposition. SAMSON also includes the total solar radiation data required for dry vapor deposition, which may be added to ISCST3 in the future. Alternatively, some meteorological files necessary for running ISCST3 are also available on the SCRAM BBS for NWS stations located throughout the country (SCRAM BBS is part of the Office of Air Quality and Planning and Standards Technology Transfer Network [OAQPS TTN]). The meteorological data, preprocessors, and user's guides are also located on the SCRAM web site at "http://www.epa.gov/scram001/index.htm". However, these files do not contain surface pressure, types of precipitation (present weather), or precipitation amount. Although the ISCST3 model is not very sensitive to surface pressure variations, and a default value may be used, precipitation types and amounts are necessary for air modeling wet deposition. Precipitation data are available from the National Climatic Data Center (NCDC), and are processed by PCRAMMET to supplement the SCRAM BBS surface data. NCDC also has surface data in CD-144 format, which contains all of the surface data, including precipitation.

The SAMSON CD-ROM for the eastern, central, or western (Volumes I , II, and III) United States may be purchased from NCDC in Asheville, North Carolina.

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)

PCRAMMET and MPRM are the U.S. EPA meteorological preprocessor programs for preparing the surface and upper air data into a meteorlogical file of hourly parameters for input into the ISCST3 model. Most air modeling analyses will use PCRAMMET to process the National Weather Service data. However, both preprocessors require the modeler to replace any missing data. Before running PCRAMMET or MPRM, the air modeler must fill in missing data to complete 1 full year of values. A procedure recommended by U.S. EPA for filling missing surface and mixing height data is documented on the SCRAM BBS under the meteorological data section. If long periods of data are missing, and these data are not addressed by the U.S. EPA procedures on the SCRAM BBS, then a method must be developed for filling in missing data. One option is to fill the time periods with "surrogate place holder" data in the correct format with correct sequential times to complete preparation of the meteorological file. Place holder data are typically considered the last valid hourly data of record. Then, when ISCST3 is running, the `MSGPRO` keyword in the COntrol pathway can be used to specify that data are missing. Note that the `DEFAULT` keyword must not be used with `MSGPRO`. Since the missing data keyword is not approved generally for regulatory air modeling, the appropriate agency must provide approval prior to use. All processing of meteorological data should be completely documented to include sources of data, decision criteria for selection, consideration for precipitation amounts, preprocessor options selected, and filled missing data.

The most recently available 5 years of complete meteorological data contained on SAMSON, or more recent sources, should be used for the air modeling. It is desirable, but not mandatory, that the 5 years are consecutive. The use of less than five years of meteorological data should be approved by appropriate authorities. The following subsections describe important characteristics of the surface data.

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

3.5.1.1 Wind Speed and Wind Direction

Wind speed and direction are two of the most critical parameters in ISCST3. The wind direction promotes higher concentration and deposition if it persists from one direction for long periods during a year. A predominantly south wind, such as on the Gulf Coast, will contribute to high concentrations and depositions north of the facility. Wind speed is inversely proportional to concentration in the ISCST3 algorithms. The higher the wind speed, the lower will be the concentration. If wind speed doubles, the concentration and deposition will be reduced by one-half. ISCST3 needs 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 at 10 meters height at NWS stations. However, since some stations have wind speed recorded at a different height, the anemometer height must always be verified so that the correct value can be input into the PCRAMMET meteorological data preprocessing program. ISCST3 assumes that wind direction at stack height is the same as measured at the NWS station height. 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 (U.S. EPA 1995c).

3.5.1.2 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. It is measured at 2 meters above ground level. Air temperature is used in ISCST3 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, 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. The temperature is measured in K, so a stack gas temperature of 450 F is equal to 505 K. Ambient temperature of 90 F is equal to 305 K, and 32 F is 273 K. A large variation in ambient temperature will affect buoyant plume rise, but not as much as variations in stack gas temperature.

3.5.1.3 Opaque Cloud Cover

PCRAMMET uses opaque cloud cover to calculate the stability of the atmosphere. Stability determines the dispersion, or dilution, rate of the COPCs. Rapid dilution occurs in unstable air because of surface heating that overturns the air. With clear skies during the day, the sun heats the Earth's surface, thereby causing unstable air and dilution of the stack gas emission stream. Stable air results in very little mixing, or dilution, of the emitted COPCs. 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. The opaque cloud cover is observed at NWS stations each hour.

3.5.1.4 Cloud Ceiling Height

Cloud height is required in PCRAMMET to calculate stability. Specifically, the height of the cloud cover affects the heat balance at the Earth's surface. Cloud ceiling height is measured or observed at all NWS stations provided on the SAMSON CD-Roms and the U.S. EPA SCRAM web site.

3.5.1.5 Surface Pressure

Surface pressure is required by ISCST3 for calculating dry particle deposition. However, ISCST3 is not very sensitive to surface pressure. SAMSON and NCDC CD-144 data include surface pressure. SCRAM BBS surface data do not include surface pressure. U.S. EPA believes that, if SCRAM BBS surface data are used, a default value of 1,000 millibars can be assumed, with little impact on modeled results.

3.5.1.6 Precipitation Amount and Type

The importance of precipitation to ISCST3 results was discussed in the selection of the meteorological data period (see Section 3.5.1). Precipitation is measured at 3 feet (1 meter) above ground level. Precipitation amount and type are required to be processed by PCRAMMET or MPRM into the ISCST3 meteorological file to calculate wet deposition of vapor and particles. The amount of precipitation, or precipitation rate, 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 required as inputs to ISCST3 for vapors with a rate specified for liquid and frozen precipitation. The precipitation type in a weather report in SAMSON or CD-144 data file will identify to ISCST3 which event is occurring for appropriate use of the scavenging coefficients entered (see Section 3.7.2.6). SCRAM BBS surface data do not include precipitation data. Supplemental precipitation files from NCDC may be read into PCRAMMET for integration into the ISCST3 meteorological file.

3.5.1.7 Solar Radiation (Future Use for Dry Vapor Deposition)

The current version of ISCST3 does not use solar radiation. Several U.S. EPA models, including ADOM, incorporate algorithms for dry vapor deposition. At such time as U.S. EPA approves the draft version of ISCST3 which includes dry gas deposition, the hourly total solar radiation will be required. Solar radiation affects the respiratory activity of leaf surfaces, which affects the rate of vapor deposition. With a leaf area index identified in the ISCST3 input file in the future, the model will be able to calculate dry vapor deposition.

3.5.2 Upper Air Data

Upper air data, also referred to as mixing height data, are required to run the ISCST3 model. ISCST3 requires estimates of morning and afternoon (twice daily) mixing heights. PCRAMMET and MPRM use these estimates to calculate an hourly mixing height by using interpolation methods (U.S. EPA 1996j). The mixing height files are typically available for the years 1984 through 1991 on the U.S. EPA SCRAM web site. U.S. EPA OSW recommends that only years with complete mixing height data be used as input for air modeling. In some instances, data may need to be obtained from more than one station to complete five years of data. The selection of representative data should be discussed with appropriate authorities prior to performing air modeling.

Mixing height data for years prior to 1983, in addition to current mixing height data, may be purchased from NCDC as described in Section 3.5.1. The years selected for upper air data must match the years selected for surface data. If matching years of mixing height data are not available from a single upper air station, another upper air station should be used for completing the five years.

3.6 METEOROLOGICAL PREPROCESSORS AND INTERFACE PROGRAMS

After the appropriate surface and upper air data is selected following the procedures outlined in Section 3.5, additional data manipulation is necessary before the data is used with the ISCST3 model. The following subsections describe the meteorological preprocessors and interface programs used for these manipulation tasks. To eliminate any need to repeat air modeling activities, U.S. EPA OSW recommends that the selection of representative mixing height and surface data be approved by the appropriate regulatory agency before preprocessing or air modeling is conducted. Agency approval also is recommended in the selection of site-specific parameter values required as input to the meteorological data preprocessors.

3.6.1 PCRAMMET

U.S. EPA OSW recommends preparing a meteorological file for ISCST3 that can be used to calculate any concentration or deposition. By preparing a file that PCRAMMET terms a "WET DEPOSITION" file, all required parameters will 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.

PCRAMMET may use SAMSON, SCRAM web site, and NCDC CD-144 surface data files. U.S. EPA OSW recommends using the SAMSON option in PCRAMMET to process the SAMSON surface data and U.S. EPA SCRAM web site mixing height data. The PCRAMMET User's Guide in the table "Wet Deposition, SAMSON Data" (U.S. EPA 1995g) identifies the PCRAMMET input requirements for creating an ASCII meteorological file for running ISCST3 to calculate air concentration, and wet and dry deposition. The meteorological file created for ISCST3 will contain all of the parameters needed for air modeling of concentration and deposition.

PCRAMMET requires the following input parameters representative of the application site:

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

The PCRAMMET User's Guide contains detailed information for preparing the required meteorological input file for the ISCST3 model (U.S. EPA 1995g). 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. Representative values specific to the site to be modeled should be carefully selected using the tables in the PCRAMMET User's Guide or reference literature. The selected values should be approved prior to processing the meteorological data.

3.6.1.1 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 with a stable atmosphere. 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, PCRAMMET requires 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 prevent PCRAMMET from generating an error message. A value of 2.0 meter for *L* should be used when the land use surrounding the site is rural (see Section 3.2.2.1). 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:

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

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

3.6.1.2 Anemometer Height

The height of the wind speed measurements is required by ISCST3 to calculate wind speed at stack top. The wind sensor (anemometer) height is identified in the station history section of the Local Climatological

Data Summary available from NCDC for every National Weather Service station. 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 valid representative data during years of operation at more than one height. The modeler must verify the correct measurement height for each year of data prior to processing with PCRAMMET and running the ISCST3 model. ISCST3 modeled results are very sensitive to small variations in wind speed.

3.6.1.3 Surface Roughness Height at Measurement Site

Surface roughness 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 above the ground before it is "captured" for deposition on the ground. Dramatic differences in ISCST3 calculations may result from slight variations in surface roughness. For surface meteorological data from a National Weather Station, a value of 0.10 meters for the "measurement site" typically may be used without prior approval. Surface roughness is proportional, but not equal, to the physical height of the obstacles. The table in Section 3.2.2.2 lists the roughness heights that can be used as input values. These values are based on the general land use in the vicinity of the measurement site. These values should be considered in discussions with the appropriate agency modeler prior to air modeling.

3.6.1.4 Surface Roughness Height at Application Site

Determination of surface roughness height is also required at the facility (application site) for performing PCRAMMET processing to prepare an ISCST3 meteorological file. ISCST3 model results are very sensitive to the value used in PCRAMMET for this parameter. The table in Section 3.2.2.2 is applicable to the application site. A site-specific computation of a single surface roughness value representative of the site is required using the method described in Section 3.2.2.2. The computed value of surface roughness height for the application site, along with maps or photographs illustrating land use, must be approved by the appropriate agency prior to use.

3.6.1.5 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 for calculating hourly values of Monin-Obukhov length. PCRAMMET 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. Typical albedo values are presented in Table 3-2. As shown in Table 3-2, albedo values vary from 0.10 to 0.20 on water surfaces from summer to winter. The most variability is for cultivated farmland, which varies from 0.14 during spring when land is tilled to expose dark earth, to 0.60 in winter when areas are snow-covered.

Based on the information in Table 3-2, albedos are estimated to vary in rural areas from 0.14 to 0.20 for cultivated land, and from 0.18 to 0.20 for grassland. For urban areas, the variation without snow is from 0.14 to 0.18. For practical purposes, the selection of a single value for noon-time albedo to process a complete year of meteorological data is desirable. 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 representative of 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. The permitting authority should review proposed values used in the processing of the meteorological data.

TABLE 3-2

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

Notes:

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.

3.6.1.6 Bowen Ratio

The Bowen ratio is a measure of the amount of moisture at the surface. The presence of moisture affects the heat balance resulting from evaporative cooling, which, in turn, affects the hourly Monin-Obukhov length calculated by PCRAMMET. Surface moisture is highly variable. Daytime Bowen ratios are presented in Table 3-3.

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, U.S. EPA OSW recommends a single Bowen ratio value of 2.0 for urban areas; and 0.7 for rural forests, grasslands, and cultivated lands. Appropriate authorities should review proposed values used in the processing of the meteorological data.

3.6.1.7 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 3-4 presents anthropogenic heat flux (Q_f) values that have been calculated for several urban areas around the world (U.S. EPA 1995g). In rural areas, U.S. EPA OSW recommends that a value 0.0 Watts/m^2 be used 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 3-3

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
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

TABLE 3-3

DAYTIME BOWEN RATIO BY LAND USE, SEASON, AND PRECIPITATION CONDITIONS (Continued)

Land Use	Season[a]			
	Spring	Summer	Autumn	Winter
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

Note:

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.

TABLE 3-4

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)

Note:

Source—Oke (1978)

3.6.1.8 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 3-4, and recommendations presented in the PCRAMMET User's Manual based on Oke (1982), U.S. EPA OSW recommends values of 0.15 for rural areas and 0.27 for urban areas (U.S. EPA 1995g).

3.6.2 MPRM

For on-site data, a new version of MPRM is used to mesh on-site data with NWS data in the preparation of the meteorological input file. MPRM performs the same meteorological file preparation as PCRAMMET, except the source of the surface data in MPRM consists of on-site measurements (U.S. EPA 1996j). MPRM includes extensive QA/QC 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 mixing height data. NCDC data are purchased to correspond to the collection period of the on-site surface data. Mixing height data available on SCRAM's web site ends in 1991. A delay of about 3 months can occur for obtaining mixing height data from NCDC to process with recent on-site surface data.

Inputs to MPRM for preparing an ISCST3 meteorological file for concentration and deposition are the same as for PCRAMMET. Section 3.6.1 provides methods for determining values for these parameters.

Draft versions of ISCST3 and MPRM are available for review which implement dry vapor deposition. These versions are GDISCDFT (dated 96248) and GDMPRDFT (dated 96248), respectively. They may be found on the U.S. EPA SCRAM web site under "Topics for Review". These draft models are not the current regulatory versions and should not be used without approval from the appropriate permitting authority.

3.7 ISCST3 MODEL INPUT FILES

A thorough instruction of how to prepare the input files for ISCST3 is presented in the ISC3 User's Guide, Volume I (U.S. EPA 1995f), which is available for downloading from the SCRAM BBS. An example ISCST3 input file is provided in Figure 3-2. This example illustrates a single year run (1984), for particle phase COPC emissions from a single stack, to compute acute (1-hour average) and chronic (annual average) and provide single year results in one hour and annual average plot file for post-processing. The specification of a terrain grid file in the TG pathway is optional. Each air modeling analysis has unique issues and concerns that should be addressed in the risk assessment report. U.S. EPA OSW recommends that the air modeling methodology be consistent in data collection, model set-up, and model output. This will assist both the modeler and U.S. EPA in communicating and interpreting model results. The risk assessment report should document each section of the ISCST3 input file to identify consistent methods.

Three sets of ISCST3 runs are required for each COPC emission source. As discussed in Section 3.4, separate ISCST3 runs are required to model vapor phase COPCs, particle phase COPCs, and particle-bound phase COPCs for each source (stack or fugitive) of COPCs. The ISCST3 "Control Secondary Keywords" used for these three runs are:

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

ISCST3 requires site-specific inputs for source parameters, receptor locations, meteorological data, and terrain features. The model is prepared 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:

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 RISK ASSESSMENT REPORT

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

3.7.1 COntrol Pathway

Model options (MODELOPT) are specified in the COntrol pathway to direct ISCST3 in the types of computations to perform. U.S. EPA OSW recommends that air modeling specify the `DFAULT` parameter to use 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 for vapor and particles. The `DDEP` and `WDEP` parameters specify dry and wet deposition. The `DEPOS` specifies computation of total (wet and dry) deposition flux. Since ISCST3 currently does not include an algorithm for the dry deposition of vapor phase COPCs, only wet deposition is specified for vapor phase runs. Note that dry deposition of vapor phase is addressed in the pathway equations during the risk assessment using the concentration of the vapor phase and a deposition velocity. `DRYDPLT` and `WETDPLT` are used for plume depletion resulting from dry and wet removal. U.S. EPA OSW recommends the following command lines for each of the three runs (these are for rural areas; substitute `URBAN` for urban areas):

Vapor:

```
CO MODELOPT DFAULT CONC WDEP WETDPLT RURAL
```

Particle Phase:

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

Particle-Bound:

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

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
```

For each of the three runs for each emission source, 5 years of off-site (e.g., National Weather Service from SAMSON) meteorological data are completed. For sites with meteorological data collected on-site, the appropriate permitting authority should be notified for the data period required for a risk assessment. The averaging times (`AVERTIME`) should be specified as '`ANNUAL`' to compute chronic (annual average) health risk, and '1' to compute acute health risks based on the maximum 1-hour average concentrations over the 5-year period (see Section 3.11). Each phase run may be repeated 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, the modeler may 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. The modeler should 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 required 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 required to obtain 1-year or 5-year air modeling values.

In addition, ISCST3 allows the specification of COPC half-life and decay coefficients. Unless approved by the permitting authority with documentation of COPC-specific data, these keywords should not be used when conducting air modeling to support risk assessments. The `TERRHGTS` keyword with the `ELEV` parameter typically should be used 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.

U.S. EPA OSW also recommends that `SAVEFIL` be used 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. The save interval should be no longer than 5 days for large runs. If two save files are used, and a failure occurs during writing to the savefile, no more than 10 days will be lost. The `INITFILE` command should be used 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`, the ISCST3 run can be restarted 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 the `MULTYEAR` keyword is not used for computing long-term averages and 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 pahse 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
```

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

3.7.2 SOurce Pathway

As discussed in Section 3.3, ISCST3 normally uses a unit emission rate of 1.0 g/s. Additional source characteristics required by the model (typically obtained from 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

3.7.2.1 Source Location

The location keyword of the SOurce pathway `(SO LOCATION)` identifies 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 3.10. The source location must be entered into ISCST3. Locations should be entered in UTM coordinates. The easterly coordinate is entered to the nearest meter; for example, 637524 meters UTM-E (no commas are used). The northerly coordinate is entered to the nearest meter; for example, a northerly coordinate of 4,567,789 meters UTM-N is entered as 4567789. The base elevation of each stack must be entered in meters. Base elevation may be obtained from a USGS topographic map, facility plot plans or USGS digital data base.

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.
```

3.7.2.2 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. The unit emission rate is entered as 1.0 g/s. Stack height is the height above plant base elevation on the `SO LOCATION` keyword. Stack exit temperature is the most critical stack parameter for influencing concentration and deposition. High stack temperatures result in high buoyant plume rise, which, in turn, lowers concentration and deposition rates. Stack temperatures should be based on stack sampling tests for existing stacks. For new or undefined stacks, manufacturer's data for similar equipment should be used. Stack exit velocity should be calculated from actual stack gas flow rates and stack diameter. Actual stack gas flow rates should be determined for existing stacks during stack sampling. Representative values for new or undefined sources should be obtained from manufacturer's data on similar equipment. Stack diameter is the inside diameter of the stack at exit.

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
```

3.7.2.3 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. The dimensions are calculated by using the U.S. EPA program BPIP, as described in Section 3.2.4.

The BPIP output file is input 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
```

3.7.2.4 Particle Size Distribution

ISCST3 requires particle size distribution for determining deposition velocities. U.S. EPA OSW recommends site-specific stack test data for existing sources. New or undefined sources may use the particle size distribution presented in Table 3-1.

The following example is the ISCST3 input for particle phase run. From Table 3-1, the distribution for 9 mean diameter sizes includes the data required 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
```

3.7.2.5 Particle Density

Particle density is also required for modeling the air concentration and deposition rates of particles. Site-specific measured data on particle density should be determined for all existing sources when possible. For new or undefined sources requiring air modeling, a default value for particle density of 1.0 g/cm^3 may be used. 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 input in 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
```

3.7.2.6 Scavenging Coefficients

Wet deposition flux is calculated within ISCST3 by multiplying a scavenging ratio by the vertically integrated concentration. The scavenging ratio is the product of a scavenging coefficient and a precipitation rate. Studies have shown 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, vapors are assumed to be scavenged at the rate of the smallest particles with behavior in the atmosphere that is assumed to be 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).

To use the wet deposition option in ISCST3, users must input scavenging coefficients for each particle size and a file that has hourly precipitation data. For wet deposition of vapors, a scavenging coefficient for a 0.1-μm particle may be input to simulate wet scavenging of very small (molecular) particles. Alternatively, site-specific measured washout data or a calculation based on Henry's Law constant may be approved by the appropriate permitting authority prior to analysis. Wet deposition results only during precipitation. Scavenging coefficients should be determined for each particle size from the best fit curve based on the work of Jindal and Heinhold (1991) presented in the ISC3 User's Guide (U.S. EPA 1995f). The curves are limited to a maximum particle size of 10-μm, so all scavenging coefficients for particle sizes greater than or equal to 10-μm are assumed to be equal. This assumption follows research on wet scavenging of particles (Jindal and Heinhold 1991).

The ISCST3 model input also differentiates between frozen and liquid scavenging coefficients. As a conservative estimate, the frozen scavenging coefficients are assumed to be equal to the liquid scavenging coefficients (PEI and Cramer 1986). If desired, the user may input separate scavenging coefficients for frozen precipitation. Research on sulfate and nitrate data has shown that frozen precipitation scavenging coefficients are about one-third of the values of liquid precipitation (Scire, Strimaitis, and Yamartino 1990; Witby 1978).

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
```

3.7.3 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, because polar grids provide coverage over large areas with a reduced number of receptor grid nodes, thereby reducing computer run times. However, U.S. EPA Region 6 experience indicates that, although the use of polar grids may reduce computer run times, air modelers typically choose a different option, because 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).

Receptor grid node arrays may be generated by using ISCST3 grid generation. However, assigning terrain elevations for each receptor grid node in an array associated with the generated grid can result in errors. One method of obtaining a Cartesian grid with terrain elevations is to open the USGS DEM file in a graphics program (e.g., SURFER®). Selection of the grid option samples the DEM file, at the user-specified spacing, over a range of east (x) and north (y) values. The specified x and y locations extract terrain elevation (z) from the DEM file at the desired receptor grid node for air modeling with the appropriate terrain elevations at each receptor grid node. These x, y, and z values are saved as a text file with one receptor grid node per line. A text editor is used to prefix each line with "`RE DISCCART`" to specify a discrete receptor grid node in ISCST3 format. Commercial receptor grid generators are also available. One commercial program (Lakes Environmental Software Inc.) 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 and including terrain elevations (in meters):

```
RE STARTING
RE ELEVUNIT METERS
RE DISCCART 630000. 565000. 352.
RE DISCCART 630500. 565000. 365.
RE DISCCART 631000. 565000. 402.

RE DISCCART 635000. 570000. 387.
RE FINISHED
```

U.S. EPA OSW recommends 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 with the origin at the centroid of a polygon formed by the locations of the stack emission sources. This receptor grid node array should consist of a Cartesian grid with grid nodes spaced 100 meters apart extending from the centroid of the emission sources out to 3 kilometers from the centroid. 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. This same receptor grid node array is included in the REceptor pathway for all ISCST3 runs for all years of meteorological data and for all emission sources. During the risk assessment, air parameter (concentration and deposition) values for a single receptor grid node within the array may be selected for evaluation of a specific exposure scenario location, or an area average of air parameter values at multiple receptor grid nodes may be computed to represent the average concentration or deposition over a watershed or water body (see Chapter 4).

Terrain elevations should be specified 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:

Worldwide Web:	http://edcwww.cr.usgs.gov/pub/data/dem/250
FTP (two options):	ftp://edcwww.cr.usgs.gov/pub/data/dem/250 ftp://edcftp.cr.usgs.gov/pub/data/dem/250

This data has horizontal spacing between digital terrain values of approximately 90 meters which provides sufficient accuracy for air modeling.

In addition to the receptor grid node array evaluated for each facility out to 10 kilometers, other grid node arrays may be considered for evaluation of water bodies and their watersheds located beyond 10 kilometers. Grid node spacing of 500 meters between nodes is recommended for grid node arrays positioned at distances greater than 10 kilometers from the emission source. An equally spaced grid node array facilitates subsequent computation of area averages for deposition rates onto the waterbody and its associated watershed.

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.

3.7.4 MEteorological Pathway

The file containing meteorological data is specified in the MEteorological pathway. PCRAMMET creates individual files for each of 5 years, as ASCII files, to be read into ISCST3 for computing hourly concentrations and deposition rates. The modeler may specify a single year of meteorological data in each ISCST3 run, or combine the total period of meteorological data into a single meteorological file for processing by ISCST3 in a single 5-year run. When combining meteorological files, the modeler is cautioned to consider the following:

- Preprocess each year separately using PCRAMMET or MPRM into an ASCII format
- Combine the years into a single file (using a text editor or DOS COPY command)
- The first line (header) of the combined file is read by ISCST3 for comparison to the Surface and Upper Air Station ID numbers specified in the input file ME pathway
- The header for subsequent years is read by ISCST3 only if not 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. For air modeling analysis which 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), the modeler should 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., for the period 1984-1988, the anemometer was relocated from 20 feet to 10 meters on December 15, 1985), the modeler should run each year separately to specify the correct anemometer height in the ISCST3 input file ME pathway which corresponds to the correct height for that year of meteorological data.

Details of specifying the meteorological data file are in the ISC3 User's Guide (Section 3.5.1.1). Each year within the file must be complete with a full year of data (365 days, or 366 days for leap years). The anemometer height must be verified for the surface station from Local Climate Data Summary records, or other sources, such as the state climatologist office. U.S. EPA OSW recommends that the anemometer height `ANEMHGHT` for the wind speed measurements at the surface station be correctly identified before air modeling.

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
```

3.7.5 Terrain Grid (TG) Pathway

The computation of 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 the inclusion of a terrain grid file in the TG pathway significantly increases model execution time, U.S. EPA OSW recommends that a terrain grid file is not 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 described 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. U.S. EPA OSW requires that all emission sources and receptor grid nodes be specified in UTM coordinates (not 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, U.S. EPA OSW recommends that the terrain elevations in the TG file be presented 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
```

3.7.6 OUtput Pathway

ISCST3 provides numerous output file options in addition to the results in the output summary file specified in receptor tables (RECTABLE). The plot file is most useful for facilitating post-processing of the air parameter values in the model output. The plot file lists the x and y coordinates and the concentration or deposition rate values for each averaging period in a format that can be easily pulled into a post-processing program (or spreadsheet). Note that the ISCST3 generated 'plot' file is not the same format as the ISCST3 generated 'post' file. U.S. EPA OSW recommends 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 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 of the annual average results for all sources in the run for each receptor grid node.

3.8 ISCST3 MODEL EXECUTION

Model execution time should be considered 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. For example, an ISCST3 particle run of one source with 800 receptor grid nodes, on 1 year of meteorological data, with the options for air concentration, wet and dry deposition, and plume depletion required about 40 hours on a personal computer with a 486 processor running at 66 megahertz (486/66). The same run can be completed in about 10 hours on a 586/120 personal computer. Five years of meteorological data and an additional 1,600 receptor grid nodes result in total run times of 120 hours for 1 year, and 600 hours for a 5-year analysis on a 486/66 personal computer. Run time on a 586/120 personal computer is estimated at about 150 hours. A significant loss of modeling effort and analysis time can be prevented by verifying input parameters and conducting test runs prior to executing the ISCST3 runs.

Long run times result mainly from two algorithms—plume depletion and terrain grid file. ISCST3 run times are increased as much as tenfold for runs applying plume depletion. U.S. EPA OSW believes that constituent mass must be 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 conservative. However, the nominal benefits of including a terrain grid file do not justify the added run times. Therefore, plume depletion should always be included, but terrain grid files are not recommended.

3.9 USE OF MODELED OUTPUT

The ISCST3 modeled output (air concentrations and deposition rates) are provided on a unit emission rate (1.0 g/s) basis from the combustion unit or emission source, and are not COPC-specific. The estimating media equations presented in Chapter 5 and Appendix B require the model output (air parameters, see Table 3-5) directly without converting the unit based output to COPC-specific output. However, there may 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.

3.9.1 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) would also be linear if the COPC-specific emission rate was used in the air model. Section 3.3 discussed the use of the unit emission rate and advanced the theory that a unit emission rate should be used instead of the COPC-specific emission rate in order to preclude having to run the ISCST3 model separately for each individual COPC. The use of a unit emission rate in the air modeling is advocated because a common ratio relationship can be developed between the unit emission rate and the COPC-specific emission rate based on the fact that in the air model, both individual relationships are linear. This ratio relationship can be expressed by the following equation:

$$\frac{\text{COPC-Specific Air Concentration}}{\text{COPC-Specific Emission Rate}} = \frac{\text{Modeled Output Air Concentration}}{\text{Unit Emission Rate}} \qquad \text{Equation 3-2}$$

TABLE 3-5

AIR PARAMETERS FROM ISCST3 MODELED 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
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 f rom 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
Dywwv	Unitized yearly (water body or watershed) average wet 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

Use of this equation requires that three of the variables be known. The modeled output air concentration (or deposition rate) is provided by the air model, the unit emission is 1.0 g/s, and the COPC-specific emission rate; which is obtained directly from stack or source test data.

3.9.1.1 Determination of the COPC-Specific Emission Rate (*Q*)

The COPC-specific emission rate can usually be determined with information obtained directly from the trial burn report. The COPC-specific emission rate from the stack is a function of the stack gas flow rate and the stack gas concentration of each COPC; which can be calculated from the following equation:

$$Q = SGF \cdot \frac{SGC \cdot CFO_2}{1 \times 10^6} \qquad \text{Equation 3-4}$$

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 determining COPC-specific emission rates for fugitive emission sources can be found 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).

3.9.1.2 Converting Unit Output to COPC-Specific Output

Once the three of the four variables in Equation 3-1 are known, the COPC-specific air concentrations and deposition rates can be obtained directly by multiplication, as follows:

$$\begin{array}{c}\text{COPC-Specific}\\ \text{Air Concentration}\end{array} = \frac{\text{Modeled Output Air Concentration} \cdot \text{COPC-Specific Emission Rate}}{\text{Unit Emission Rate}} \qquad \text{Equation 3-3}$$

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). Deposition is calculated similarly, proportional to the emission rate of each COPC. Readers are reminded once again that this 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.

3.9.2 Output from the ISCST3 Model

The ISCST3 output is structured and the risk assessor must understand how to read the output 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 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 execution of the ISCST3 model
`inputfile.INP:`	is the input file name selected by the modeler
`outputfile.OUT:`	is the output file name selected by the modeler, typically the same as the input file name

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. The output file (PART84R.OUT) from the run will automatically be written by ISCST3 during model execution.

```
ISCST3 PART84R.INP PART84R.OUT
```

The output 'plot file' is specified by the modeler in the ISCST3 input file OUtput pathway and created by ISCST3 during the run (see Section 3.7.6). Figure 3-3 is an example of the first few lines in the particle phase plot file with single-year annual average concentration, total deposition, dry deposition and wet deposition values for each receptor grid node. The total deposition is the sum of the dry and wet components of deposition. The single-year values at each receptor grid node being evaluated must be averaged to a 5-year value. The 5-year averaged values at the receptor grid nodes selected for evaluation in the risk assessment (see Section 3.9.3), are used in the estimating media concentration equations. This file is usually imported into a post-processing program (or spreadsheet) before entry into the risk assessment computations.

Similar plot files are produced for the particle-bound and vapor phase runs. The output for the vapor phase runs will be average concentration and wet deposition. The output for the particle and particle-bound phase runs will be average concentration, dry deposition, wet deposition and total deposition. Again, the 1-year values at each receptor grid node must be averaged to a 5-year value at each node unless a single five-year ISCST3 run using a combined meteorological file is used. If the 5-year combined file is used, the results from the ISCST3 plot file may be used directly in the risk assessment without averaging over the five years.

All values are defined as used in the estimating media concentration equations (see Chapter 5).

3.9.3 Use of Model Output in Estimating Media Equations

Section 3.4 discussed how consideration of partitioning of the COPCs effects the development of ISCST3 modeling runs. The selection of which air modeled air parameter values (air concentrations and deposition rates) to use in the estimating media concentration equations is based on this same partitioning theory.

3.9.3.1 Vapor Phase COPCs

ISCST3 output generated from vapor phase air modeling runs are vapor phase air concentrations (unitized *Cyv* and unitized *Cywv*) 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 dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene, which have vapor phase fractions, Fv, less than five percent. The air concentration (unitized *Cyv*) and wet vapor deposition (unitized *Dywv*) from the vapor phase run is also used in the estimating media concentration equations for mercury. Values for these

COPCs are selected from the vapor phase run because the mass of the COPC emitted by the combustion unit is assumed to have either all or a portion of its mass in the vapor phase (see Appendix A-3).

3.9.3.2 Particle Phase COPCs

ISCST3 output generated from particle phase air modeling runs are 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). Values for inorganic and relatively non-volative COPCs are selected from the particle phase run because the mass of the COPC emitted by the combustion unit is assumed to have all of its mass in the particulate phase (see Appendix A-3), apportioned across the particle size distribution based on mass weighting.

3.9.3.3 Particle-Bound COPCs

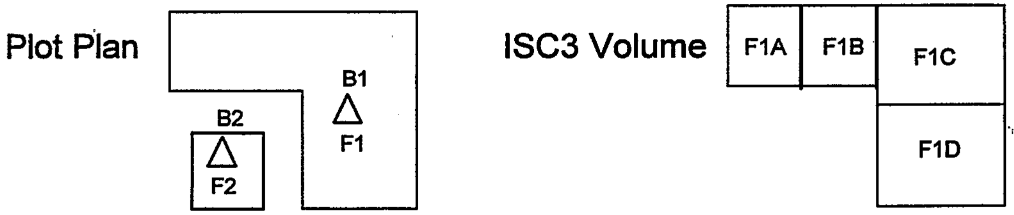
ISCST3 output generated from particle-bound air modeling runs are air concentration (unitized *Cyp*), dry deposition (unitized *Dydp*), wet deposition (unitized *Dywp*), and combined deposition (unitized *Dytwp*) for organic COPCs and mercury (see Chapter 2 and Appendix A-3) at receptor grid nodes based on the unit emission rate. These values are used in the estimating media concentration equations for all COPC organics and mercury to account for a portion of the vapor condensed onto the surface of particulates. Values for these COPCs are selected from the particle-bound run because the mass of the COPC emitted by the combustion unit is assumed to have a portion of its mass condensed on particulates (see Appendix A-3), apportioned across the particle size distribution based on surface area weighting.

3.10 MODELING OF FUGITIVE EMISSIONS

Fugitive source emissions, as defined in Chapter 2, should be modeled using the procedures presented throughout this chapter for stack source emissions. However, the fugitive emissions should be represented in the ISCST3 input file SOurce pathway as either "area" or "volume" source types. Fugitive emissions of volatile organics are modeled only in the vapor phase. Fugitive emissions of ash are modeled 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 combustion units that include storage vessels, pipes, valves, seals and flanges. The horizontal area of the fugitive source (which can be obtained from the facility plot plan) is entered into the ISCST3 input file following the instructions presented in the ISC3 User's Guide, Volume I (U.S. EPA 1995f). The height of the fugitive source is defined as the top of the vertical extent of the equipment. If the vertical extent of the fugitive source is not known, a default height of ground level (release height of zero) may be input, providing a conservative estimate of potential impacts. The ISCST3 model run time is faster for volume source types than for area source types, and should be considered for most applications. The methods in the ISCST3 User's Guide should be followed in defining the input parameters to represent the fugitive source.

The following example is for organic fugitive emissions modeled as a volume source type. For a facility which may have 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 required.



One run is required for each of the two stacks as point sources. One run is required for each of the two fugitive areas as volume sources (Note: modeler may alternatively model as an area source). Since the emissions are fugitive volatile organics, only the vapor phase is modeled. 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, the release height is specified as 2.5 meters (½ of vertical extent of fugitive emissions). The initial vertical dimension is specified as 1.16 meters (vertical extent of 5 meters divided by 4.3 as described in the ISC3 User's Guide).

FIGURE 3-3

EXAMPLE PLOT FILE

```
*ISCST3 (96113):  Example Particle Phase Run, Single Year 1990
*MODELING OPTIONS USED:
* CONC    DEPOS  DDEP   WDEP   RURAL  ELEV          DFAULT                                            DRYDPL WETDPL

*        PLOT FILE OF ANNUAL VALUES FOR SOURCE GROUP: ALL
*        FOR A TOTAL OF    21 RECEPTORS.
*        FORMAT: (6(1X,F13.5),1X,F8.2,2X,A6,2X,A8,2X,I8,2X,A8)
*     X            Y         AVERAGE CONC  TOTAL DEPO    DRY DEPO      WET DEPO   ZELEV      AVE    GRP   NUM HRS    NET
ID
*____________ _____________  ___________  _________   ___________   _________  ______   ______  _____  _______
_________
691600.00000 3342050.00000     0.29900     0.28658      0.20024      0.08634    4.00   ANNUAL  ALL     8760      NA
691700.00000 3342050.00000     0.30203     0.35416      0.23884      0.11532    5.00   ANNUAL  ALL     8760      NA

691800.00000 3342050.00000     0.25174     0.42461      0.25976      0.16485    5.00   ANNUAL  ALL     8760      NA

691900.00000 3342050.00000     0.13256     0.50524      0.23852      0.26672    5.00   ANNUAL  ALL     8760      NA

692000.00000 3342050.00000     0.00322     0.61790      0.05850      0.55940    5.00   ANNUAL  ALL     8760      NA

692100.00000 3342050.00000     0.00000     6.32022      0.00000      6.32022    6.00   ANNUAL  ALL     8760      NA

692200.00000 3342050.00000     0.00319     0.32218      0.06577      0.25641    6.00   ANNUAL  ALL     8760      NA

692300.00000 3342050.00000     0.13768     0.39938      0.21734      0.18204    6.00   ANNUAL  ALL     8760      NA

692400.00000 3342050.00000     0.23546     0.33855      0.20975      0.12880    6.00   ANNUAL  ALL     8760      NA

692500.00000 3342050.00000     0.25673     0.27475      0.17903      0.09572    6.00   ANNUAL  ALL     8760      NA

692600.00000 3342050.00000     0.24706     0.22195      0.14812      0.07384    6.00   ANNUAL  ALL     8760      NA

691600.00000 3342150.00000     0.37348     0.40644      0.25958      0.14685    5.00   ANNUAL  ALL     8760      NA

691700.00000 3342150.00000     0.37166     0.51388      0.31119      0.20269    5.00   ANNUAL  ALL     8760      NA

691800.00000 3342150.00000     0.34332     0.68794      0.39582      0.29212    5.00   ANNUAL  ALL     8760      NA
```

```
691900.00000 3342150.00000    0.22930    0.98039     0.54883    0.43156    5.00   ANNUAL  ALL   8760    NA
692000.00000 3342150.00000    0.03473    0.90823     0.37421    0.53402    6.00   ANNUAL  ALL   8760    NA
692100.00000 3342150.00000    0.00098    0.62882     0.15736    0.47146    6.00   ANNUAL  ALL   8760    NA
692200.00000 3342150.00000    0.02605    0.48160     0.15582    0.32578    7.00   ANNUAL  ALL   8760    NA
692300.00000 3342150.00000    0.17300    0.49313     0.22998    0.26315    7.00   ANNUAL  ALL   8760    NA
692400.00000 3342150.00000    0.24520    0.29443     0.19715    0.09729    7.00   ANNUAL  ALL   8760    NA
692500.00000 3342150.00000    0.25561    0.23482     0.16744    0.06738    7.00   ANNUAL  ALL   8760    NA
```

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, the area on the plot plan must be subdivided (ISC3 Volume) to create square areas for input to ISCST3. The four areas depicted represent subdivision into square areas. The resulting four square areas are input into a single ISCST3 run for Fugitive source F1 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 proportionately the unit emission rate (1.0 gram per second) 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). By specifying all sources to be included in the model results from ISCST3 (SO SRCGROUP ALL), the ISCST3 model will appropriately combine all four volume source subdivisions of fugitive source F1 into combined impact results for fugitive source F1. The resulting air parameter values in the plot files may be used directly in the risk assessment equations, the same as if a stack emission 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 should 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. The horizontal area of the storage pile is entered into the ISCST3 input file following the ISCST3 User's Guide, Volume I (U.S. EPA 1995f). The height of emissions is input as the top of the pile. If the vertical extent is not known, the height may be input as ground level (or zero height). Fugitive ash will typically be modeled as area source type. However, volume source type may be considered by the appropriate regulatory agency prior to air modeling. The methods in the ISCST3 User's Guide should be followed in defining the input parameters to represent the ash release as an area source.

The COntrol parameters should follow the recommendations for setting up a particulate phase computation.

```
CO CONC DDEP WDEP DEPOS
```

The emissions characterization and source type must be documented.

3.11 MODELING ACUTE RISK

The acute, or short-term, effects due to the direct inhalation of vapor phase, particle phase and particle-bound phase COPCs is generally considered in a risk assessment. Since only the ambient air concentrations are included in the direct inhalation pathway, the air parameters specific for the acute assessment may be computed in the same ISCST3 runs that compute the air parameters for the long-term chronic effects. More complete discussions of the acute risk assessment are found in Sections 4.2, 4.3, and 7.5.

From the air modeling perspective, the goal is to compute the highest 1-hour average air concentration for each phase 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, two specifications must be made in the ISCST3 input files.

First, the COntrol pathway must specify the 1-hour average as one of the averaging times. The example of this specification is included in Section 3.7.1. The ISCST3 input file should include:

```
CO AVERTIME 1 ANNUAL
```

where the '1' specifies the 1-hour averaging time to be computed for each hour of meteorological data. Recall that the 'ANNUAL' is specified for the chronic effects.

Second, the OUtput pathway should 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 will be 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 appropriate value needed for the acute risk assessment is the highest 1-hour concentration for the 5-year period. Each of the five single year values must be reviewed to identify the highest for all five years at each receptor grid node. However, if a combined 5-year meteorological file is run, the plot file will already identify the highest value for the 5-year period at each grid node with no additional processing required. The plot file for the one-hour average concentrations is specified in the OUtput pathway as:

```
OU PLOTFILE 1 ALL FIRST BTR841.PLT
```

where '1' specifies the 1-hour averaging period, 'ALL' specifies including all sources in the ISCST3 run, 'FIRST' specifies including only the first highest value at each receptor grid node, and 'BTR841.PLT' is the name for the plot file, which 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:

- Exposure Setting Characterization
- Recommended Exposure Scenarios
    - Subsistence Farmer
    - Subsistence Farmer Child
    - Adult Resident
    - Child Resident
    - Subsistence Fisher
    - Subsistence Fisher Child
    - Acute Risk
- Exposure Scenario Locations

The purpose of this chapter is to provide guidance in the identification of "exposure scenarios" that should be evaluated in the risk assessment to estimate the type and magnitude of human exposure to COPC emissions from hazardous waste combustion units (including fugitive emissions). Identification of the exposure scenarios to be evaluated includes characterization of exposure setting, identification of recommended exposure scenarios, and selection of exposure scenario locations.

An exposure scenario is a combination of "exposure pathways" to which a single "receptor" may be subjected. Human receptors may come into contact with COPCs emitted to the atmosphere from hazardous waste combustion units via two primary exposure "routes," either directly—via inhalation; or indirectly—via subsequent ingestion of water, soil, vegetation, and animals that become contaminated by COPCs through the food chain.

Exposure to COPCs may occur via numerous exposure pathways, which represent combinations of receptors and exposure routes. Each exposure pathway consists of four fundamental components: (1) an exposure route; (2) a source and mechanism of COPC release (see Chapter 2); (3) 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); and (4) a point of potential human contact with the contaminated medium, which is referred to as the exposure point and consists of a specific receptor exposed at a specific point. Humans, plants, and animals in the assessment area may take up COPCs directly from the air or indirectly via the media receiving deposition (e.g., soil, vegetation, or water).

The exposure scenarios recommended for evaluation in this guidance are generally conservative in nature and not intended to be entirely representative of actual scenarios at all sites. Rather, they are intended to allow standardized and reproducible evaluation of risks across most sites and land use areas, with conservatism incorporated to ensure protectiveness of potential receptors not directly evaluated, such as special subpopulations and regionally specific land uses. U.S. EPA OSW believes that the recommended exposure scenarios and associated assumptions presented in this chapter are reasonable and conservative, and that they represent a scientifically sound approach that allows protection of human health and the environment while recognizing the uncertainty associated with evaluating real world exposure. Unless site-specific conditions warrant exception, as approved by the permitting authority, U.S. EPA OSW recommends that these scenarios be used, at a minimum, as an initial evaluation to indicate primary risk concerns. Any exceptions, such as a deletion or modification of a recommended exposure scenario, scenario location (see Section 4.3), or both, should be well-documented and approved by the permitting authority.

The following sections describe how to (1) characterize the exposure setting, (2) identify the U.S. EPA OSW-recommended exposure scenarios, and (3) select the exposure scenario locations to be evaluated in the risk assessment.

4.1 EXPOSURE SETTING CHARACTERIZATION

The purpose of characterizing the exposure setting is to identify the non-worker related human activities and receptors in the assessment area—both inside and outside of the facility property boundary—that may be impacted as a result of exposure to emissions from one or more of a facility's emission sources. Exposure setting characterization is generally focused on identifying current and reasonable potential human activities or land uses that provides the basis for evaluation of recommended exposure scenarios (see Section 4.2) that ensure protection of the general public, versus direct evaluation of worker related exposures. This is because there are other guidance and regulations for occupational exposures to hazardous waste and hazardous waste combustion emissions within the facility boundary, such as U.S. Occupational Safety and Health Administration (OSHA), which promulgates health standards based on exposures to workers for a 40-hour work week. However, there may be some instances (e.g., acute risk) where worker exposure at nearby facilities or commercial areas within the assessment area are considered within the risk assessment.

Exposure setting characterization is generally limited to the assessment area that is defined by a 50-km radius, taken from the centroid of a polygon (also used as the origin of ISCST3 receptor grid node array, see Chapter 3) identified by the UTM coordinates of the facility stacks. A 50-km radius is generally recognized limit of the ISCST3 air dispersion model and predecessors (U.S. EPA 1990e; 1994c). However, resources for characterizing the exposure setting should initially be focused on the areas surrounding the emission sources and extending out to about 3-km; where the most significant deposition has been observed in most cases. The assessment area should include facility and non-facility property since experience has shown that some facilities located on substantial property may rent portions of the property to the public for farming, ranching, or recreational purposes (e.g., fishing). Therefore, land use and water bodies—both inside and outside the facility property boundary—should be considered for evaluation.

The purpose of characterizing the exposure setting is to identify current and reasonable potential human activities or land uses that provides the basis for evaluation of recommended exposure scenarios (see Section 4.2), and that may be impacted as a result of exposure to emissions from one or more of a facility's emission sources. The following subsections provide information on (1) current and reasonable potential future land use, (2) waterbodies and their associated watersheds, and (3) special subpopulations. Characterization of the exposure setting specific to each site's land use and each facility's emissions is critical to ensuring that relevant and accurate estimates of exposure are considered in the risk assessment.

4.1.1 Current and Reasonable Potential Future Land Use

Current and reasonable potential future land use are important factors to consider in characterizing the exposure setting; and when overlayed with the air dispersion modeling results, will define which recommended exposure scenarios and their locations should be evaluated in the risk assessment. In addition to current land use, reasonable potential future land use is also important because risk assessments evaluate the potential risks from facilities over long periods of time (greater than 30 years). Therefore, it is important to identify exposure scenario locations that are not only based on the current use of land, but also exposure scenario locations that consider reasonable potential future uses.

Current land use, and indications of future land use, can typically be identified by reviewing hard copy and/or electronic versions of land use land classification (LULC) maps, topographic maps, and aerial photographs. Sources and general information associated with each of these data types or maps are presented below. Also, as noted in Chapter 3, the UTM coordinate system format (NAD27 or NAD83) for all mapping information should be verified to ensure consistency and prevent erroneous georeferencing of locations and areas.

> ***Land Use Land Cover (LULC) Maps*** - LULC maps can be downloaded directly from the USGS web site (http://mapping.usgs.gov/index.html), at a scale of 1:250,000, in a file type GIRAS format. LULC maps can also be downloaded from the EPA web site (ftp://ftp.epa.gov/pub), at a scale of 1:250,000, in an Arc/Info export format. Exact boundaries of polygon land use area coverages, in areas being considered for evaluation, should be verified using available topographic maps and aerial photographic coverages.
>
> ***Topographic Maps*** - Topographic maps are readily available in both hard copy and electronic format directly from USGS or numerous other vendors. These maps are commonly at a scale of 1:24,000, and in a file type TIFF format with TIFF World File included for georeferencing.
>
> ***Aerial Photographs*** - Hard copy aerial photographs can be purchased directly from USGS in a variety of scales and coverages. Electronic format aerial photographs or Digital Ortho Quarter Quads (DOQQs) can also be purchased directly from USGS, or from an increasing number of commercial sources. Properly georeferenced DOQQs covering a 3-km or more radius of the assessment area, 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 selection of exposure scenario locations.

While obviously these data types or maps do not represent the universe of information available on human activities or land use, they are readily available from a number of government sources (typically accessible via the Internet), usually can be obtained for free or at low cost, and when used together provide sufficient information to reliably identify and define, in a defensible manner, land use areas to be considered for evaluation in the risk assessment. However, while the use of these or other data can be very accurate, verifying identified land use areas by conducting a site visit is recommended, if feasible. Also, discussions with representatives of private and government organizations which routinely collect and evaluate land use data (agricultural extension agencies, U.S. Department of Agriculture, natural resource and park agencies, and local governments) can be helpful in updating current land use information or providing information regarding future land use. Information on reasonable potential future land use can also be obtained from local planning and zoning authorities, which may help determine what level of development is now allowed under current regulations and what development is expected in the future.

Any known or reasonable potential future use of the land should be defined. For instance, a reasonable potential land use for a rural area that is currently characterized by open fields and intermittent housing, could reasonably be a residential subdivision that is developed in the future. Conversely, areas characterized as a tidal swamp could reasonably indicate that these areas will not become farms.

Areas with differing current and reasonable potential future land use characteristics should be defined for consideration of ISCST3 modeled receptor grid nodes within the defined land use area as possible exposure scenario locations (see Section 4.3). Land use characterization should identify population centers in the area (e.g., communities, residential developments, and rural residences), farms and ranches, and other land use type that may support recommended exposure scenarios as discussed in Section 4.2. For example, if an assessment area includes a farm and a small residential community, both of these areas should be identified so that receptor grid nodes within these areas can be further considered as possible exposure scenario locations (see Sections 4.2 and 4.3). The risk assessor should focus on land use areas potentially impacted by COPC emissions from facility emission sources being evaluated in the risk assessment.

Information on site-specific physiographic features may also be considered to provide a frame of reference for comparing default variables and associated assumptions (e.g., plant types, soil characteristics, land use, etc.) applied in the fate and transport models. For the purposes of the risk assessment, the presence, type, and extent of physiographic features can readily be determined by using the following sources: (1) USGS topographic maps, (2) Soil Conservation Service reports, (3) county and local land use maps, and (4) information from state departments of natural resources or similar agencies.

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. Risk assessors should 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. Risk assessors should focus initially on those land use areas impacted by emissions of COPCs.

4.1.2 Water Bodies and Their Associated Watersheds

Water bodies and their associated watersheds are important factors in evaluating some of the recommended exposure scenarios discussed in Section 4.2. For example, the identification of surface water bodies at locations in the assessment area receiving deposition from emission sources indicates the potential for COPC exposures from ingestion of fish, and possibly drinking water (drinking water is evaluated only if the local population obtains drinking water from surface water sources). In addition to identifying human uses associated with water bodies potentially impacted by COPC emissions, the surface area and exact location with respect to evaluating receptor grid nodes positioned within the water body should be defined. Likewise, the area and exact location with respect to evaluating receptor grid nodes positioned within the watershed should also be defined. Discussion on selection of exposure scenario locations associated with water bodies, and evaluating the ISCST3 air parameter concentrations at receptor grid nodes within the water body and associated watershed can be found in Section 4.3.

Use, area, and location of water bodies and their associated watersheds can typically be identified by reviewing the same hard copy and /or electronic versions of land use land classification (LULC) maps, topographic maps, and aerial photographs used in identification of land use. Sources and general information associated with each of these data types or maps are presented in Section 4.1.1.

Additional information on water body use can also be obtained through discussions with local authorities (e.g., state environmental agencies, fish and wildlife agencies, or local water control districts) about viability to support fish populations and drinking water sources, or current postings of fish advisories. However, risks will generally be estimated for a water body even if a fish advisory is posted. Surface water bodies that are used for drinking water sources in the assessment area should generally be evaluated in the risk assessment. While water bodies closest to the facility will generally have higher deposition rates, estimated risk is also determined by other physical parameters, including the area extent or size of the water body and the associated watershed, and by the properties of the COPCs being emitted.

For water bodies identified as potentially impacted from emission sources and selected for evaluation, the area extent of the associated watershed that contributes water to the water body should also be identified and defined by UTM coordinates. The area extent of a watershed is generally defined by topographic highs that result in downslope drainage into the water body. The watershed can be important to determining the overall water body COPC loading, because pervious and impervious areas of the watershed, as well as the soil concentration of COPCs resulting from emissions from facility sources, are also used in the media concentration equations to calculate the water body COPC concentrations resulting from watershed runoff (see Chapter 5 and Appendix B). The total watershed area that contributes water to the water body can be very extensive relative to the area that is impacted from facility emissions. Therefore, it is important that the area extent of all watersheds to be evaluated should be approved by the permitting authority, to ensure that the watershed and it's contribution to the water body is defined appropriately in consideration of the exposure scenario location (e.g., location on the water body of the drinking water intake, fishing pier, etc.) for the water body being evaluated and subsequent estimated risk.

For example, if facility emissions impact principally a land area that feeds a specific tributary that drains to a large river system and immediately upstream of a drinking water intake point, the risk assessor should 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.

As previously discussed, additional water body and watershed parameters (on an average annual basis) to be determined include the following:

- Water body surface area
- Watershed surface area
- Impervious watershed area
- Average surface water volumetric flow rate
- Current velocity of surface water body
- Depth of surface water body column
- 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. Water body current velocities and volumetric flow rates should generally be average values on an annual basis. State or local geologic surveys may keep records on water bodies. Volumetric flow rates for smaller streams or lakes can be calculated as the product of the watershed area and one-half of the local average annual surface runoff. Current velocities can be calculated as the volumetric flow rate divided by the cross-sectional area (current velocities are not used in the equations for lakes). Depths of water bodies can sometimes be obtained from state or local sources. Discussions on determining the USLE rainfall/erosivity factor are included in Chapter 5 and Appendix B.

RECOMMENDED INFORMATION FOR 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
- Rational for selection or exclusion from evaluation, water bodies within the assessment area
- Documentation of water body area, watershed area, impervious area, volumetric flow rate, current velocity, depth of water column, 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
- Information on water body use that may justify inclusion or exclusion of the water body from evaluation in the risk assessment

4.1.3 Special Subpopulation Characteristics

Special subpopulations are defined as human receptors or segments in the population that may be potentially at higher risk due to receptor sensitivity to COPCs (e.g., elderly, infants and children, fetus of pregnant women). The assumptions specified in this guidance to complete the risk assessment (such as the conservative 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) have been developed to protect human health—including special subpopulations. However, in addition to evaluation of the recommended exposure scenarios (see Section 4.2), the risk assessment may need to directly address special subpopulations in impacted areas because of characteristics of the exposure setting or to address specific community concerns; including new U.S. EPA policy focused on consistently and explicitly evaluating environmental health risks to infants and children in all risk assessments (U.S. EPA 1995j). For example, a day care center or hospital that is located in an area potentially impacted by facility emissions. Based on site-specific exposure characteristics, exposures to children at the day care center or to the sick in the hospital may need to be addressed because these receptors may be especially sensitive to the adverse effects of the COPCs emitted, and because the specific exposure setting is particularly conducive to exposure. Therefore, special subpopulations in such areas should be identified. Section 4.2 provides additional discussion on how potential exposure of special subpopulations can be evaluated consistent with evaluation of recommended exposure scenarios.

Because concerns about special subpopulations can arise at any time in the permitting process, the U.S. EPA OSW recommends that special subpopulations potentially at higher risk be identified in the exposure setting characterization for the risk assessment. Characterization should identify special subpopulations 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 may represent a special subpopulation (see Section 6.2.3.1).

RECOMMENDED INFORMATION FOR RISK ASSESSMENT REPORT

- Identification and/or mapping of the locations of special subpopulations 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

U.S. EPA OSW recommends the following exposure scenarios (also see Table 4-1):

- Subsistence Farmer
- Subsistence Farmer Child
- Adult Resident
- Child Resident
- Subsistence Fisher
- Subsistence Fisher Child

- Acute Risk

These are the same exposure scenarios recommended by U.S. EPA (1994g) with the exception of the child farmer, child fisher, and acute risk. Evaluation of the subsistence 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). The subsistence fisher child and acute risk scenarios advocated by this HHRAP are included for two primary reasons: (1) to be consistent with the adult/child pairings recommended for the resident and subsistence farmer scenarios, and (2) to ensure that the risk assessment evaluates all receptors that may be significantly exposed to emissions from facility sources.

In addition to the recommended exposure scenarios presented above, U.S. EPA OSW recommends evaluation of special subpopulations (as defined in Section 4.1.3) and communities of concern 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. Evaluation of special subpopulations or community concerns should be initially conducted by applying the recommended exposure scenario(s) (e.g., adult resident, child resident, acute risk) that are most representative of the exposure setting for the subpopulation to be evaluated; utilizing the maximum modeled air parameter values specific to the location (see Section 4.3). If initial evaluation, using the appropriate conservative recommended exposure scenarios, indicates potential risks at regulatory levels of concern, or if the subpopulation is not adequately represented by some of the exposure pathways in the initial evaluation, a refined evaluation more representative of the site-specific exposure setting characterization may be required by evaluating the specific exposure pathways applicable to the exposure occurring at the location.

For example, for a children's school or day care center located in an area impacted by facility sources, potential exposure to children at this location can be evaluated by completing the child resident scenario at the location of the school or day care. In most cases, evaluation of the child resident scenario at the school will be overly conservative because the ingestion of homegrown produce exposure pathway is most likely not occurring at that location. If necessary, a more refined evaluation that does not include ingestion of homegrown produce (only if supported by site-specific exposure setting characterization) can be conducted to provide a more accurate quantitative estimate of potential risk.

Although some of the recommended exposure scenarios are referred to as "subsistence", the actual mass per day amounts of food items (see homegrown ingestion rates, Appendix C) evaluated in the recommended exposure pathways (see Table 4-1) are more comparable to reasonable versus subsistence amounts; and therefore, may not preclude ingestion of significant amounts of food items not represented in the exposure pathways of the exposure scenario subject to evaluation. As indicated in Table 4-1, specific regional exposure setting characteristics may warrant that the permitting authority consider inclusion of additional recommended exposure pathways when evaluating an exposure scenario for a specific regional exposure setting. For example, the recommended subsistence farmer exposure scenario does not automatically include the fish ingestion exposure pathway. However, in some areas of the country, it is common for farms to also have stock ponds which are fished on a regular basis for the farmer's consumption. Since the recommended homegrown ingestion rates for produce and animal products (already considered in the evaluation) are not significant enough to reasonably prevent the farmer from also ingesting the fish caught from the local pond, the permitting authority may consider inclusion of the fish ingestion exposure pathway when evaluating a subsistence farmer exposure scenario at such locations that would reasonably indicate such an exposure setting (e.g., farms with stock ponds or near productive water bodies). This same type of example could also be considered 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 supporting fishing or wetlands supporting crawfish harvest.

U.S. EPA OSW also recommends that infant exposure to PCDDs and PCDFs via the ingestion of their mother's breast milk be evaluated as an additional exposure pathway at all recommended exposure scenario locations. Chapter 2 and Appendix C also further describe the ingestion of breast milk exposure pathway.

Also, 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 child resident (nonfarmer), U.S. EPA OSW recommends that the direct inhalation exposure pathways be evaluated for all receptors.

U.S. EPA OSW does not typically recommend that the following exposure pathways be evaluated as part of any exposure scenario:

> ***Ingestion of Ground Water*** - U.S. EPA (1990e) 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. Evaluation of potential exposure to COPCs through ingestion of drinking water from surface water bodies is anticipated 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 inhalation of resuspended dust was insignificant. Evaluation of exposure through direct inhalation of vapor and particle phase COPCs and incidental ingestion of soil are anticipated 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 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 1996g; 1995h). In addition, the estimation of potential COPC exposure via the dermal exposure pathway is associated with significant uncertainties. 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 the estimation of compound-specific absorption fractions (ABS). Therefore, U.S. EPA OSW recommends not evaluating the dermal exposure to soil pathway as part of the recommended exposure scenarios. However, if either a facility of a permitting authority feel that site-specific conditions indicate dermal exposure to soil may contribute significantly to total soil-related exposures, U.S. EPA OSW recommends following the methodologies described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press). Dermal exposure is further discussed in Section 6.2.3.2 of this guidance.

Inhalation of COPCs and Ingestion of Water by Animals - These exposure pathways have not been included in the recommended exposure scenarios because the contribution of these pathways to total risk is anticipated to be negligible in comparison with that of the exposure pathways being evaluated. However, these exposure pathways may need to be evaluated on a case-by-case basis considering site-specific exposure setting characteristics.

U.S. EPA OSW-recommended exposure scenarios are further discussed in the following subsections. Table 4-1 presents the exposure pathways that should be evaluated for each of the recommended exposure scenarios.

4.2.1 Subsistence Farmer

The subsistence farmer exposure scenario is evaluated to account for the combination of exposure pathways to which a receptor may be exposed in a farm or ranch exposure setting. U.S. EPA OSW recommends including this scenario, because indirect ingestion routes may represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1990e; 1994l; 1994g; NC DEHNR 1997); the significance of these exposures is primarily related to the potential for COPCs to bioaccumulate up the food chain. The evaluation of these exposure scenarios are consistent with U.S. EPA (1994g) and NC DEHNR (1997). As indicated in Table 4-1, the subsistence farmer is assumed to be 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

TABLE 4-1

RECOMMENDED EXPOSURE SCENARIOS FOR EVALUATION IN A HUMAN HEALTH RISK ASSESSMENT

	Recommended Exposure Scenarios[a]						
Exposure Pathways	Subsistence Farmer	Subsistence Farmer Child	Adult Resident	Child Resident	Subsistence Fisher	Subsistence Fisher Child	Acute Risk[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	•	•	—	—	—	—	—
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	c	c	c	—

Notes:

[a] Exposure scenarios are defined as a combination of exposure pathways evaluated for a receptor at a specific exposure scenario location (receptor grid node).

[b] The acute risk 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 and PCDFs 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] Regional specific exposure setting characteristics (e.g., presence of ponds on farms or within semi-rural residential areas, presence of lite livestock within semi-rural residential areas) may warrant that the permitting authority consider inclusion of this exposure pathway when evaluating a recommended exposure scenario (see Section 4.2).

- 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; see Chapter 2)

Previous U.S. EPA guidance documents (for example, U.S. EPA 1993h and U.S. EPA 1994f) have not included evaluating the concentration of COPCs in chicken and eggs. NC DEHNR (1997) considers chicken and egg ingestion pathways only for exposures to dioxins and furans, because *BCF* values were available in the literature only for dioxins and furans. Currently, biotransfer factors can be derived from literature data for other organic compounds and metals. Therefore, U.S. EPA OSW recommends the evaluation of the concentrations of all COPCs via chicken and egg ingestion exposure pathways. 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.

For the subsistence farmer scenario, the receptor is assumed to consume a fraction from each food group (beef, pork, poultry, eggs, and milk) to make up a total consumption rate, and all amounts consumed are assumed to be homegrown. This allows estimation of the relative contribution of COPC-specific risk from ingestion of each food group. If site-specific information is available that demonstrates that a subsistence farmer does not raise beef, poultry, or pork, and that raising any of these livestock would not occur for a reasonable potential future subsistence farmer at a location, then elimination of one or more of these exposure pathways from the risk evaluation could justifiably be considered. However, intakes rates of the food items consumed in the remaining exposure pathways may need to be adjusted upward to ensure that the total amount consumed (summed fraction from each food group) is representative of a subsistence level.

It should be noted that the ingestion rates of beef, poultry, eggs, and pork recommended (see Chapter 6 and Appendix C) for the subsistence farmer scenario represent a fraction of the total amount of meat and eggs consumed. Therefore, the approach of conducting the initial evaluation assuming ingestion of all meat groups by the subsistence farmer scenario does not grossly overestimate the total amount of meat a farmer or rancher could reasonably consume.

When evaluating the ingestion of drinking water from surface water for the subsistence farmer scenario, the potential for ingestion of cistern water at farm or ranch locations should also be considered in addition to surface water sources. If it can be determined based on available information, including site-specific information, interviews with local health departments, or other local information sources, that cistern water is likely to or could be used for a drinking water source, ingestion of cistern water should be evaluated similar to ingestion of water from a surface water body. Quantitative evaluation can be completed using the applicable estimating media concentration equations for ingestion of drinking water as presented in Chapter 5 and Appendix B.

The ingestion of fish exposure pathway is not recommended for automatic inclusion when evaluating the subsistence farmer exposure scenario. However, as indicated in the notes to Table 4-1, U.S. EPA OSW does recommend that the fish ingestion pathway be considered for evaluation 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. Quantitative evaluation can be completed using the applicable estimating media concentration equations for ingestion of fish as presented in Chapter 5 and Appendix B. Also, the permitting authority may elect to evaluate the subsistence fisher and subsistence fisher child exposure scenarios (see Sections 4.2.5 and 4.2.6) at farm or ranch locations where on-site farm ponds are used as a potential source of fish for the purpose of human consumption.

Exposure of an infant to PCDDs and PCDFs via the ingestion of breast milk is evaluated as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

4.2.2 Subsistence Farmer Child

The subsistence farmer child exposure scenario is evaluated to account for the combination of exposure pathways to which a receptor may be exposed in a farm or ranch setting. U.S. EPA OSW recommends including the subsistence farmer child scenario, because indirect ingestion routes may represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1990e; 1994l; 1994g;

NC DEHNR 1997); the significance of these exposures is primarily related to the potential for COPCs to bioaccumulate up the food chain. The evaluation of this exposure scenario is consistent with U.S. EPA (1994g) and NC DEHNR (1997), and new U.S. EPA policy focused on 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 and Section 4.2.1, the subsistence farmer child is assumed to be exposed to COPCs emitted from the facility through the same exposure pathways as the subsistence farmer.

4.2.3 Adult Resident

The adult resident exposure scenario is evaluated to account for the combination of exposure pathways to which a receptor may be exposed in an urban or rural (nonfarm) setting. U.S. EPA OSW recommends including the adult resident scenario, because potential exposure to COPCs through ingestion of homegrown produce has been shown to be potentially significant; the significance of these exposures is primarily related to the potential for COPCs to bioaccumulate up the food chain (U.S. EPA 1990e; 1994l; 1994g; NC DEHNR 1997). The evaluation of this exposure scenario is consistent with the evaluation of the "Home Gardener" scenario recommended by U.S. EPA (1994g) and NC DEHNR (1997). As indicated in Table 4-1, the adult resident is assumed to be 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; see Chapter 2)

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. Adult residents are assumed to grow some of their own produce (NC DEHNR 1997).

The ingestion of fish exposure pathway is not recommended for automatic inclusion when evaluating the resident adult exposure scenario. However, as indicated in the notes to Table 4-1, U.S. EPA OSW does recommend that the fish ingestion pathway be considered for evaluation if regional or site-specific exposure setting characteristics (e.g., presence of ponds within semi-rural residential areas that support fish for human consumption) are identified that warrant consideration. The permitting authority may elect to evaluate the subsistence fisher and subsistence 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 the purpose of human consumption.

Exposure of an infant to PCDDs and PCDFs via the ingestion of breast milk is evaluated as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

4.2.4 Child Resident

The child resident exposure scenario is evaluated to account for the combination of exposure pathways to which a child receptor may be exposed in an urban or rural (nonfarm) setting. U.S. EPA OSW recommends including the adult resident child scenario, because indirect ingestion routes may represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1990e; 1994l; 1994g; NC DEHNR 1997); the significance of these exposures is primarily related to the potential for COPCs to bioaccumulate up the food chain. The evaluation of this exposure scenario is consistent with the evaluation of the “Child of the Home Gardener” scenario recommended by U.S. EPA (1994g) and NC DEHNR (1997), and new U.S. EPA policy focused on 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 and Section 4.2.3, the child resident is assumed to be exposed to COPCs emitted from the facility through the same exposure pathways as the resident adult. The child resident is assumed to ingest some produce grown by the adult resident (NC DEHNR 1997).

4.2.5 Subsistence Fisher

The Subsistence Fisher exposure scenario is evaluated to account for the combination of exposure pathways to which a receptor may be exposed in an urban or rural setting where fish is the main component of the receptor diet. U.S. EPA OSW recommends including the subsistence fisher scenario, because indirect ingestion routes may represent significant potential exposure to COPCs released from combustion sources (U.S. EPA 1990e; 1994l; 1994g; NC DEHNR 1997); the significance of these exposures is primarily related to the potential for COPCs to bioaccumulate up the food chain. The evaluation of this exposure scenario is consistent with U.S. EPA (1994g) and NC DEHNR (1997). As indicated in Table 4-1, the subsistence fisher is assumed to be 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; see Chapter 2)

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. Subsistence fishers are assumed to grow some of their own produce (NC DEHNR 1997). There may be many subsistence fishers throughout parts of several U.S. EPA regions. In fact, areas that are suspected to include large numbers of subsistence fishers, such as southeast Texas and southern Louisiana, are also areas with numerous hazardous waste combustion units.

Exposure of an infant to PCDDs and PCDFs via the ingestion of breast milk is evaluated as an additional exposure pathway, separately from this exposure scenario (see Chapter 2).

4.2.6 Subsistence Fisher Child

The subsistence fisher child exposure scenario is evaluated to account for the combination of exposure pathways to which a receptor may be exposed in an urban or rural setting where fish is the main component of the receptor diet. U.S. EPA (1994g) and NC DEHNR (1997) do not specifically recommend evaluation of this exposure scenario. However, the evaluation of this exposure scenario is consistent with the adult/child pairings recommended by U.S. EPA (1994g) and NC DEHNR (1997) for the subsistence farmer and residents, and new U.S. EPA policy focused on 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 and Section 4.2.5, the subsistence fisher child is assumed to be exposed to COPCs emitted from the facility through the same exposure pathways as the subsistence fisher. The subsistence fisher child is assumed to ingest some produce grown by the subsistence fisher; this assumption is similar to that for adult and child residents (NC DEHNR 1997).

4.2.7 Acute Risk Scenario

In addition to long-term chronic effects evaluated in the other recommended exposure scenarios, the acute exposure scenario is evaluated to account for short-term effects of exposure to maximum 1-hour concentrations of COPCs in emissions (see Chapter 3) from the facility through direct inhalation of vapors and particles (see Table 4-1 and Chapter 7). A receptor may be exposed in an urban or rural setting where human activity or land use supports any of the recommended exposure scenarios, as well as, in commercial and industrial land use areas (excluding workers from the facility under direct evaluation in the risk assessment) not typically evaluated by application of the other recommended exposure scenarios. Workers from the facility under direct evaluation in the risk assessment are excluded in most cases, because there are other guidance and regulations for occupational exposures to hazardous waste and hazardous waste combustion emissions within the facility boundary (e.g., OSHA).

Further discussion evaluation of this recommended exposure scenario and associated exposure pathway, including numeric equations, parameters values, and COPC specific inputs, can be found in Chapter 7 and Appendices A, B, and C.

4.3 SELECTION OF EXPOSURE SCENARIO LOCATIONS

Exposure scenario locations are the receptor grid nodes (defined by UTM coordinates during air dispersion modeling, see Chapter 3) selected as the location for evaluating one or more of the recommended exposure scenarios. Specific receptor grid nodes are selected as exposure scenario locations based on evaluation of the magnitude of air parameter values estimated by ISCST3 (see Chapter 3) specific to current and reasonable potential future land use areas as defined during the exposure setting characterization (see Section 4.1). Air parameter values specific to the receptor grid node, selected as an exposure scenario location, are then used as inputs to the estimating media concentration equations when evaluating the recommended exposure scenario(s) for that location. U.S. EPA OSW would like to note that the methodology and resulting selection of receptor grid nodes as exposure scenario locations is one of the most critical parts of the risk assessment process, ensuring standardization across all facilities evaluated and reproducibility of results. The estimates of risk can vary significantly in direct response to which receptor grid nodes are selected as exposure scenario locations; and therefore, which ISCST3 modeled air parameter values are used as inputs into the estimating media equations.

To ensure consistent and reproducible risk assessments, U.S. EPA OSW recommends that, at a minimum, the following procedures be used in the selection of receptor grid nodes as exposure scenario locations, and that selected exposure scenario locations correspond to actual ISCST3 modeled receptor grid node locations defined by UTM coordinates. In addition to consistency and reproducibility, these procedures ensure that the exposure scenario location(s) selected for evaluation over a specified land use area do not overlook locations within that same land use area that would result in higher risk. This is especially important when considering the complexity of multiple modeled air parameters and phases per location, potentially multiple facility emission sources, and multiple source-specific COPCs. This approach also provides more complete risk evaluation of areas surrounding the facility; information often required later in the permitting process and in risk communication to the surrounding public. Therefore, U.S. EPA OSW recommends that, at a minimum, a risk assessment initially evaluate 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 as follows:

> ***Step 1: Define Land Use Areas To Be Evaluated*** - Current and reasonable potential future land use areas, water bodies, and watersheds identified during exposure setting characterization for evaluation in the risk assessment, should be defined and mapped using UTM coordinates in a format (NAD27 or NAD83 UTM) consistent with that used to define locations of facility emission sources and the ISCST3 receptor grid nodes.
>
> ***Step 2: Identify Receptor Grid Node(s) Within Each Land Use Area To Be Evaluated*** - For each land use area to be evaluated, identify the receptor grid nodes within that area or on the boundary of that area (defined in Step 1) that represent the location of highest yearly average concentration 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) and all emission sources at the facility combined. This results in the selection of one or more receptor grid nodes as one or more exposure scenario locations, within the land use area to be evaluated, that meet the following criteria:

- Highest modeled unitized vapor phase air concentration
- Highest modeled unitized vapor phase wet deposition rate
- Highest modeled unitized particle phase air concentration
- Highest modeled unitized particle phase wet deposition rate
- Highest modeled unitized particle phase dry deposition rate
- Highest modeled unitized particle-bound phase air concentration
- Highest modeled unitized particle-bound phase wet deposition rate
- Highest modeled unitized particle-bound phase dry deposition rate

With the exception of water bodies and watersheds (discussed in Step 4 below), only ISCST3 modeled air parameters corresponding to a single receptor grid node should be used per exposure scenario location as inputs into the media equations, without averaging or statistical manipulation. However, based generally on the number and location of facility emission sources, multiple exposure scenario locations may be selected for a specific land use area being evaluated. Application of these criteria for land use areas being evaluated in U.S. EPA Region 6 for actual sites, using actual modeled air parameters, indicates that only 1 to 3 receptor grid nodes are typically selected per land use area. This is because, in most cases, the location of highest air concentration and deposition rate occurs at the same receptor grid node. It should also be noted, that while these criteria minimize overlooking maximum risk within a land use area, they do not preclude the risk assessor 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 based on the criteria specified above, additional receptor grid nodes within the assessment area may need to be considered as exposure scenario locations for the evaluation of acute risk or site-specific risk considerations (e.g., special subpopulations). In land use areas to be evaluated for acute risk (could potentially include commercial and industrial land use areas), 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 and all emission sources at the facility combined, should be selected as the exposure scenario location(s). For site-specific risk considerations, the closest receptor grid node to the exposure point being evaluated should be considered for the exposure scenario location. However, in some cases, a more conservative approach may require selection of 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 evaluation of water bodies and their associated watersheds, the receptor grid nodes within their area extent or "effective" areas (defined and mapped in Step 1) should be considered. For water bodies, the risk assessor can select the receptor grid node with the highest modeled air parameter values or average the air parameter values for all receptor grid nodes within the area of the water body. For watersheds, the modeled air parameter values should be averaged for all receptor grid nodes within the area extent or effective area of the watershed (excluding the area of the water body). For water bodies and watersheds, air parameters to be considered as required by the estimating media concentration equations in Chapter 5 and Appendix B include 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) and all emission sources at the facility combined.

For the purpose of evaluating potential exposure routes other than ingestion of fish, the subsistence fisher and subsistence fisher child should be assumed to be located at selected exposure scenario locations where the adult resident scenario is evaluated. In addition, the subsistence fisher and subsistence fisher child exposure scenarios should be assumed to be exposed through ingestion of fish from the water body having the highest modeled combined deposition, and can or does support fish populations. In some cases, site specific conditions may require that the subsistence fisher and subsistence fisher child exposure scenarios be evaluated assuming exposure through ingestion of fish be calculated using COPC water concentrations from one water body, and exposure from ingestion of drinking water be calculated using COPC water concentrations from a different water body.

The recommended ISCST3 modeled receptor grid node array extends out about 10 km from facility emission sources (see Chapter 3). To address evaluation of land use areas, water bodies, or watersheds located beyond the coverage provided by the recommended receptor grid node array (greater than 10 km from the facility), the ISCST3 modeling can be conducted with an additional receptor grid node array specified to provide coverage of the area of concern, or the steps above can be executed using the closest receptor grid nodes from the recommended array. However, using the closest receptor grid nodes from the recommended receptor grid node array will in most cases provide an overly conservative estimate of risk since the magnitude of air parameter values at these receptor grid nodes would most likely be higher than at receptor grid nodes located further from the facility sources and actually within the area of concern.

Chapter 5
Estimation of Media Concentrations

What's Covered in Chapter 5:

- Calculation of COPC Concentrations in Air for Direct Inhalation
- Calculation of COPC Concentrations in Soil
- Calculation of COPC Concentrations in Produce
- Calculation of COPC Concentrations in Beef and Dairy Products
- Calculation of COPC Concentrations in Pork
- Calculation of COPC Concentrations in Chicken and Eggs
- Calculation of COPC Concentrations in Drinking Water and Fish

The purpose of this chapter is to describe the estimating media concentration equations and associated parameters used in evaluation of the recommended exposure scenarios presented in Chapter 4. The origin and development of each of these equations, and description of associated parameters, are presented in most cases. The equations are also presented in Appendix B without derivation, and organized according to exposure pathway. Discussions of ISCST3 modeled unitized air parameters and compound specific parameters required in the estimating media concentration equations are presented in Chapter 3 and Appendix A-3, respectively. Appendix A-3 also provides recommended values for the compound specific parameters. Equations for use in modeling phase allocation and speciation of mercury concentrations are presented and discussed in Appendix B. Also, it should be noted that reference made throughout Chapter 5 to particle phase is generic and made without distinction between particle and particle-bound.

Section 5.1 describes the estimating media concentration equations used to support evaluation of direct inhalation of COPCs. Section 5.2 describes the estimating media concentration equations for soils contaminated by COPCs. Section 5.3 describes the estimating media concentration equations used to determine COPC concentrations in produce. Sections 5.4 through 5.6 describe equations used to determine COPC concentrations in animal product (such as milk, beef, pork, poultry, and eggs) resulting from animal ingestion of contaminated feed and soil. Section 5.7 describes equations used to determine 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.

5.1 CALCULATION OF COPC CONCENTRATIONS IN AIR FOR DIRECT INHALATION

COPC concentrations in air are calculated by summing the vapor phase and particle phase air concentrations of COPCs. Air concentrations used in the evaluation of long-term or chronic exposure, via direct inhalation, should be calculated using unitized yearly air parameter values as specified in Appendix B, Table B-5-1. Air concentrations used in the evaluation of short-term or acute exposure, via direct inhalation, should be calculated using unitized hourly air parameter values as specified in Appendix B, Table B-6-1.

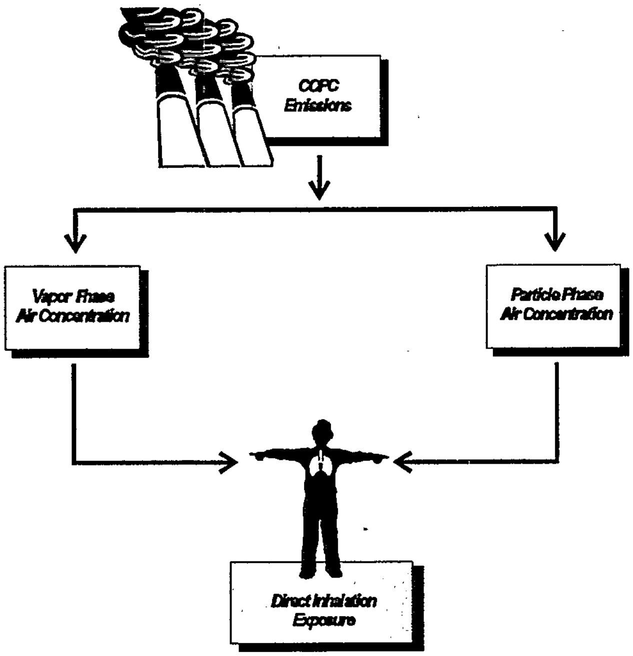


Figure 5-1 - COPC Concentration in Air for Direct Inhalation

5.2 CALCULATION OF COPC CONCENTRATIONS IN SOIL

COPC concentrations in soil are calculated by summing the vapor phase and particle phase deposition of COPCs to the soil. Wet and dry deposition of particles and vapors are considered, with dry deposition of vapors calculated from the vapor air concentration and the dry deposition velocity. The calculation of soil concentration incorporates a term that accounts 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. Equations for the calculation of soil concentration and soil losses of COPCs are presented in Appendix B, Tables B-1 for land use areas, and Tables B-4 for watersheds (see Section 5.7).

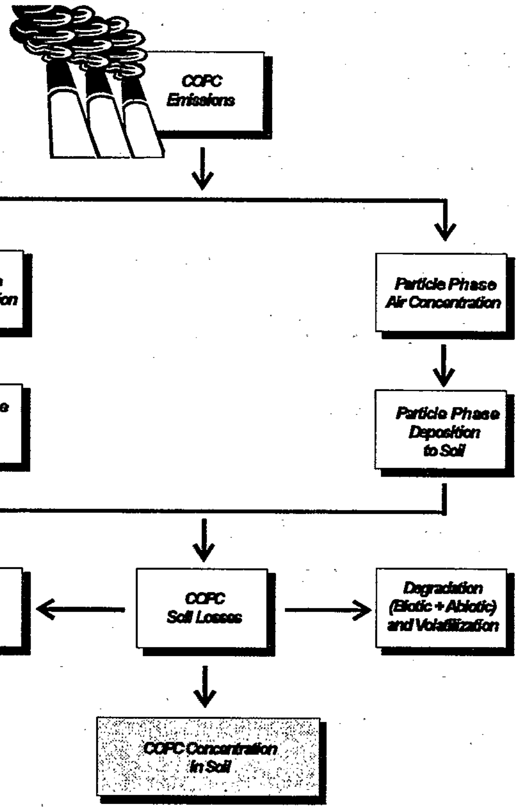


Figure 5-2 - COPC Concentration in Soil

Soil concentrations may require many years to reach steady state. As a result, the equations used 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, U.S. EPA OSW recommends using two variations of the equation (average soil concentration over exposure duration):

(1) one variation to be used if the exposure duration is greater than or equal to the operating lifetime of the emission source or time period of combustion, and

(2) the other form to be used if the exposure duration is less than the operating lifetime of the emission source or time period of combustion.

For noncarcinogenic COPCs, U.S. EPA OSW recommends using the second form of the carcinogenic equation to calculate the highest 1-year annual average soil concentration; typically occurring at the end of the operating lifetime of the emission source. These equations are described 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. Loss of COPCs from the soil is modeled by using rates that depend on the physical and chemical characteristics of the soil. These variables and their use are described in the following subsections, along with the recommended equations.

5.2.1 Calculating Cumulative Soil Concentration (*Cs*)

U.S. EPA (1990e) recommended the use of the following equation—adapted from Travis, Baes, and Barnthouse (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)
$Dydp$ = 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)
100 = Units conversion factor (mg-m^2/kg-cm^2)
Z_s = Soil mixing zone depth (cm)
BD = Soil bulk density (g soil/cm^3 soil)

$$Cs = \frac{100 \cdot (Dydp + Dywv + L_{dif}) \cdot [1.0 - \exp\ (-ks \cdot tD)]}{Z_s \cdot BD \cdot ks} \qquad \text{Equation 5-1A}$$

U.S. EPA (1993h) stated that this equation evaluated deposition of particle phase COPCs, but fails to consider vapor phase deposition or diffusion. To account for vapor phase diffusion, U.S. EPA (1993h) recommended using the following equation:

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)

However, subsequent U.S. EPA guidance (1994g) recommended the use of the original Equation 5-1, recommended by U.S. EPA (1990e), but limited its use to calculating cumulative soil concentration (*Cs*) for 2,3,7,8-TCDD only. The discussion stated that the COPC soil loss constant (*ks*) is equal to 0 for all other COPCs (U.S. EPA 1994g). For COPCs other than 2,3,7,8-TCDD, the following equation—which eliminates the COPC soil loss constant—was recommended by U.S. EPA (1994g):

$$Cs = 100 \cdot \frac{Dydw + Dyww}{Z_s \cdot BD} \cdot tD \qquad \text{Equation 5-1B}$$

where

Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
100 = Units conversion factor (mg-m^2/kg-cm^2)
$Dydw$ = Dyd (s/m^2-yr)
$Dyww$ = Dyw (s/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 two different equations (Equations 5-1C and 5-1D) for use 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.

U.S. EPA OSW recommends the use of Equations 5-1C, 5-1D, and 5-1E to calculate the cumulative soil concentration (Cs). The use of these equations is further described in Appendix B, Table B-1-1.

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)
T_1 = Time period at the beginning of combustion (yr)
ks = COPC soil loss constant due to all processes (yr^{-1})

tD = Time period over which deposition occurs (time period of combustion) (yr)
Cs_{tD} = Soil concentration at time tD (mg/kg)
T_2 = Length of exposure duration (yr)

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), when an exposure duration that is less than or equal to the operating lifetime of the emission source or hazardous waste combustion unit ($T_2 \leq tD$), Equation 5-1C is recommended; when an exposure duration greater than the operating lifetime of the hazardous waste combustion unit ($T_1 < tD < T_2$), Equation 5-1D is recommended. For noncarcinogenic COPCs, Equation 5-1E is recommended.

The COPC soil concentration averaged over the exposure duration, represented by Cs, should be used for carcinogenic compounds, 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, the highest annual average COPC soil concentration occurring during the exposure duration period should be used for noncarcinogenic COPCs. The highest annual average COPC soil concentration, Cs_{tD}, will typically occur at the end of the operating life of the emission source or the time period of combustion.

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends using the highest 1-year annual average soil concentration, determined by 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 may 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*). The constant *ks* is determined by using the soil's physical, chemical, and biological characteristics to consider the loss resulting from:

(1) leaching,

(2) runoff,

(3) erosion,

(4) biotic and abiotic degradation, and

(5) volatilization.

U.S. EPA (1990e) recommended the use of the following equation to calculate the soil loss constant (*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})

U.S. EPA OSW recommends that Equation 5-2A be used to calculate the COPC soil loss constant (*ks*). This equation is further described in Appendix B, Table B-1-2. The use of Equation 5-2A is consistent with U.S. EPA (1993h), U.S. EPA (1994g), U.S. EPA (1994r), 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})

As highlighted in Section 5.2.1, the use of Equation 5-2A in Equations 5-1C and 5-1D assumes that COPC loss can be defined by using first-order reaction kinetics. First-order reaction rates depend on the concentration of one reactant (Bohn, McNeal, and O'Connor 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 1990e).

COPC loss in soil can also follow zero or second-order reaction kinetics. Zero-order reaction kinetics are independent of reactant concentrations (Bohn, McNeal, and O'Connor 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, McNeal, and O'Connor 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, the use of Equation 5-2A may also overestimate loss rates for each process (Valentine 1986). When possible, the common occurrence of all loss processes should be taken into account. Combined rates of soil loss by these processes can be derived experimentally; values for some COPCs are presented in U.S. EPA (1986c).

Sections 5.2.2.1 through 5.2.2.5 discuss issues associated with the calculation of the *ksl*, *kse*, *ksr*, *ksg*, and *ksv* variables.

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 addressed in the literature (U.S. EPA 1990e). Lyman et al. (1982) states that degradation rates can be assumed to follow first order kinetics in a homogenous media. Therefore, the half-life of a compound can be related 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, the degradation rate can be calculated. However, literature sources do not provide sufficient data for all such mechanisms, especially for soil. U.S. EPA guidance (1994g) had stated that *ksg* values for all COPCs other than 2,3,7,8-TCDD should be set equal to zero. Appendix A-3 presents U.S. EPA OSW recommended values for this compound specific variable.

Recommended Values for:
COPC Loss Constant Due to Biotic and Abiotic Degradation (ksg)

See Appendix A-3

The rate of biological degradation in soils depends on the concentration and activity 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 variables in a single soil system. However, the use of simple rate expressions may be appropriate at low chemical concentrations (e.g., nanogram per kilogram soil) at which a first-order dependence on chemical concentration may be reasonable. The rate of biological degradation is COPC-specific, depending on the complexity of the COPC and the usefulness of the COPC to the microorganisms. Some substrates, 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 may limit the biodegradation of COPCs in the soil environment (Valentine and Schnoor 1986) include:

(1) availability of the COPC,

(2) nutrient limitations,

(3) toxicity of the COPC, and

(4) inactivation or nonexistence of enzymes capable of degrading the COPC.

Chemical degradation of organic compounds can be a significant mechanism for removal of COPCs in soil (U.S. EPA 1990e). 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, and these expressions are helpful when division into component reactions is not 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. The total rate of hydrolysis in soil can be predicted by adding the rates in the soil and water phases, which are assumed to be first-order reactions at a fixed pH (Valentine 1986). Methods for estimating these hydrolysis constants are described by Lyman et al. (1982).

Organic and inorganic compounds also undergo oxidation-reduction (redox) reactions in the soil (Valentine 1986). Organic redox reactions involve the exchange of oxygen and hydrogen atoms by the reacting molecules. Inorganic redox reactions may involve the exchange of atoms or electrons by the reactants. In soil systems where the identities of oxidant and reductant species are not specified, a first-order rate constant can be obtained 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 (1993h) recommended the use of 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)
Kd_s = Soil-water partition coefficient (mL water/g soil)
BD = Soil bulk density (g soil/cm^3 soil)
Z_s = Soil mixing zone depth (cm)
θ_{sw} = Soil volumetric water content (mL water/cm^3 soil)

Unit soil loss (X_e) is calculated by using the Universal Soil Loss Equation (USLE) (See Section 5.7.2). Soil bulk density (*BD*) is described in Section 5.2.5.2. Soil volumetric water content (θ_{sw}) is described in Section 5.2.5.4. Site-specific variables associated with Equation 5-3 are further discussed in Appendix B.

U.S. EPA guidance (1994g and 1994r) have stated that all *kse* values are equal to zero. U.S. EPA (1994r) stated that *kse* is equal to zero because of contaminated soil eroding onto and off of the site.

Consistent with U.S. EPA guidance (1994g and 1994r), U.S. EPA OSW recommends that the constant for the loss of soil resulting from erosion (*kse)* should be set equal to zero.

Recommended Value for: **COPC Loss Constant Due to Erosion (*kse*)**
0

For additional information on addressing *kse*, U.S. EPA OSW recommends consulting the methodologies described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press). The use of *kse* values is also further described in Appendix B, Table B-1-3.

5.2.2.3 COPC Loss Constant Due to Runoff (*ksr*)

Consistent with U.S. EPA (1993h; 1994r) and NC DEHNR (1997), U.S. EPA OSW recommends that Equation 5-4 be used to calculate the constant for the loss of soil resulting from surface runoff (*ksr*). The use of this equation is further described 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)
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)

Earlier U.S. EPA guidance (1994g) has stated that all *ksr* values should be set equal to zero. Soil bulk density (*BD*) is described in Section 5.2.5.2. Soil volumetric water content (θ_{sw}) is described in Section 5.2.5.4.

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 that Equation 5-5 be used to calculate the COPC loss constant due to leaching (*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)
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 (1993h) determined that Equation 5-5 does not properly account for surface runoff. U.S. EPA (1994g) stated that all *ksl* values should be set equal to zero.

More recent guidance (U.S. EPA 1993h; U.S. EPA 1994r; NC DEHNR 1997) have recommended Equation 5-5A to calculate the COPC loss constant due to leaching. Consistent with U.S. EPA (1993h), U.S. EPA (1994r), and NC DEHNR (1997), U.S. EPA OSW recommends that Equation 5-5A be used to calculate the COPC loss constant due to leaching (*ksl*) to account for runoff. The use of this equation is further described 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)
Z_s = Soil mixing zone depth (cm)
Kd_s = Soil-water partition coefficient (cm^3 water/g soil)
BD = Soil bulk density (g soil/cm^3 soil)

Appendix B describes the determination of site-specific variables associated with Equation 5-5A. The average annual volume of water ($P + I - RO - E_v$) available to generate leachate is the mass balance of all water inputs and outputs from the area under consideration. These variables are described in Section 5.2.5.3. Soil bulk density (*BD*) is described in Section 5.2.5.2. Soil volumetric water content (θ_{sw}) is described in Section 5.2.5.4.

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) recommended the use of Equation 5-6 to calculate the constant for the loss of soil resulting from volatilization (*ksv*).

$$ksv = Ke \cdot Kt \qquad \text{Equation 5-6}$$

where

ksv = COPC loss constant due to volatization (yr^{-1})
Ke = Equilibrium coefficient (s/cm-yr)
Kt = Gas phase mass transfer coefficient (cm/s)

U.S. EPA (1993h) did not identify a reference for Equation 5-6. However, U.S. EPA (1993h) stated that Equation 5-6 had not 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 (1993h) 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 (1993h) recommended that the volatilized residues of semi-volatile COPCs (such as dioxin) not be considered, but that additional research be conducted to determine the magnitude of the uncertainty introduced for volatile COPCs. U.S. EPA (1994g) stated that all *ksv* values should be set to zero.

U.S. EPA guidance (1994r) and NC DEHNR (1997) recommended calculating *ksv* values using Equation 5-6A. Equation 5-6A appears to incorporate equations that U.S. EPA (1990e) recommended for use in 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 volatization (yr^{-1})
3.1536×10^{7} = Units conversion factor (s/yr)
H = Henry's Law constant ($atm\text{-}m^3/mol$)
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)
W = Average annual wind speed (m/s)
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)
A = Surface area of contaminated area (m^2)
0.78 = Empirical constant (unitless)
-0.67 = Empirical constant (unitless)
-0.11 = Empirical constant (unitless)
0.482 = Units conversion factor [(3600 s/hr)$^{0.78}$(100 cm/m)/(3600 s/hr)] · (empirical constant 0.0292)

U.S. EPA (1990e) recommended that Equation 5-7 be used to calculate *Ke* and Equation 5-8 be used 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)
W = Average annual wind speed (m/s)
Sc_a = Schmidt number for gas phase (unitless)
d_e = Effective diameter of contaminated media (m)

0.482 = Units conversion factor $[(3600\ \text{s/hr})^{0.78}(100\ \text{cm/m})/(3600\ \text{s/hr})] \cdot$ (empirical constant 0.0292)

Equations 5-9 and 5-10 are used to calculate the Schmidt number for gas phase (Sc_a) and the effective diameter of contaminated media (d_e) respectively (U.S. EPA 1990e).

$$Sc_a = \frac{\mu_a}{\rho_a \cdot D_a} \qquad \text{Equation 5-9}$$

$$d_e = \sqrt{\frac{4 \cdot A}{\pi}} \qquad \text{Equation 5-10}$$

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)

Consistent with U.S. EPA guidance (1994g) and based on the need for additional research to be conducted to determine the magnitude of the uncertainty introduced for modeling volatile COPCs from soil, U.S. EPA OSW recommends that, until identification and validation of more applicable models, the constant for the loss of soil resulting from volatilization (*ksv)* should be set equal to zero.

Recommended Value for: **COPC Loss Constant Due to Volatilization (*ksv*)**
0

In cases where high concentrations of volatile organic compounds are expected to be present in the soil, U.S. EPA OSW recommends consulting the methodologies described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press). The use of *ksv* values is also further described in Appendix B, Table B-1-6.

5.2.3 Calculating the Deposition Term (*Ds*)

U.S. EPA OSW recommends that Equation 5-11 be used to calculate the deposition term *(Ds)*. This equation is further described in Appendix B, Table B-1-1. The use of Equation 5-11 to calculate the deposition term is consistent with U.S. EPA (1994r) and NC DEHNR (1997), which both incorporate a deposition term (*Ds*) into Equation 5-1C.

Recommended Equation for Calculating:
Deposition Term (*Ds*)

$$Ds = \left[\frac{100 \cdot Q}{Z_s \cdot BD}\right] \cdot [F_v \cdot (0.31536 \cdot Vdv \cdot Cyv + Dywv) + (Dydp + Dywp) \cdot (1 - F_v)] \qquad \text{Equation 5-11}$$

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)
F_v = Fraction of COPC air concentration in vapor phase (unitless)
0.31536 = Units conversion factor (m-g-s/cm-µg-yr)
Vdv = Dry deposition velocity (cm/s)
Cyv = Unitized yearly average air concentration from vapor phase (mg-s/g-m^3)
$Dywv$ = 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)

Chapter 3 describes determination of modeled air parameters *Cyv*, *Dywv*, *Dydp*, and *Dywp*. Appendix B describes determination of the site-specific parameters associated with Equation 5-11. Appendix A-3 describes determination of the compound-specific parameter *Fv*.

5.2.4 Universal Soil Loss Equation (USLE)

U.S. EPA OSW recommends that the universal soil loss equation (USLE) be used to calculate the unit soil loss (X_e). This equation is further described in Section 5.7.2 and in Appendix B, Table B-4-13.

5.2.5 Site-Specific Parameters for Calculating Cumulative Soil Concentration

Calculating average soil concentration over the exposure duration (Cs) requires the use of site-specific parameters including the following:

- Soil mixing zone depth (Z_s)
- Soil bulk density (BD)
- Available water ($P + I - RO - E_v$)
- Soil volumetric water content (θ_{sw})

Determination of values for these parameters is further described in the following subsections, and in Appendix B.

5.2.5.1 Soil Mixing Zone Depth (Z_s)

When exposures to COPCs in soils are modeled, the depth of contaminated soils is important in calculating the appropriate soil concentration. COPCs deposited onto soil surfaces may be moved into lower soil profiles by tilling, whether manually in a garden or mechanically in a large field.

In general, U.S. EPA (1990e) and U.S. EPA (1992d) have 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 are assumed to be retained in the shallower, upper soil layer. In this case, earlier U.S. EPA guidances (U.S. EPA 1990e; U.S. EPA 1993h) have typically recommended a value of 1 centimeter.

The assumption made to determine the value of Z_s 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:

(1) ingestion of plants contaminated by root uptake and by volatilization from soil;

(2) direct ingestion of soil by humans, cattle, swine, or chicken; and

(3) 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. However, in calculations of plant uptake resulting from volatilization, only the uppermost soil layer is considered.

U.S. EPA (1990e) recommended that soil mixing depths be selected 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.

Consistent with U.S. EPA (1990e), U.S. EPA OSW recommends the following values for the soil mixing zone depth (Z_s).

Recommended Values for:
Soil Mixing Zone Depth (Z_s)

1 cm - untilled
20 cm - tilled

U.S. EPA guidance (1990e) stated that any volatile COPCs are not likely to be associated with particulates soon after emission from the combustion unit; before deposition onto the soil. However, semi-volatile COPCs and volatile COPCs emitted in sufficiently high concentrations may be deposited in particulate form and exhibit volatilization losses from soils. COPCs subject to volatilization losses may be moved to 20 centimeters by tilling and will not readily volatilize from this depth. The volatilization rate will reflect only the COPC concentration at the soil surface.

5.2.5.2 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). Consistent with U.S. EPA (1990e; 1994g) and presented in Hoffman and Baes (1979), U.S. EPA OSW recommends the following value for the soil dry bulk density (*BD*).

Recommended Value for: **Soil Dry Bulk Density (*BD*)**
1.50 g/cm^3

U.S. EPA (1994r) recommended that wet soil bulk density be determined 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.

5.2.5.3 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 may apply in the various U.S. EPA regions.

The average annual precipitation (*P*), irrigation (*I*), runoff (*RO*), and evapotranspiration (E_v) rates and other climatological data may be obtained 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)—may also be found in resources such as Geraghty, Miller, van der Leeden, and Troise (1973). Surface runoff may also be estimated by using the Curve Number Equation developed by the U.S. Soil Conservation Service (U.S. EPA 1990e). U.S. EPA (1985b) cited isopleths of mean annual cropland runoff corresponding to various curve numbers developed by Stewart, Woolhiser, Wischmeier, Caro, and Frere (1975). Curve numbers are assigned to an area on the basis of soil type, land use or cover, and the hydrologic conditions of the soil (U.S. EPA 1990e).

Using these different references, however, introduces uncertainties and limitations. For example, Geraghty, Miller, van der Leeden, and Troise (1973) presented isopleths for annual surface water contributions that include interflow and ground water recharge. As noted in U.S. EPA (1994g), these values should be adjusted downward to reflect surface runoff only. U.S. EPA (1994g) recommended that these values be reduced by 50 percent.

5.2.5.4 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 U.S. EPA regions. Consistent with earlier guidance documents, (U.S. EPA 1993k; U.S. EPA 1994g; NC DEHNR 1977), U.S. EPA OSW recommends a value for θ_{sw} of 0.2 ml/cm^3.

Recommended Value for: **Soil Volumetric Water Content (θ_{sw})**
0.2 ml/cm^3

5.3 CALCULATION OF 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, consideration of indirect exposure separates produce into two broad categories—aboveground produce and belowground produce. In addition, aboveground produce should be further subdivided into exposed and protected aboveground produce for consideration of contamination as a result of indirect exposure.

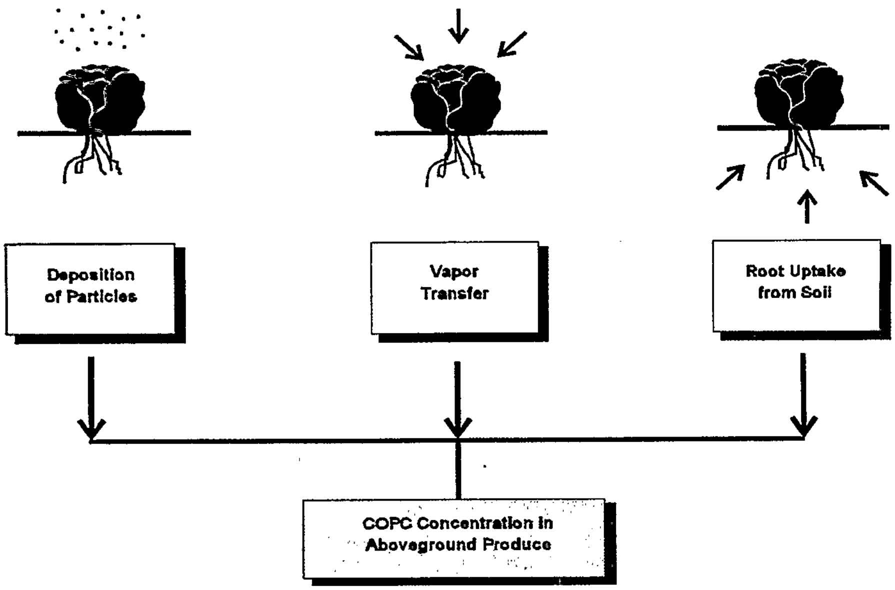


Figure 5-3 COPC Concentration in Produce

Aboveground Produce

Aboveground exposed produce is 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).

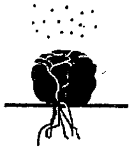
The total COPC concentration in aboveground exposed produce is calculated as a sum of contamination occurring through all three of these mechanisms. However, edible portions of aboveground protected produce, such as peas, corn, and melons, are covered by a protective covering; hence, they are protected from contamination through deposition and vapor transfer. Therefore, 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 used to calculate COPC concentrations in exposed and protected aboveground produce.

Belowground Produce

For belowground produce, contamination is assumed to occur only through one mechanism—root uptake of COPCs available from soil (Section 5.3.3). Contamination of belowground produce via direct deposition of particles and vapor transfer are not considered because the root or tuber is protected from contact with contaminants in the vapor phase. Appendix B further describes the equations and parameters used to calculate COPC concentrations in belowground produce.

Generally, risks associated with exposure of VOCs via food-chain pathways have not been considered significant, 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, U.S. EPA OSW recommends that all COPCs, including VOCs, be evaluated for each exposure pathway.

5.3.1 Aboveground Produce Concentration Due to Direct Deposition (*Pd*)



Earlier guidance documents (U.S. EPA [1990e]) and U.S. EPA [1993h]) proposed that COPC concentrations in aboveground vegetation resulting from wet and dry deposition onto plant surfaces of leafy plants and exposed produce (*Pd*) be calculated as follows:

$$Pd_i = \frac{1{,}000 \cdot [Dyd + (Fw \cdot Dyw)] \cdot Rp_i \cdot [1.0 - \exp(-kp \cdot Tp_i)]}{Yp_i \cdot kp} \qquad \text{Equation 5-13}$$

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 = 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).

U.S. EPA OSW recommends the use of Equation 5-14 to calculate COPC concentration in exposed and aboveground produce due to direct deposition. The use of this equation is further described in Appendix B, Table B-2-7.

Recommended Equation for Calculating:
Aboveground Produce Concentration Due to Direct Deposition (*Pd*)

$$Pd = \frac{1{,}000 \cdot Q \cdot (1 - F_v) \cdot [Dydp + (Fw \cdot Dywp)] \cdot Rp \cdot [1.0 - \exp(-kp \cdot Tp)]}{Yp \cdot kp} \qquad \text{Equation 5-14}$$

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 = Fraction of COPC wet deposition that adheres to plant surfaces (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)

Chapter 3 describes the determination of the modeled air parameters *Dydp* and *Dywp*. Appendix A-3 describes determination of *Fv*. Appendix B describes determination of *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 (1990e) stated that NRC models assumed a constant of 0.2 for *Rp* for dry and wet deposition of particles (Boone, Ng, and Palm 1981). However, Shor, Baes, and Sharp (1982) suggested that diversity of plant growth necessitated vegetation-specific *Rp* values.

As summarized in Baes, Sharp, Sjoreen, and Shor (1984), experimental studies of pasture grasses identified a correlation between initial *Rp* values and productivity (standing crop biomass [*Yp*]) (Chamberlain 1970):

$$Rp = 1 - e^{- \; Yp} \qquad \text{Equation 5-14A}$$

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, Sharp, Sjoreen, and Shor [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, Sharp, Sjoreen, and Shor (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, Sharp, Sjoreen, and Shor (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, Sharp, Sjoreen, and Shor 1984). Class-specific empirical constants ( ) that were developed include the following:

- 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, which is based on a weighted average class-specific *Rp* values. Specifically, class-specific *Rp* values were calculated by 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, Sharp, Sjoreen, and Shor (1984).
- Fruits, fruiting vegetables, and legumes were assigned the empirical constant (0.0324) originally developed by Baes, Sharp, Sjoreen, and Shor (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 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 1997c). 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) has resulted in an overall *Rp* value of 0.2 (RTI 1997).

For purposes of consistency, the produce classes have been combined into two groups—exposed fruit and exposed vegetables. The exposed produce empirical constant ( ) was used to calculate *Rp*. Since the exposed vegetable category includes leafy and fruiting vegetables, *Rp* was calculated for leafy and fruiting vegetables. The exposed vegetable *Rp* was then determined 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 were derived from the intake of homegrown produce discussion presented in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c). U.S. EPA recommends the use of 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 were 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, Sharp, Sjoreen, and Shor (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, Sharp, Sjoreen, and Shor (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 (1990e) 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 (1990e) 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 (1993h; 1994r) 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).

U.S. EPA OSW recommends use of a plant surface loss coefficient (*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.

Recommended Value for: Plant Surface Loss Coefficient (*kp*)
18

The primary uncertainty associated with this variable is that the calculation of *kp* does not consider chemical degradation processes. However, information regarding chemical degradation of contaminants on plant surfaces is limited. The inclusion of chemical degradation processes would result in decreased half-life values and thereby increase *kp* values. Note that effective plant concentration decreases as *kp* increases. Therefore, use of a *kp* value that does not consider chemical degradation processes is conservative. 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), U.S. EPA (1993h), U.S. EPA (1994r), and NC DEHNR (1997) recommended treating *Tp* as a constant, based on the average period between successive hay harvests. This period was estimated at 60 days (0.164 years) by Belcher and Travis (1989) and represents the length of time that aboveground vegetation (in this case, hay) would be exposed to contaminant deposition before being harvested. *Tp* is calculated 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)

Consistent with previous guidance, U.S. EPA OSW recommends using a *Tp* value of 0.164 year as the best available default value.

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 the use of 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 may be used to estimate a site-specific *Tp* value. Calculated plant concentrations will be affected most if the site-specific value of *Tp* is significantly less than 60 days.

5.3.1.4 Standing Crop Biomass (Productivity) (*Yp*)

U.S. EPA (1990e) stated that the best estimate of *Yp* is productivity, which Baes, Sharp, Sjoreen, and Shor (1984) and Shor, Baes, and Sharp (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) recommend using this equation and calculate *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 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.

Class-specific *Yp* values were estimated by 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 by using average class-specific conversion factors (Baes, Sharp, Sjoreen, and Shor 1984). U.S. EPA (1994r) and U.S. EPA (1995e) calculated class-specific *Yp* values and then used relative ingestion rates of each group to determine the weighted average *Yp* value of 1.6. However, the produce classes and relative ingestion values used by U.S. EPA (1994r) and U.S. EPA (1995e) 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 U.S. EPA (1995e) has resulted in an overall *Yp* value of 1.7 (RTI 1997).

For consistency, the produce classes have been combined into two groups—exposed fruit and exposed vegetables. The exposed vegetable *Yp* was determined by summing *Yh* values for leafy and fruiting vegetables and dividing by the sum of *Ah* values for leafy and fruiting vegetables. The relative ingestion rates used to determine an overall average weighted *Yp* value were derived from the homegrown produce discussions presented in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c). U.S. EPA recommends the use of the weighted average *Yp* value of 2.24 as a default *Yp* value based on this value representing the most complete and thorough information available.

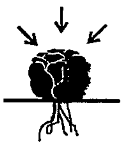
Recommended Value for: Standing Crop Biomass (Productivity) (*Yp*)
2.24

Unweighted *Yp* and ingestion rates used for the weighting were 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, the magnitude of the uncertainty introduced by the default *Yp* value can be estimated.

5.3.2 Aboveground Produce Concentration Due to Air-to-Plant Transfer (*Pv*)



The methodology used to estimate COPC concentration in exposed and aboveground produce due to air-to-plant transfer (*Pv*) considers limitations of COPCs concentrations to transfer from plant surfaces to the inner portions of the plant. These limitations result from mechanisms responsible for inhibiting the transfer of the lipophilic COPC (e.g., the shape of the produce) and the removal of the COPCs from the edible portion of the produce (e.g., washing, peeling, and cooking).

U.S. EPA OSW recommends the use of Equation 5-18 to calculate aboveground produce concentration due to air-to-plant transfer (*Pv*). The use of this equation is further described 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)

Chapter 3 describes the determination of the modeled air parameter *Cyv*. Appendix A-3 describes determination of *Fv* and Bv_{ag}. Appendix B further describes use of Equation 5-18, including determination of *Fw* and ρ_a. As discussed below in Section 5.3.2.1, the parameter VG_{ag} is dependent on lipophilicity of the COPC, and assigned a value of 0.001 for lipophilic COPCs (log K_{ow} greater than 4) or a value of 1.0 for COPCs with a log K_{ow} less than 4.

5.3.2.1 Empirical Correction Factor for Aboveground Produce (VG_{ag})

The parameter VG_{ag} has been incorporated into Equation 5-18 to address the potential overestimation for lipophilic COPCs to be transferred 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 and, 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, which is estimated for unspecified vegetables as follows:

$$VG_{rootveg} = \frac{M_{skin}}{M_{vegetable}} \qquad \text{Equation 5-19}$$

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)

If it is assumed 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. 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). With 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, Homberger, Neuner, Ranalder, Vetter, and Vuilleumier (1982), U.S. EPA (1994m) identified other processes—such as peeling, cooking, and cleaning—that will further reduce the vegetable concentration. U.S. EPA (1994m) recommended a $VG_{rootveg}$ value of 0.01 for lipophilic COPCs, which is less than the aforementioned estimates of 0.09 and 0.03 for the carrot and potato, but greater than the estimate would be if the correction factor was adjusted for cleaning, washing, and peeling, as described by Wipf, Homberg, Neuner, Ranalder, Vetter, and Vuilleumier (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."

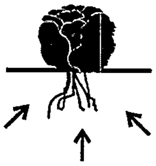
U.S. EPA (1994m) recommends a lipophilic COPC VG_{ag} of 1.0 for pasture grass because of a direct analogy to exposed azalea and grass leaves. Pasture grass is described as "leafy vegetation." However, the leafy vegetable group, as defined in Section 5.3.1.1, is composed of bulkier produce such as broccoli, 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) recommend a lipophilic COPC VG_{ag} value of 0.01, for all classes of aboveground produce.

U.S. EPA OSW recommends using a lipophilic COPC (log K_{ow} greater than 4) VG_{ag} value of 0.01 for all aboveground exposed produce. For COPCs with a log K_{ow} less than 4, U.S. EPA OSW recommends using a VG_{ag} value of 1.0, because these COPCs are assumed 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 the assumption of VG_{ag} values for leafy vegetables (such as lettuce) and for legumes (such as snap beans). Underestimation may be introduced by assuming a VG_{ag} value of 0.01 for legumes and leafy vegetables 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 also result in COPC concentrations in aboveground exposed produce, aboveground protected produce, and belowground produce. Consistent with previous guidance (U.S. EPA 1994m; U.S. EPA 1994r; and U.S. EPA 1995e), U.S. EPA OSW recommends the use of Equations 5-20A and 5-20B to calculate COPC concentration aboveground and belowground produce due to root uptake (*Pr*). The use of this equation is further described 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)
Br = Plant-soil bioconcentration factor for produce (unitless)
$VG_{rootveg}$ = Empirical correction factor for belowground produce (unitless)
Kd_s = Soil-water partition coefficient (L/kg)
Cs = Average soil concentration over exposure duration (mg COPC/kg soil)
RCF = Root concentration factor (unitless)

Equation 5-20A is based on the soil-to-aboveground plant transfer approach developed by Travis and Arms (1988). This approach is appropriate for evaluation of exposed and protected aboveground produce; however, it may not be appropriate for soil-to-belowground plant transfers. For belowground produce, U.S. EPA (1994m) and U.S. EPA (1995e) presented 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, the COPC soil concentration (Cs) must be divided by the COPC-specific soil-water partition coefficient (Kd_s) (U.S. EPA 1994m).

Appendix A-3 describes determination of compound specific parameters Br, Kd_s, and RCF. Appendix B and Section 5.2 describe calculation of Cs. Similar to VG_{ag} and as discussed in Section 5.3.2.1, $VG_{rootveg}$ is based on the lipophilicity of the COPC. Consistent with U.S. EPA (1994m), U.S. EPA OSW recommends a value of 0.01 for lipophilic COPCs (log K_{ow} greater than 4) based on root vegetables like carrots and potatoes, because it appears to be the most complete and thorough information available. For COPCs with a log K_{ow} less than 4, U.S. EPA OSW recommends 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 CALCULATION OF COPC CONCENTRATIONS IN BEEF AND DAIRY PRODUCTS

COPC concentrations in beef tissue and milk products are estimated on the basis of the amount of COPCs that cattle are assumed to consume through their diet. The cattle's diet is assumed to consist of:

(1) forage (primarily pasture grass and hay),

(2) silage (forage that has been stored and fermented), and

(3) grain.

Additional contamination may occur through the cattle's ingestion of soil. The total COPC concentration in the feed items (e.g., forage, silage, and grain) is calculated 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 and protected, depending on whether it has a protective outer covering. Because the outer covering on the protected feed acts as a barrier, it is assumed that there is negligible contamination of protected feed through deposition of particles and vapor transfer. In this analysis, grain is classified as protected feed. As a result, grain contamination is assumed to occur only through root uptake. Contamination of exposed feed items, including forage and silage, is assumed to occur through all three mechanisms.

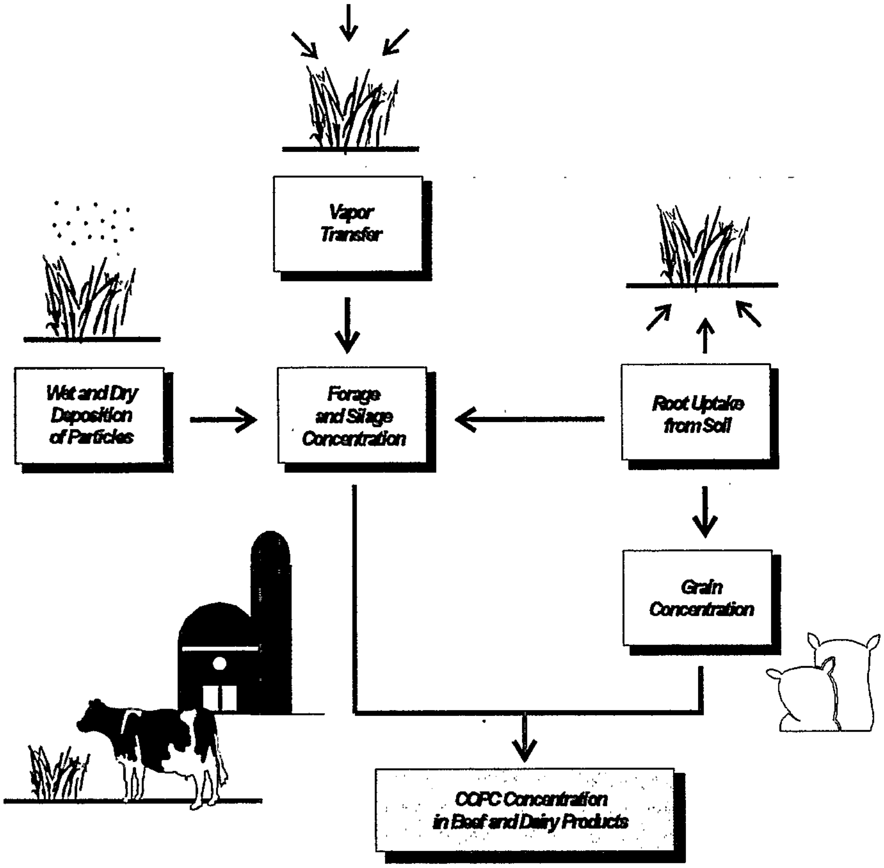


Figure 5-4 - COPC Concentration in Beef and Dairy Products

The amount of grain, silage, forage, and soil consumed is assumed to vary between dairy and beef cattle. Sections 5.4.4 (beef) and 5.4.5 (dairy) describe methods for estimating consumption rates and subsequent COPC concentrations in cattle. Consistent with previous guidance (U.S. EPA 1990e; U.S. EPA 1994a; NC DEHNR 1997), U.S. EPA OSW recommends that 100 percent of the plant materials eaten by cattle be assumed to have been grown on soil contaminated by emission sources. Therefore, 100 percent of the feed items consumed are assumed to be contaminated.

Appendix B, Tables B-3-1 through B-3-11, describe calculation of (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). Equation 5-14, described in Section 5.3.1, is recommended for calculation of COPC concentrations resulting from direct deposition onto plant surfaces of leafy plants and exposed produce (*Pd*). Therefore, U.S. EPA OSW recommends that Equation 5-14 also be used in calculating forage and silage concentrations due to direct deposition. Appendix B further describes calculation of COPC concentrations in forage and silage. Appendix A-3 describes determination of compound specific parameters *Fv*, *Bv*, and *Br*, which are calculated for forage and silage exactly as they are calculated for aboveground produce. *Rp*, *kp*, *Tp*, and *Yp* for use in calculating forage and silage concentrations are described in Sections 5.4.1.1 through 5.4.1.4.

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), U.S. EPA (1995b) and NC DEHNR (1997), U.S. EPA OSW recommends that Equation 5-14 be used to calculate *Rp* values for forage and silage.

Substituting the Baes, Sharp, Sjoreen, and Shor (1984) empirical constant (γ) 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 *Rp* is calculated to be 0.5. Substituting the Baes, Sharp, Sjoreen, and Shor (1984) empirical constant (γ) 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 *Rp* value is calculated to be 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:

(1) 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.

(2) The empirical constant for silage developed by Baes, Sharp, Sjoreen, and Shor (1984) for use in Chamberlain's empirical relationship may also fail to accurately represent site-specific silage types.

(3) The range of empirical constants recommended by Baes, Sharp, Sjoreen, and Shor (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, the use of 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*)

Equation 5-15 (Section 5.3.1.2) presents the calculation of the plant surface loss coefficient *kp* for aboveground produce. The *kp* factor is derived in exactly the same manner for cattle forage and silage, and the uncertainties of *kp* for cattle forage and silage are similar to its 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, *Tp* is treated as a constant, based on the average period between successive hay harvests. This period, which was estimated at 60 days by Belcher and Travis (1989), represents the length of time that aboveground vegetation (in this case, hay) would be exposed to particle deposition before being harvested. Using Equation 5-16 (Section 5.3.1.3), *Tp* is calculated to be 0.16 year for cattle silage.

For cattle forage, Equation 5-16 is modified 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 average period between hay harvests is assumed to be 60 days, and the average period between successive grazing is assumed to be 30 days. *Tp* is therefore calculated 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, and are 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 (1990e) stated that the best estimate of *Yp* is productivity, which is defined in Equation 5-17. This definition of *Yp* requires consideration of dry harvest yield (*Yh*) and area harvested (*Ah*).

U.S. EPA OSW recommends that forage *Yp* be calculated as a weighted average of the calculated pasture grass and hay *Yp* values. Weightings are assumed to be 0.75 for forage and 0.25 for hay, 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). An unweighted pasture grass *Yp* of 0.15 kg DW/m^2 is assumed (U.S. EPA 1994r; U.S. EPA 1994m). An unweighted hay *Yp* of 0.5 kg DW/m^2 is calculated by 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 are multiplied by their weighting factors (0.75 and 0.25, respectively), and then added to calculate the weighted forage *Yp* of 0.24 kg DW/m^2.

U.S. EPA OSW recommends that a production-weighted U.S. average *Yp* of 0.8 kg DW/m^2 be assumed for silage (Shor, Baes, and Sharp 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, the magnitude of the uncertainty introduced by the default *Yp* value can be estimated. 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) should be adjusted to reflect site-specific conditions, as appropriate.

5.4.2 Forage and Silage Concentrations Due to Air-to-Plant Transfer (*Pv*)

COPC concentration in aboveground produce resulting from air-to-plant transfer (*Pv*), is calculated by using Equation 5-18 (Section 5.3.2). *Pv* is calculated for cattle forage and silage similarly to the way that it is calculated for aboveground produce. A detailed discussion of *Pv* is provided in Section 5.3.2. Differences in VG_{ag} values for forage and silage, as compared to the values for aboveground produce described in Section 5.3.2.1, are presented below in Section 5.4.2.1. The calculation of *Pv* is further described in Appendix B.

5.4.2.1 Empirical Correction Factor for Forage and Silage (VG_{ag})

U.S. EPA (1994m) 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 (1994m) and NC DEHNR (1997) recommended a VG_{ag} to reduce estimated concentrations of COPCs in specified types of vegetation. Such a factor can be used to reduce estimated silage concentrations if it is assumed that there is insignificant translocation of COPCs deposited on the surface of bulky silage to the inner parts of the vegetation. Application of a silage VG_{ag} would be relevant if the silage cannot 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.

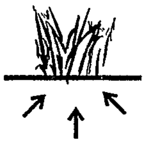
U.S. EPA (1994m) did not recommend a VG_{ag} value for silage. NC DEHNR (1997) recommended a VG_{ag} factor of 0.5 for bulky silage but does not present a specific rationale for this value. U.S. EPA (1995b) noted that a volume ratio of outer whole surface area to volume of vegetation could be used to assign a silage VG_{ag} value, if specific assumptions—concerning the proportions of each type of vegetation of which silage may consist—were known. However, in the absence of specific assumptions 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.

U.S. EPA OSW recommends the use of 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*)



COPC concentration in aboveground and belowground produce resulting from root uptake is calculated by using Equations 5-20A and 5-20B (Section 5.3.3). *Pr* is also calculated for cattle forage, silage, and grain in exactly the same way that it is calculated for aboveground produce. A detailed discussion describing calculation of *Pr* is provided in Section 5.3.3. The calculation of *Pr* is further described in Appendix B.

5.4.4 Beef Concentration Resulting from Plant and Soil Ingestion (A_{beef})

Consistent with U.S. EPA (1995h), U.S. EPA OSW recommends that COPC concentration in beef tissue (A_{beef}) be calculated by using Equation 5-22. 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 the introduction of 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). The use of this equation is further described in Appendix B, Table 3-10.

Recommended Equation for Calculating:
Concentration of COPC in Beef (A_{beef})

$$A_{beef} = \left(\sum (F_i \cdot Qp_i \cdot P_i) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{beef} \cdot MF \qquad \text{Equation 5-22}$$

---

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)

Appendix A-3 describes determination of the compound specific parameter Ba_{beef}. The parameters F_i, Qp_i, P_i, Qs, Cs, Bs, and MF are described in Sections 5.4.4.1 through 5.4.4.7, respectively.

5.4.4.1 Fraction of Plant Type *i* Grown on Contaminated Soil and Eaten by the Animal (Cattle)(F_i)

Consistent with U.S. EPA (1990e), U.S. EPA (1994r), and NC DEHNR (1997), U.S. EPA OSW recommends that 100 percent of the plant materials eaten by cattle be assumed to have been grown on soil contaminated by the emission sources being evaluated. U.S EPA OSW recommends 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 should be estimated (kg DW/day) for each category of plant feed. Forage, silage, and grain feeds should be included in this estimate (U.S. EPA 1990e; U.S. EPA 1994r; NC DEHNR 1997).

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 quantity of forage, grain, and silage eaten by cattle was recommended by NC DEHNR (1997), U.S. EPA (1994r), U.S. EPA (1990e), and Boone, Ng, and Palm (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, Ng, and Palm (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, Ng, and Palm (1981) NAS (1987)
U.S. EPA (1990e) Subsistence Farmer Beef Cattle	8.8	0.47	2.5	Boone, Ng, and Palm (1981) McKone and Ryan (1989)
Boone, Ng, and Palm (1981)	8.87	1.9	2.5	Boone, Ng, and Palm (1981)

With the exception of a higher grain ingestion rate, Boone, Ng, and Palm (1981) rates are consistent with those recommended by U.S. EPA (1990e), U.S. EPA (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.

U.S. EPA OSW recommends 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)

The total COPC concentration in forage, silage, and grain should be calculated by using Equation 5-23. Values for *Pd*, *Pv*, and *Pr* should be derived 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)

$$P_i = \sum_i (Pd + Pv + Pr)$$ 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. U.S. EPA OSW recommends a value of 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

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.

5.4.4.5 Average Soil Concentration Over Exposure Duration (*Cs*)

COPC concentration in soil should be calculated as discussed in Section 5.2.1, by using Equations 5-1C, 5-1D, and 5-1E. Also, Appendix B further describes calculation of the soil concentration.

5.4.4.6 Soil Bioavailability Factor (*Bs*)

Soil bioavailability factor, *Bs,* is defined 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* value would be equal to or greater than 1.0.

Until more COPC-specific data becomes available for this parameter, U.S. EPA OSW recommends 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*) represents the estimated amount of COPC that remains in fat and muscle. Based on a study by Ikeda et al. (1980), U.S. EPA (1995h) utilized a COPC-specific *MF* to account for metabolism in animals and humans. Consistent with U.S. EPA (1995h), U.S. EPA recommends a *MF* of 0.01 for bis(2-ethylhexyl)phthalate (BEHP), and 1.0 for all other COPCs. Evidence indicates BEHP is more readily metabolized and excreted by mammalian species than other contaminants (ATSDR 1987). Considering the recommended values for this variable, *MF* has no quantitative effect on A_{beef} (with the exception of BEHP).

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 in this guidance are intake driven, the use of a metabolism factor applies only to food sources used in evaluating indirect human exposure, including ingestion of beef, milk, and pork. In summary, use of a *MF* does not apply 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})

Equation 5-22 (Section 5.4.4) describes the calculation of COPC concentrations in beef cattle (A_{beef}). Equation 5-22 can be modified to calculate COPC milk concentrations (A_{milk}), as follows:

Recommended Equation for Calculating:
Concentration of COPC in Milk (A_{milk})

$$A_{milk} = \left(\sum (F_i \cdot Qp_i \cdot P_i) + Qs \cdot Cs \cdot Bs\right) \cdot Ba_{milk} \cdot MF \qquad \text{Equation 5-24}$$

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)

U.S. EPA OSW recommends the use of Equation 5-24 to estimate dairy cattle milk COPC concentration (A_{milk}). Appendix A-3 describes determination of the compound specific parameter Ba_{milk}. The use of this equation is further described in Appendix B, Table B-3-11.

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.

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. The daily quantity of feed consumed by cattle should be estimated on a dry weight basis for each category of plant feed.

NC DEHNR (1997) recommended the use of 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 U.S. EPA (1994r) identified plant ingestion rates only for subsistence farmers.

The following daily quantity of forage, grain, and silage eaten by dairy cattle was recommended by NC DEHNR (1997), U.S. EPA (1994r), U.S. EPA (1990e), and Boone, Ng, and Palm (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, Ng, and Palm (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, Ng, and Palm (1981) NAS (1987)

Source	Forage (kg/day DW)	Grain (kg/day DW)	Silage (kg/day DW)	References
U.S. EPA (1990e) Subsistence Dairy Farmer Cattle	11.0	2.6	3.3	Boone, Ng, and Palm (1981) McKone and Ryan (1989)
Boone, Ng, and Palm (1981)	11.0	2.6	3.3	Boone, Ng, and Palm (1981)

U.S. EPA (1990e) notes 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 (1993d).

Based on more recent references (NAS 1987; U.S. EPA 1993d) which recommend a feed ingestion rate of 20 kg/day DW, U.S. EPA OSW recommends 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 the estimation of Qp_i include the estimation of forage, grain, and silage ingestion rates, which will vary from site to site. The assumption of 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. U.S. EPA OSW recommends the following soil ingestion rates 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 subsistence farmer 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. It was assumed that the more conservative 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. The assumption of 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).

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 recommended for beef cattle (Section 7.4.5.7).

5.5 CALCULATION OF COPC CONCENTRATIONS IN PORK

COPC concentrations in pork tissue are estimated on the basis of the amount of COPCs that swine are assumed to consume through their diet; assumed to consist of silage and grain. Additional COPC contamination of pork tissue may occur through the ingestion of soil by swine.

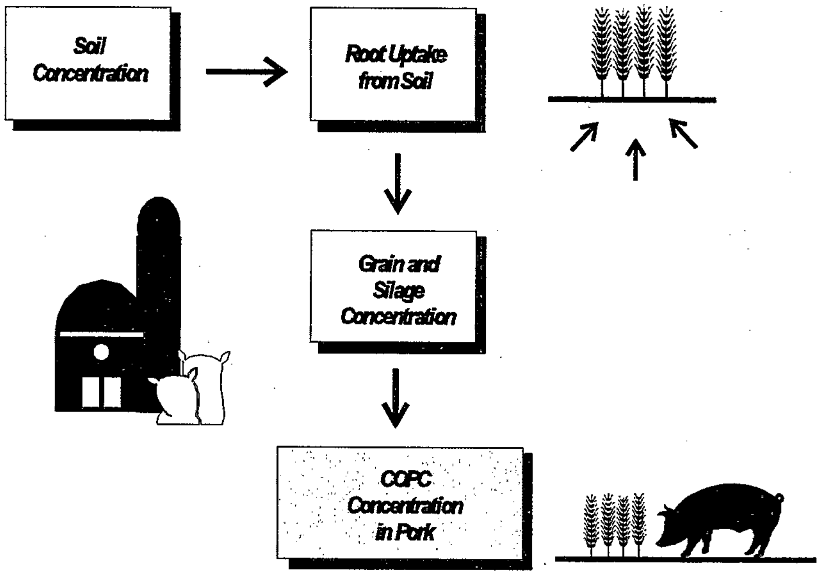


Figure 5-5 - COPC Concentration in Pork

5.5.1 Concentration of COPC In Pork

Equation 5-22 (Section 5.4.4) describes the calculation of COPC concentration in beef cattle (A_{beef}). Equation 5-22 can be modified to calculate COPC concentrations in swine (A_{pork}), as follows:

Recommended Equation for Calculating: Concentration of COPC in Pork (A_{pork})

$$A_{pork} = \left(\sum (F_i \cdot Qp_i \cdot P_i) + Qs \cdot Cs \cdot Bs \right) \cdot Ba_{pork} \cdot MF \qquad \text{Equation 5-25}$$

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)

U.S. EPA OSW recommends that Equation 5-25 be used to calculate COPC pork concentrations (A_{pork}). Appendix A-3 describes determination of the compound specific parameter Ba_{pork}. This equation is further described in Appendix B, Table B-3-12. The discussion in Section 5.4.5 of the variables F_i, Qp_i, P_i, Qs, Cs and MF for beef cattle generally applies to the corresponding variables for pork. However, different assumptions are made for pork. These differences are summarized in the following subsections.

5.5.1.1 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).

5.5.1.2 Quantity of Plant Type *i* Eaten by the Animal (Swine) Each Day (Qp_i)

As discussed in Section 5.4.4.2, the daily quantity of forage, silage, and grain feed consumed by beef cattle is estimated for each category of feed material. However, daily ingestion rates for pork are estimated differently than for beef cattle. U.S. EPA (1990e), U.S. EPA (1994r), and NC DEHNR (1997) recommended that only silage and grain feeds be included in estimates of daily plant quantity eaten by swine. Because swine are not grazing animals, they are assumed not to eat forage (U.S. EPA 1990e). The daily quantity of plant feeds (kilograms of DW) consumed by swine should be estimated for each category of plant feed.

U.S. EPA (1990e) and NC DEHNR (1997) did not differentiate between subsistence and typical hog farmers as for cattle. U.S. EPA (1990e) and NC DEHNR (1997) recommended grain and silage ingestion rates for swine as 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, Ng, and Palm (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, Ng, and Palm (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.

U.S. EPA OSW recommends the use of the more conservative ingestion rates. These rates are presented below:

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. Site-specific data can be used to mitigate this uncertainty. In addition, the assumption of uniform contamination of plant materials consumed by swine produces some uncertainty.

5.5.1.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Swine) (P_i)

The calculation of P_i for pork is identical to that for beef cattle (Section 5.4.4.3).

5.5.1.4 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 U.S. EPA (1990e) and NC DEHNR (1997):

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 (1993h) was cited as the reference for the soil ingestion rate of 8 percent of dry matter intake.

Consistent with NC DEHNR (1997), U.S. EPA OSW recommends 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 the assumption of uniform contamination of soil ingested by swine.

5.5.1.5 Average Soil Concentration Over Exposure Duration (*Cs*)

The calculation of *Cs* for pork is the same as for beef cattle (Section 5.4.4.5).

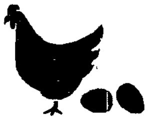
5.5.1.6 Soil Bioavailability Factor (*Bs*)

The calculation of *Bs* for pork is the same as for beef cattle (Section 5.4.4.6)

5.5.1.7 Metabolism Factor (*MF*)

The recommended values for MF are identical to those recommended for beef cattle (Section 5.4.4.7).

5.6 CALCULATION OF COPC CONCENTRATIONS IN CHICKEN AND EGGS



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. The uptake of COPCs via inhalation and via ingestion of water is assumed to be insignificant. Chickens are assumed to be free-range animals that have contact with soil; and therefore, are assumed to consume 10 percent of their diet as soil, a percentage that is consistent with the study from which the biotransfer factors were obtained (Stephens, Petreas, and Hayward 1995). The remainder of the diet (90 percent) is assumed to consist of grain. Grain ingested by chickens is assumed to have originated from the exposure scenario location; therefore, 100 percent of the grain consumed is assumed to be contaminated.

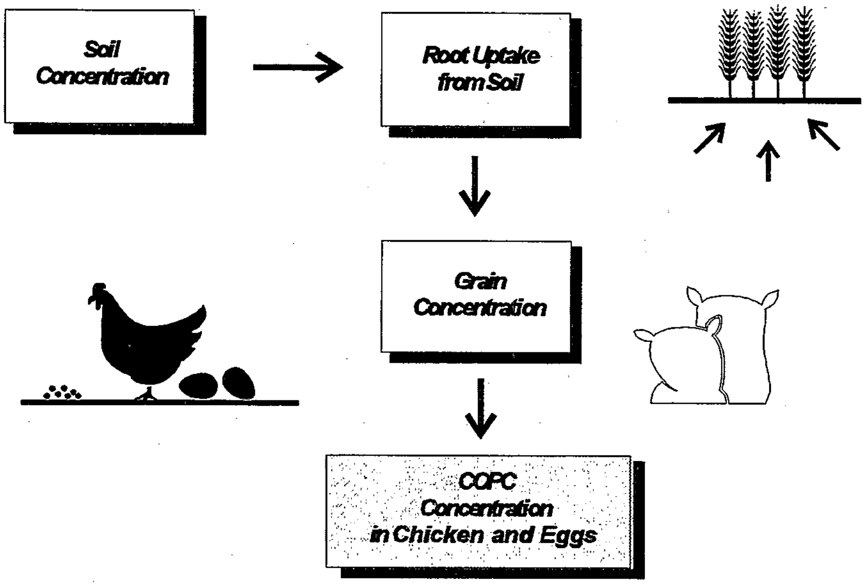


Figure 5-6 - COPC Concentration in Chicken and Eggs

The COPC concentration in grain is estimated by using the algorithm for aboveground produce described in Section 5.3. Grain is considered to be a feed item that is protected from deposition of particles and vapor transfer. As a result, only contamination due to root uptake of COPCs is considered in the calculation of COPC concentration in grain. Equations for calculating concentrations in chicken and eggs are presented in Appendix B. The methodology used to derive biotransfer factors and the COPC-specific values for chicken and eggs are presented in Appendix A-3.

5.6.1 Concentration of COPC in Chicken and Eggs

Consistent with NC DEHNR (1997), U.S. EPA OSW recommends the use of Equation 5-26 to calculate COPC concentrations in chicken and eggs (Stephens, Petreas, and Hayword 1995). COPC concentrations in chicken and eggs should be determined 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})

$$A_{chicken} \; or \; A_{egg} = (\Sigma[F_i \cdot Qp_i \cdot Pi] + Qs \cdot Cs \cdot Bs) \cdot (Ba_{egg} \; or \; Ba_{chicken}) \qquad \text{Equation 5-26}$$

---

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-3 describes determination of compound specific parameters $Ba_{chicken}$ and Ba_{egg}. The remaining parameters are discussed in Appendix B and in the following subsections.

5.6.1.1 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).

5.6.1.2 Quantity of Plant Type *i* Eaten by the Animal (Chicken) Each Day (Qp_i)

As discussed in Section 5.4.4.2, the daily quantity of forage, silage, and grain feed consumed by beef cattle is estimated for each category of feed material. 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 1990e). Chickens are assumed not to consume any silage. The daily quantity of plant feeds (kilograms of DW) consumed by chicken only should be estimated for grain feed.

Consistent with Ensminger (1980), Fries (1982), and NAS (1987), U.S. EPA OSW recommends the use of 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, the assumption of uniform contamination of plant materials consumed by chicken produces some uncertainty.

5.6.1.3 Concentration of COPC in Plant Type *i* Eaten by the Animal (Chicken) (P_i)

The total COPC concentration is the COPC concentration in grain and should be calculated by using Equation 5-27. Values for *Pr* should be derived by using Equation 5-20.

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)

This equation is further described in Appendix B.

5.6.1.4 Quantity of Soil Eaten by the Animal (Chicken) Each Day (*Qs*)

COPC concentration in chickens also results from intake of soil. As discussed earlier, chickens are assumed to consume 10 percent of their total diet as soil, a percentage that is consistent with the study from Stephens, Petreas, and Hayward (1995). U.S. EPA OSW recommends 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 the assumption of uniform contamination of soil ingested by chicken.

5.6.1.5 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).

5.6.1.6 Soil Bioavailability Factor (*Bs*)

The calculation of *Bs* for chicken is the same as for beef cattle (Section 5.4.4.6)

5.7 CALCULATION OF COPC CONCENTRATIONS IN DRINKING WATER AND FISH

COPC concentrations in surface water are calculated for all water bodies selected for evaluation in the risk assessment; specifically, evaluation of the drinking water and/or fish ingestion exposure pathways. Mechanisms considered for determination of COPC loading of the water column are:

(1) Direct deposition,

(2) Runoff from impervious surfaces within the watershed,

(3) Runoff from pervious surfaces within the watershed,

(4) Soil erosion over the total watershed,

(5) Direct diffusion of vapor phase COPCs into the surface water, and

(6) Internal transformation of compounds chemically or biologically.

Other potential mechanisms may require consideration on a case-by-case basis (e.g., tidal influences), however, contributions from other potential mechanisms are assumed to be negligible in comparison with those being evaluated.

The USLE and a sediment delivery ratio are used to estimate the rate of soil erosion from the watershed. Surface water concentration algorithms 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, the assumptions are made 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. Partitioning between water and sediment varies with the COPC. The total concentration of each COPC is partitioned between the sediment and the water column.

The equations used to estimate surface water concentrations are presented in Appendix B-4. To evaluate the COPC loading to a water body from its associated watershed, the COPC concentration in watershed soils should be calculated. As described in Section 5.2, the equation for 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 equations for calculating COPC concentrations in watershed soils, and COPC concentrations in the water body.

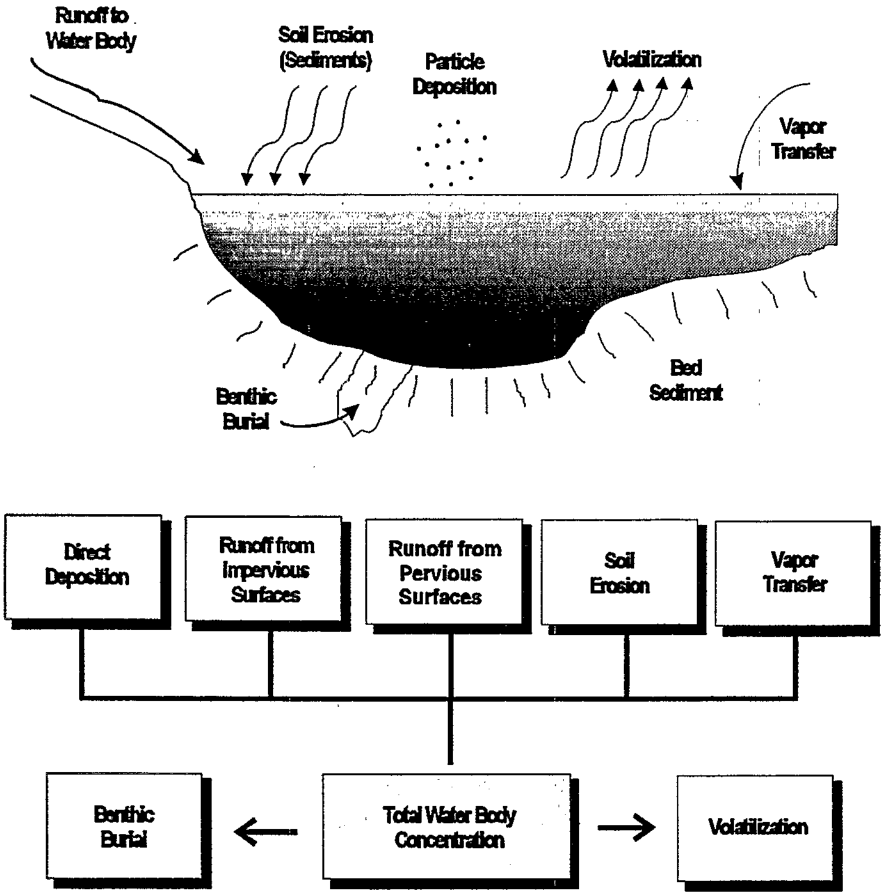


Figure 5-7 - COPC Loading to the Water Body

5.7.1 Total COPC Load to the Water Body (L_T)

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of 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 wet vapor phase COPC direct deposition load to water body (g/yr)
L_{dif} = Vapor phase COPC diffusion (dry deposition) 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 daughter products, U.S. EPA OSW recommends a default value for this variable of zero. However, if a permitting authority determines that site-specific conditions indicate calculation of internal transfer should be considered, U.S. EPA OSW recommends following the methodologies described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press). Calculation of each of the remaining variables (L_{DEP}, L_{dif}, L_{RI}, L_R, and L_E) is discussed in the following subsections.

5.7.1.1 Total (Wet and Dry) Particle Phase and Wet Vapor Phase COPC Direct Deposition Load to Water Body (L_{DEP})

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends Equation 5-29 to calculate the load to the water body from the direct deposition of wet and dry particles and wet 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 Wet Vapor Phase Direct Deposition Load to Water Body (L_{DEP})

$$L_{DEP} = Q \cdot [\ F_v \cdot Dywwv + (1 - F_v\ ) \cdot Dytwp\ ] \cdot A_W \qquad \text{Equation 5-29}$$

---

where

L_{DEP} = Total (wet and dry) particle phase and wet 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)
$Dywwv$ = Unitized yearly (water body and watershed) average wet deposition from vapor phase (s/m^2-yr)
$Dytwp$ = Unitized yearly (water body and watershed) average total (wet and dry) deposition from vapor phase (s/m^2-yr)
A_W = Water body surface area (m^2)

Chapter 3 describes the determination of the modeled air parameters, *Dywwv* and *Dywwv*. The determination of water body surface area, A_w, is described in Chapter 4 and Appendix B. Appendix A-3 describes determination of the compound-specific parameters, F_v.

5.7.1.2 Vapor Phase COPC Diffusion (Dry Deposition) Load to Water Body (L_{dif})

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends using Equation 5-30 to calculate the dry vapor phase COPC diffusion load to the water body (L_{dif}). The equation is described in detail in Appendix B, Table B-4-12.

Recommended Equation for Calculating:
Vapor Phase COPC Diffusion (Dry Deposition) Load to Water Body (L_{Dif})

$$L_{dif} = \frac{K_v \cdot Q \cdot F_v \cdot Cywv \cdot A_W \cdot 1 \times 10^{-6}}{\dfrac{H}{R \cdot T_{wk}}} \qquad \text{Equation 5-30}$$

where

L_{dif} = Vapor phase COPC diffusion (dry deposition) 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 and 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)

The overall COPC transfer rate coefficient (K_v) is calculated by using Equation 5-40. The equation is also presented in Appendix B, Table B-4-19. Consistent with U.S. EPA (1994r) and U.S. EPA (1993h), U.S. EPA OSW recommends a water body temperature (T_{wk}) of 298 K (or 25°C). Chapter 3 describes the determination of the modeled air parameter, *Cywv*. The determination of water body surface area, A_w, is described in Chapter 4 and Appendix B. Appendix A-3 describes determination of compound-specific parameters, F_v, H, and R.

5.7.1.3 Runoff Load from Impervious Surfaces (L_{RI})

In some watershed soils, a fraction of the wet and dry deposition in the watershed will be to impervious surfaces. Dry deposition may accumulate and be washed off during rain events. Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of 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})

$$L_{RI} = Q \cdot [\ F_v \cdot Dywwv + (1.0 - F_v\ ) \cdot Dytwp\ ] \cdot A_I \qquad \text{Equation 5-31}$$

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)
$Dywwv$ = Unitized yearly (water body and watershed) average wet deposition from vapor phase (s/m^2-yr)
$Dytwp$ = Unitized yearly (water body and watershed) average total (wet and dry) deposition from vapor phase (s/m^2-yr)
A_I = Impervious watershed area receiving COPC deposition (m^2)

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.

5.7.1.4 Runoff Load from Pervious Surfaces (L_R)

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of 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)
θ_{sw} = Soil volumetric water content (mL water/cm^3 soil)
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 the determination of site-specific parameters, RO, A_L, A_I, BD, and θ_{sw}. The calculation of the COPC concentration in watershed soils (Cs) are discussed in Section 5.2.1 and Appendix B, Table B-4-1. Soil bulk density (BD) is described in Section 5.2.5.2. Soil water content (θ_{sw}) is described in Section 5.2.5.4.

5.7.1.5 Soil Erosion Load (L_E)

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of 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)
θ_{sw} = Soil volumetric water content (mL water/cm^3 soil)
Kd_s = Soil-water partition coefficient (mL water/g soil)
0.001 = Units conversion factor (k-cm^2/mg-m^2)

Unit soil loss (X_e) is described in Section 5.7.2. Watershed sediment delivery ratio (*SD*) is calculated 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 5.2.5.2. Soil water content (θ_{sw}) is described in Section 5.2.5.4.

5.7.2 Universal Soil Loss Equation - USLE

U.S. EPA OSW recommends that the universal soil loss equation (USLE), Equation 5-33A, be used 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 determination of the site- and watershed-specific values for each of the variables associated with Equation 5-33A. The use of Equation 5-33A is consistent with U.S. EPA (1994g) and U.S. EPA (1994r).

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*)

U.S. EPA OSW recommends the use of Equation 5-34 to calculate 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, 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 (1993h) recommended the use of Equation 5-34 to calculate the sediment delivery ratio.

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, the following may be a consideration:

(1) the distance from the emission source,

(2) the location of the area affected by deposition fallout with respect to the point at which drinking water is extracted or fishing occurs

(3) the watershed hydrology.

5.7.4 Total Water Body COPC Concentration (C_{wtot})

U.S. EPA OSW recommends the use of Equation 5-35 to calculate total water body COPC concentration (C_{wtot}). The total water body concentration 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. 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 (1993h) recommended values ranging from 0.01 to 0.05. Consistent with U.S. EPA (1994r), U.S. EPA OSW recommends 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.

5.7.4.1 Fraction of Total Water Body COPC Concentration in the Water Column (f_{wc}) and Benthic Sediment (f_{bs})

U.S. EPA OSW recommends using Equation 5-36A to calculate fraction of total water body COPC concentration in the water column (f_{wc}), and Equation 5-36B to calculate 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)

U.S. EPA (1993h) and NC DEHNR (1997) recommended the use of Equations 5-36A and 5-36B to calculate the fraction of total water body concentration occurring in the water column (f_{wc}) and the bed sediments (f_{bs}). The 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-3). 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 (1993g). However, due to variability in water body specific values for this variable, U.S. EPA OSW recommends the use of water body-specific measured *TSS* values representative of long-term average annual values for the water body of concern. Average annual values for *TSS* are generally expected to be in the range of 2 to 300 mg/L; with additional information on anticipated *TSS* values available in the U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press).

If measured data is not available, or of unacceptable quality, a calculated *TSS* value can be obtained 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, 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, the water-body specific measured *TSS* value should be compared to the calculated *TSS* value obtained using Equation 5-36C. If the measured and calculated *TSS* values differ significantly, parameter values used in calculating X_e should be re-evaluated. This re-evaluation of *TSS* and X_e should also be conducted if the calculated *TSS* value is outside of the normal range expected for average annual measured values, as discussed above.

Bed sediment porosity (θ_{bs}) can be calculated from the bed sediment concentration by using the following equation (U.S. EPA 1993h):

$$\theta_{bs} = 1 - \frac{C_{BS}}{\rho_s} \qquad \text{Equation 5-37}$$

where

θ_{bs} = Bed sediment porosity ($L_{water}/L_{sediment}$)
ρ_s = Bed sediment density (kg/L)
C_{BS} = Bed sediment concentration (kg/L)

U.S. EPA OSW recommends the following default value for bed sediment porosity (θ_{bs}), which was adapted from U.S. EPA (1993h) and NC DEHNR (1997):

Recommended Value for:
Bed Sediment Porosity (θ_{bs})

$$\theta_{bs} = 0.6 \ L_{water}/L_{sediment}$$

(assuming $\rho_s = 2.65$ kg/L [bed sediment density] and $C_{BS} = 1$ kg/L [bed sediment concentration])

Concentrations for the bed sediment (C_{BS}) and depth of upper benthic sediment layer (d_{bs}) range from 0.5 to 1.5 meters and 0.01 to 0.05 meters, respectively. However, in accordance with U.S. EPA (1993h), U.S. EPA (1994r) and NC DEHNR (1997), 0.1 kg/L is a reasonable concentration for most applications of the bed sediment (C_{BS}), and 0.03 meter is the default depth of the upper benthic layer (d_{bs}). The default depth of 0.03 meters is based on the midpoint of the range presented above.

5.7.4.2 Overall Total Water Body COPC Dissipation Rate Constant (k_{wt})

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of 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, Equations 5-36A and 5-36B, and calculated by using the equations presented in Appendix B, Table B-4-16.

5.7.4.3 Water Column Volatilization Rate Constant (k_v)

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends using Equation 5-39 to calculate water column volatilization rate constant. The equation is also presented in Appendix B, Table B-4-18.

Recommended Equation for Calculating:
Water Column Volatilization Rate Constant (k_v)

$$k_v = \frac{K_v}{d_z \cdot (1 + Kd_{sw} \cdot TSS \cdot 1 \times 10^{-6})} \qquad \text{Equation 5-39}$$

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)

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-3. The overall transfer rate coefficient (K_v) is described in Section 5.7.4.4.

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.

Consistent with U.S. EPA (1993h), U.S. EPA (1993g), and NC DEHNR (1997), U.S. EPA OSW recommends the use of Equation 5-40 to calculate the overall transfer rate coefficient (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 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.

The liquid and gas phase transfer coefficients, K_L and K_G, respectively, vary with the type of water body. The liquid phase transfer coefficient (K_L) is calculated by using Equations 5-41A and 5-41B (described in Section 5.7.4.5). The gas phase transfer coefficient (K_G) is calculated by using Equations 5-42A and 5-42B (described 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 compound-specific and are presented in Appendix A-3. 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, is used to adjust for the actual water temperature. Volatilization is assumed to occur much less readily in lakes and reservoirs than in moving water bodies.

5.7.4.5 Liquid Phase Transfer Coefficient (K_L)

U.S. EPA OSW recommends 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:

$$K_L = \sqrt{\frac{(1 \times 10^{-4}) \cdot D_w \cdot u}{d_z}} \cdot 3.1536 \times 10^7$$

Equation 5-41A

For quiescent lakes or ponds:

$$K_L = (C_d^{0.5} \cdot W) \cdot (\frac{\rho_a}{\rho_w})^{0.5} \cdot \frac{k^{0.33}}{\lambda_z} \cdot (\frac{\mu_w}{\rho_w \cdot D_w})^{-0.67} \cdot 3.1536 \times 10^7$$

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)
$_z$ = 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)

For a flowing stream or river, the transfer coefficients are controlled by flow-induced turbulence. For these systems, the liquid phase transfer coefficient is calculated by using Equation 5-41A, which is the O'Connor and Dobbins (1958) formula, as presented in U.S. EPA (1993h).

For a stagnant system (quiescent lake or pond), the transfer coefficient is controlled by wind-induced turbulence. For quiescent lakes or ponds, the liquid phase transfer coefficient can be calculated by using Equation 5-41B (O'Connor 1983; U.S. EPA 1993h).

The total water body depth (d_z) for liquid phase transfer coefficients is discussed in Section 5.7.4.1.

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends the use of the following default values. These values are further described in Appendix A-3:

(1) a diffusivity of chemical in water ranging (D_w) from 1.0×10^{-5} to 8.5×10^{-2} cm^2/s,

(2) a dimensionless viscous sublayer thickness ($_z$) of 4,

(3) a von Karman's constant (k) of 0.4,

(4) a drag coefficient (C_d) of 0.0011 which was adapted from U.S. EPA (1993h),

(5) 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),

(6) a density of water (ρ_w) of 1 g/cm^3 (Weast 1986),

(7) a viscosity of water (μ_w) of a 0.0169 g/cm-s corresponding to water temperature (Weast 1986).

5.7.4.6 Gas Phase Transfer Coefficient (K_G)

U.S. EPA OSW recommends 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:

$$K_G = 36500 \text{ m/yr} \qquad \text{Equation 5-42A}$$

For quiescent lakes or ponds:

$$K_G = (C_d^{0.5} \cdot W) \cdot \frac{k^{0.33}}{\lambda_z} \cdot \left(\frac{\mu_a}{\rho_a \cdot D_a}\right)^{-0.67} \cdot 3.1536 \times 10^7 \qquad \text{Equation 5-42B}$$

---

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)
$_z$ = 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)

U.S. EPA (1993h) 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, U.S. EPA OSW recommends that the gas phase transfer coefficient be computed by using the equation presented in O'Connor (1983) (Equation 5-42B).

Consistent with U.S. EPA (1994r) and NC DEHNR (1997), U.S. EPA OSW recommends 1.81×10^{-4} g/cm-s for the viscosity of air corresponding to air temperature.

5.7.4.7 Benthic Burial Rate Constant (k_b)

U.S. EPA OSW recommends 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)
1×10^{-6} = Units conversion factor (kg/mg)
1×10^3 = Units conversion factor (g/kg)

The benthic burial rate constant (k_b), which is calculated in Equation 5-43, can also be expressed in terms of the rate of burial (Wb):

$$Wb = k_b \cdot d_{bs}$$ 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 by using Equation 5-43. 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 (1993g). U.S. EPA OSW recommends the following default value for bed sediment concentration (C_{BS}).

Recommended Default Value for: **Bed Sediment Concentration (C_{BS})**
1.0 kg/L

Section 5.7.2 discusses the unit soil loss (X_e). Section 5.7.3 discusses sediment delivery ratio (SD) and watershed area evaluated receiving COPC deposition (A_L). Section 5.7.4 discusses the depth of the upper benthic sediment layer (d_{bs}). 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-3. Section 5.7.4.1 discusses total suspended solids concentration (TSS).

The calculated value for k_b should range from 0 to 1.0; with low k_b values expected for water bodies characteristic of no or limited sedimentation (rivers and fast flowing streams), and k_b values closer to 1.0 expected 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. For k_b values calculated as a negative (water bodies with high average annual volumetric flow rates in comparison to watershed area evaluated), a k_b value of 0 should be assigned for use in calculating the total water body COPC concentration (C_{wtot}) in Equation 5-35. If the calculated k_b value exceeds 1.0, re-evaluation of the parameter values used in calculating X_e should be conducted.

5.7.4.8 Total COPC Concentration in Water Column (C_{wctot})

U.S. EPA OSW recommends 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)

The use of Equation 5-45 to calculate total COPC concentration in water column is consistent with U.S. EPA (1994r) and NC DEHNR (1997).

Total water body COPC concentration—including water column and bed sediment (C_{wtot}) and fraction of total water body COPC concentration in the water column (f_{wc})—should be calculated by using Equation 5-35 (also see Appendix B, Table B-4-15) and Equation 5-36A (also see Appendix B, Table B-4-16), respectively. Depth of upper benthic sediment layer (d_{bs}) is discussed in Section 5.7.4.1.

5.7.4.9 Dissolved Phase Water Concentration (C_{dw})

U.S. EPA OSW recommends the use of 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)

The use of Equation 5-46 to calculate the concentration of COPC dissolved in the water column is consistent with U.S. EPA (1994r) and NC DEHNR (1997).

The total COPC concentration in water column (C_{wctot}) is calculated by using the Equation 5-45 (see also Appendix B, Table B-4-23). Section 5.7.4.1 discusses the surface water partition coefficient (Kd_{sw}) and total suspended solids concentration (TSS).

5.7.4.10 COPC Concentration Sorbed to Bed Sediment (C_{sb})

U.S. EPA OSW recommends the use of 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)

The use of Equation 5-47 to calculate the COPC concentration sorbed to bed sediment is consistent with U.S. EPA (1994r) and NC DEHNR (1997).

The total water body COPC concentration—including water column and bed sediment (C_{wtot}) and the fraction of total water body COPC concentration that occurs in the benthic sediment (f_{bs})—is calculated by using the equations in Appendix B, Tables B-4-15 and B-4-16, respectively. Bed sediment and sediment pore water partition coefficient (Kd_{bs}) is discussed in Appendix A-3. Bed sediment porosity (θ_{bs}) and bed sediment concentration (C_{BS}) are discussed in Section 5.7.4.1. Depth of water column (d_{wc}) and depth of upper benthic layer (d_{bs}) are discussed in Section 5.7.4.

5.7.5 Concentration of COPC in Fish (C_{fish})

The COPC concentration in fish is calculated using either a COPC-specific bioconcentration factor (*BCF*), a COPC-specific bioaccumulation factor (*BAF*), or a COPC-specific biota-sediment accumulation factor (*BSAF*). For compounds with a $\log K_{ow}$ less than 4.0, *BCF*s are used. Compounds with a $\log K_{ow}$ greater than 4.0 (except for extremely hydrophobic compounds such as dioxins, furans, and PCBs), are assumed to have a high tendency to bioaccumulate, therefore, *BAF*s are used. While extremely hydrophobic COPCs like dioxins, furans, and PCBs are also assumed to have a high tendency to bioaccumulate, they are expected to be sorbed to the bed sediments more than associated with the water phase. Therefore, for dioxins, furans, and PCBs, *BSAF*s were used to calculate concentrations in fish. Appendix A-3, provides a detailed discussion on the sources of the COPC-specific *BCF*, *BAF*, and *BSAF* values, and the methodology used to derive them.

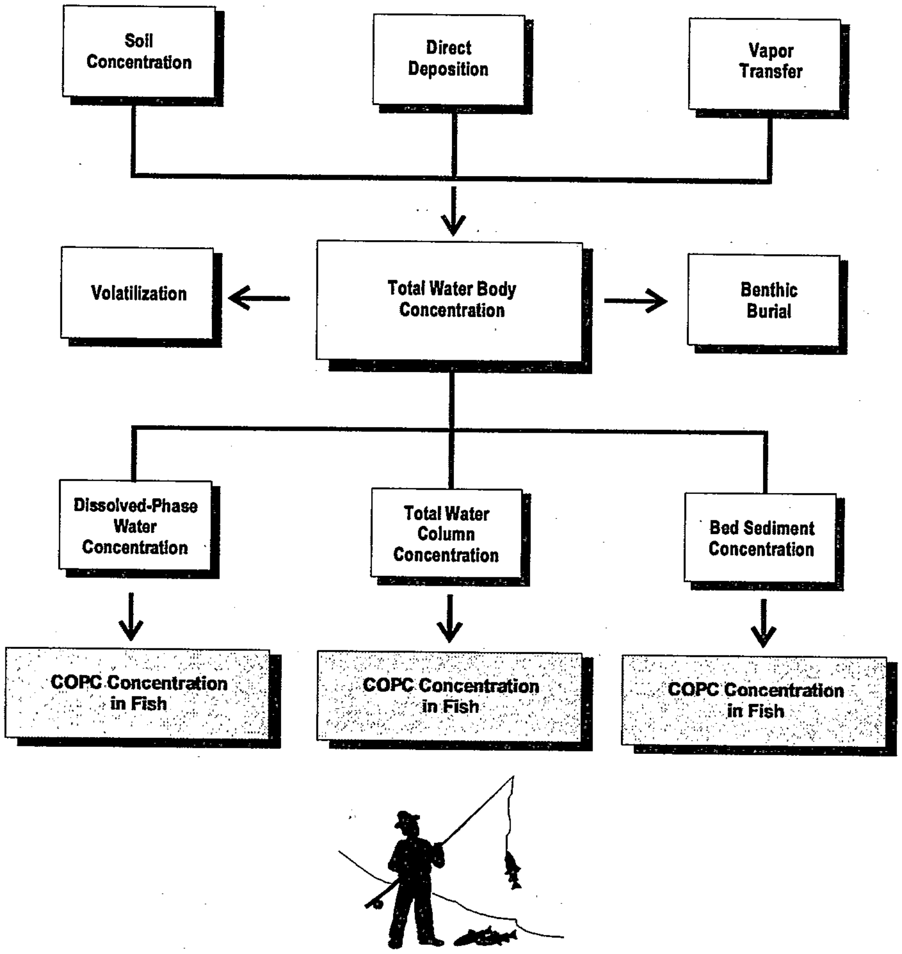


Figure 5-8 - COPC Concentration in Fish

BCF and *BAF* values are generally based on dissolved water concentrations. Therefore, when *BCF* or *BAF* values are used, the COPC concentration in fish is calculated using dissolved water concentrations. *BSAF* values are based on benthic sediment concentrations. Therefore, when *BSAF* values are used, COPC concentration in fish is calculated using benthic sediment concentrations. The equations used to calculate fish concentrations are described in the subsequent subsections.

5.7.5.1 Fish Concentration (C_{fish}) from Bioconcentration Factors Using Dissolved Phase Water Concentration

U.S. EPA OSW recommends the use of Equation 5-48 to calculate fish concentration from *BCF*s using dissolved phase water concentration. The use of 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)

The dissolved phase water concentration (C_{dw}) is calculated by using the Equation 5-46. COPC-specific BCF_{fish} values are presented in Appendix A-3.

The use of Equation 5-48 to calculate fish concentration is consistent with U.S. EPA (1994r) and NC DEHNR (1997).

5.7.5.2 Fish Concentration (C_{fish}) from Bioaccumulation Factors Using Dissolved Phase Water Concentration

U.S. EPA OSW recommends the use of 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)

The dissolved phase water concentration (C_{dw}) is calculated by using Equation 5-46. COPC-specific bioaccumulation factor (BAF_{fish}) values are presented in Appendix A-3.

5.7.5.3 Fish Concentration (C_{fish}) from Biota-To-Sediment Accumulation Factors Using COPC Sorbed to Bed Sediment

U.S. EPA OSW recommends the use of Equation 5-50 to calculate fish concentration from *BSAF*s using COPC sorbed to bed sediment 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)

The concentration of COPC sorbed to bed sediment (C_{sb}) is calculated by using Equation 5-47. U.S. EPA OSW recommended default values for the fish lipid content (f_{lipid}) and for the fraction of organic carbon in bottom sediment (OC_{sed}) are given in Appendix B, Table B-4-28. Biota-to-sediment accumulation factors (*BSAF*), which are applied only to dioxins, furans, and PCBs, are presented in Appendix A-3.

The use of Equation 5-50 to calculate fish concentration from bed sediment is consistent with U.S. EPA (1994r) and NC DEHNR (1997). Values recommended by U.S. EPA (1993h) range from 0.03 to 0.05 for the fraction of organic carbon in bottom sediment (Oc_{sed}). These values are based on an assumption of a surface soil OC content of 0.01. This document 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.

U.S. EPA (1993g) recommended a default value of 0.04 for OC_{sed}, which is the midpoint of the specified range. U.S. EPA (1993h; 1993g) recommended the use of 0.07 as the fish lipid content (f_{lipid}). This value was originally cited in Cook, Duehl, Walker, and Peterson (1991).

5.8 Use of Site-Specific vs. Default Parameter Values

As discussed in Chapter 1, most of the input parameters recommended for use in this guidance are not site-specific. After completing a risk assessment based on the default parameter values recommended in this guidance, risk assessors may choose to investigate the use of site-specific parameter values in order to provide a more representative estimate of site-specific risk. Use of parameter values other than those specified in this guidance should always be clearly described in the risk assessment report and work plan, and approved by the permitting authority. U.S. EPA OSW recommends that requests to change default parameter values include the following information, as appropriate:

1. An explanation of why the use of 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. The basis of the default parameter value, as understood by the requestor, including readable copies of the referenced literature or studies (if available);

4. 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);

5. A description of other risk assessments or projects where the site-specific parameter value has been used, and how such risk assessments or projects are similar to the risk assessment in consideration.

Chapter 6
Quantifying Exposure

What's Covered in Chapter 6:

- Generic Exposure Rate Equation
- Consumption Rate
- Exposure Frequency
- Exposure Duration
- Averaging Time
- Body Weight

This chapter describes the factors to be evaluated in quantifying the exposure received under each of the recommended exposure scenarios described in Chapter 4. The calculation of COPC-specific exposure rates for each exposure pathway evaluated involves (1) the estimated COPC media concentrations calculated in Chapter 5, (2) consumption rate, (3) receptor body weight, and (4) the frequency and duration of exposure. This calculation is repeated for each COPC and for each exposure pathway included in an exposure scenario. Exposure pathway-specific equations are presented 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 also evaluated as a separate issue in Section 7.5.

6.1 GENERIC EXPOSURE RATE EQUATION

Exposure can occur over a period of time. In the calculation of an average exposure per unit of time, the total exposure can be divided by the time period. An average exposure can be expressed in terms of body weight. All exposures quantified in the risk assessment (1) should be unitized for time and body weight, (2) are presented in units of milligrams per kilogram of body weight per day, and (3) are termed "intakes." Equation 6-1 is a generic equation used to calculate 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 media 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 and 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.

Variations of Equation 6-1 are used to calculate receptor-specific exposures to COPCs; the equations used for each exposure pathway are presented in Appendix C. The variation of input variables when exposure is quantified is also described in Appendix C.

The exposures calculated in a risk assessment are intended to represent reasonable maximum exposure (RME) conditions as further described in U.S. EPA (1989e). The use of RME values is consistent with other U.S. EPA guidance (1994g). 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 the remaining variables are set at less conservative, typical or "central tendency" values (e.g., near the 50th percentile) resulted in a product of 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 depositional rates are based on RME emissions from trial or risk burns. U.S. EPA OSW recommends that the variables set at RME values include (1) the highest ISCST3 modeled air parameter values at current and reasonable potential future exposure scenario locations, (2) the exposure frequency, and (3) the exposure duration. Body weight is typically set at average values.

6.2 CONSUMPTION RATE

Consumption rate is the amount of contaminated medium consumed per unit of time or event. Consumption rates for subsistence food types (e.g., beef for the subsistence farmer; fish for the subsistence fisher), is assumed to be 100 percent from the assessment area (e.g., farm, water body) being evaluated. Consumption rates for non-subsistence food types (e.g., home grown garden vegetables) are assumed to be a fraction of the total dietary intake for this food type.

As described in Section 6.1, exposures calculated in a risk assessment are intended to represent RME conditions. Accordingly, the HHRAP recommends default values for exposure parameters that will result in estimated RME exposures. 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 parameter consumption rate (a general term including both intake rate and inhalation rate). The risk assessment performed using recommended default parameter values may be latter refined to include supplemental calculations based on regional- or site-specific exposure parameter values, provided documentation for these regional- or site-specific exposure parameter values is provided. These supplemental calculations should be provided in addition to and should not replace calculations based on recommended default exposure parameter values. The following subsections describe exposure pathway-specific considerations regarding consumption rate.

6.2.1 Air Exposure Pathways

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 combustion emissions. Chapter 3 presented the air dispersion and deposition modeling techniques used to estimate airborne concentrations of vapors and particulates in the assessment area.

Receptors in the assessment area are directly exposed to COPCs in vapor, particulate, and particle-bound phases; as a result of normal respiration. The factors that affect exposure from vapor and particulate inhalation include vapor and particulate COPC concentrations, respiration rate during the period of exposure, and length of exposure.

As presented in Appendix C, a single default inhalation rate is provided for use across all adult receptor scenarios. However, if site-specific data is available to show that subsistence farmers and fishers have higher respiration rates due to rigorous physical activities than other receptors that data may be appropriate. Also, farmers could be assumed to typically spend more time each day in the vicinity of contaminated vapors and particulates, because farms are places of business, and typically their homes. However, any modifications of the respiration rates of receptors should be considered on a site-specific basis, and supported by documentation.

Intakes related to direct inhalation of vapors and particulates are calculated based on variations of Equation 6-1. However, as described in Chapter 7, Appendix A-3, and Appendix C, noncarcinogenic hazards and carcinogenic risks associated with direct inhalation exposures are preferentially characterized using toxicity factors (inhalation unit risk factors [URF] and reference concentrations [RfC]) based on data collected under, or normalized to, a particular set of respiratory and body weight parameters (e.g., 20 cubic meters per day [m^3/day] and 70 kg).

Inhalation of vapors and particulates will 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, it should be assumed that vapor and particulates may both be inhaled throughout the day, both indoors and outdoors.

6.2.2 Food Exposure Pathways

Plants and animals impacted by emission sources may take up emitted COPCs in the air or deposited COPCs in soil. Humans are exposed to COPCs via the food chain when they consume these plants and animals as a food source. Human intake of COPCs is determined on the basis of (1) the types of foods consumed, (2) the amount of food consumed per day, (3) the concentration of COPCs in the food, and (4) the percentage of the diet contaminated by COPCs. Chapter 5 describes procedures for determining the concentration of COPCs in food; and consideration of variations in exposure resulting from food preparation methods and type of food item (e.g., protected versus unprotected produce). Other variables, described below, may also significantly affect the estimation of exposure.

6.2.2.1 Types of Foods Consumed

The types of foods consumed will affect exposure, because different plants and animal tissues will take up COPCs at different rates. Therefore, COPC concentrations in food are determined, in part, by the type of food, and they vary with the types of food in the diet. Furthermore, the types of foods consumed vary with age, geographical region, and sociocultural factors.

6.2.2.2 Food Consumption Rate

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 (1990e) recommended that values from USDA food consumption surveys be used to complete the risk assessment process. U.S. EPA (1990e) recommended that the 1987-1988 USDA Food Consumption Survey be used 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 that the 1966-67 USDA Food Consumption Survey be used to represent the consumption rates of a more agrarian population.

U.S. EPA OSW recommends that food consumption rate information (ingestion rates) be obtained from the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c); specifically, the section regarding home produced food items. Consumption rate information is presented in Appendix C as follows: Appendix C, Table C-1-2 (produce); Appendix C, Table C-1-3 (beef, milk, pork, chicken, and eggs); and Appendix C, Table C-1-4 (fish). Wet weight to dry weight conversion factors were also obtained from the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c).

6.2.2.3 Percentage of Contaminated Food

The percentage of home grown food consumed by the individual will affect exposure, because not all of an individual's dietary intake may be contaminated. Receptors, located in a rural or suburban area, who can raise animals and grow food in gardens will have a larger percentage of their food produced locally than people living in the city.

U.S. EPA OSW, in accordance with existing U.S. EPA guidance (1990e), recommends the following assumptions regarding the percentage of contaminated food:

- With regard to aboveground and belowground produce, it is assumed that the subsistence farmer and the subsistence farmer child consumes 100 percent contaminated produce; it is assumed that 25 percent of the produce consumed by receptors for the remaining recommended exposure scenarios (adult resident, child resident, and subsistence fisher, and subsistence fisher child) is contaminated (see Appendix C, Table C-1-2).
- With regard to beef, milk, pork, chicken, and eggs, it is assumed that 100 percent of these animal tissues consumed by the subsistence farmer and the subsistence farmer child are contaminated (see Appendix C, Table C-1-3). No other receptors are assumed to consume these animal tissues.
- With regard to fish, it is assumed that 100 percent of the fish consumed by the subsistence fisher and subsistence fisher child are contaminated (see Appendix C, Table C-1-4). No other receptors are assumed to consume fish.

6.2.3 Soil Exposure Pathways

Soil ingestion, dermal exposure to soil, and inhalation of resuspended dust are potential soil exposure pathways. For the purpose of RCRA combustion permitting decisions, U.S. EPA OSW recommends that soil ingestion be considered in all risk assessments. However, dermal exposure to soil and inhalation of resuspended dust are currently recommended for evaluation only if site-specific exposure setting characteristics require that these exposure pathways be evaluated. Based on air dispersion modeling and deposition of COPCs, emission concentrations in soil will vary with distance from the source. Potential routes of exposure should be determined by 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.

6.2.3.1 Soil Ingestion

Children and adults are directly exposed to COPCs in soil when they consume 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 measure the quantities of nonabsorbable 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 consumption. 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 nonfood items, such as soil, chalk, and crayons. Such behavior is considered a temporary behavior and a normal part of a child's 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). U.S. EPA risk assessment documents do not identify a default "pica" soil ingestion rate (U.S. EPA 1989e; 1989f ; 1991b). Therefore, U.S. EPA OSW does not recommend addressing pica behavior as part of risk assessments.

If available information indicates that there are children exhibiting pica behavior in the assessment area, and it is determined that these children represent a special subpopulation potentially receiving significant exposure (see Chapter 4), these children should be considered for evaluation. This evaluation should be made on a case-by-case basis based on site-specific exposure setting characterization.

6.2.3.2 Dermal Exposure to Soil

For the purpose of RCRA combustion permitting decisions, U.S. EPA OSW does not typically recommend the use, in the evaluation of recommended exposure scenarios, of the pathway of dermal exposure to COPCs through contact with soil. However, site-specific exposure setting characteristics may require that this exposure pathway be evaluated, therefore, this section discusses dermal soil exposure.

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 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 1996g; 1995h).

Humans are exposed to COPCs by absorption through the skin when it comes into contact with contaminated soil. Factors that affect dermal exposure include (1) surface area, (2) contact time, (3) contact amount, (4) amount of time spent near the combustion source, and (5) fraction of COPCs absorbed through the skin. In general, an increased dose of COPCs potentially can 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.

As duration for which the contaminated soil stays in contact with the skin increases, so does the amount of COPCs that can be absorbed. Contact time refers to the duration of time each day that contact with soil is possible. Dermal exposure is also affected by the amount of time, each day, spent in the vicinity of the combustion source at which soil is likely to be exposed to emitted pollutants. Indoor dust and outdoor soil may both increase the daily contact. Seasonal exposure can also be considered, because regional climate will influence this variable.

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 cannot be readily absorbed through the skin, the daily intake of the COPC may be small even if other exposure characteristics, such as contact time, are favorable. However, if either a facility of a permitting authority feel that site-specific conditions indicate dermal exposure to soil may contribute significantly to total soil-related exposures, U.S. EPA OSW recommends following the methodologies described in the U.S. EPA NCEA methodology document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press).

6.2.3.3 Soil Inhalation Resulting from Dust Resuspension

U.S. EPA OSW does not typically recommend the use of soil inhalation exposure pathway of resulting from dust resuspension in the evaluation of recommended exposure scenarios. However, site-specific exposure setting characteristics may require that this exposure pathway be evaluated; this section discusses exposure to soil resulting from dust resuspension.

Inhalation of soil resulting from dust resuspension may be an issue for site-specific exposure scenario locations at which there is little vegetative cover. Application of available dust resuspension exposure estimating methodologies to deposited combustion unit emissions indicates that dust resuspension by wind erosion is not a significant pathway (U.S. EPA 1990e). Wind erosion may resuspend pollutants in contaminated soil as particulates in the air. As dust is resuspended, receptors may inhale the pollutant particles (direct inhalation of particulate matter is addressed separately). The amount resuspended depends on (1) the moisture content of the soil, (2) the fraction of vegetation cover, (3) the wind velocity, (4) soil particle size, (5) the pollutant concentration in the soil, and (6) the size of the contaminated area. Methodologies have been developed to assess the exposure to pollutants resuspended by wind erosion for landfills and Superfund sites (U.S. EPA 1985a; 1988b; 1994q); U.S. EPA OSW recommends that facilities consult these reference documents if this exposure pathway must be evaluated because of site-specific exposure scenario location conditions. Also, U.S. EPA OSW recommends reviewing the methodologies described in the U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press).

6.2.4 Water Exposure Pathways

Water exposure pathways can be used to determine COPC concentrations in drinking water obtained from surface water or collected precipitation (e.g., cisterns). Water exposure pathways are also used to determine the COPC concentration in fish. Daily exposures of individuals using these water sources for various purposes—such as fishing and drinking water—can be estimated by using various models.

Site-specific information should be used to determine the appropriate exposure pathways for each assessment area. The way in which water is used—whether it is collected precipitation or a surface water body, such as a lake, farm pond, or city reservoir—will determine possible exposure pathways. Use of 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 the 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 combustion unit.

U.S. EPA (1990e) recommended that the water input variables be varied to determine a range of exposures. An average exposure scenario might be represented by an individual that fishes and obtains drinking water from the same water source. A worst-case possibility may involve a person who (1) uses drinking water from a cistern that collects precipitation, and (2) fishes in a small farm pond.

Because annual ground-level concentrations of COPCs generally decrease with distance from the source, important factors in determining the water concentration include (1) the location of the precipitation collection apparatus, (2) surface water body onto which emitted COPCs are deposited, and (3) 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 Drinking Water Exposure from Surface Water Sources

For evaluation of a surface water body as a drinking water source, exposure is affected by the concentration of the COPC in the water, the daily amount of water ingested, and the percentage of time that the individual spends in the area serviced by that water supply system. The COPC concentration in a surface water body can be calculated as described in Chapter 5 and Appendix B; which includes consideration of contribution of COPC loading from the surrounding watershed. U.S. EPA OSW recommends that water consumption rates specified in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c) be used as described in Appendix C.

Consistent with previous U.S. EPA guidance (U.S. EPA 1990e), U.S. EPA OSW recommends that it typically be assumed that treatment processes for drinking water do not alter the deposited COPCs.

6.2.4.2 Drinking Water Exposure from Ground Water Sources

For the purpose of RCRA combustion permitting decisions, U.S. EPA OSW does not typically recommend the use, in the evaluation of exposure scenarios, of the pathways of drinking water exposure from ground water sources. Application of the methodology to combustion units has indicated that this is not a significant exposure pathway (U.S. EPA 1990e). However, COPCs may—because of special site-specific exposure scenario locations—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 used for drinking water purposes or a karst environment in which the local surface water significantly affects the quality of ground water used as a drinking water source. The methodology developed to calculate risks from the ground water pathway was originally intended for use in evaluating impacts of the landfilling of municipal sludge (U.S. EPA 1990e; 1994q). U.S. EPA OSW recommends that facilities consult these reference documents if this exposure pathway must be evaluated because of site-specific exposure setting characteristics.

6.2.4.3 Dermal Water Exposure

U.S. EPA OSW does not typically recommend the use, in the evaluation of exposure scenarios, of the dermal water exposure pathway. However, if the surface water body affected by combustion unit 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 (1) the surface area of exposed skin, (2) the COPC concentration in the water, (3) the permeability of the skin to the COPC, and (4) the length of time that the individual is in contact with the water.

6.2.4.4 Ingestion of Fish

U.S. EPA OSW recommends that fish ingestion rates specified in the 1997 *Exposure Factors Handbook* (U.S. EPA 1997c) be used as described in Appendix C. Factors that affect human exposure by ingestion of fish from a surface water body affected by combustion unit emissions include (1) sediment and water COPC concentrations, (2) the types of fish and shellfish consumed, (3) the ingestion rates for the various fish and shellfish groups, and (4) the percent of dietary fish caught in the surface water body affected by the combustion unit. 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. 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 EXPOSURE FREQUENCY

The receptors in each recommended exposure scenario are assumed to be exposed to all of the exposure scenario-specific exposure pathways 350 days per year (U.S. EPA 1989e; 1991b; 1991d). This assumption is based on the conservative estimate that all receptors spend a minimum of 2 weeks at a location other than the exposure scenario location selected in Section 4.3.

6.4 EXPOSURE DURATION

Exposure duration is the length of time that a receptor is exposed via a specific exposure pathway. Although a receptor is no longer exposed to COPCs via the direct inhalation exposure pathway after an emission source ceases operation, a receptor is exposed via the indirect exposure pathways for as long as the receptor remains in the assessment area. Therefore, U.S. EPA OSW recommends using default RME values to estimate exposure duration for specified receptors.

Consistent with U.S. EPA (1990e), U.S. EPA OSW assumes 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 unit emissions. For existing facilities, U.S. EPA (1990e) assumes that this period of time can be represented by default time periods of 30, 60, or 100 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 (estimated to be possibly as long as 60 or 100 years), because it is an integral part of the facility operations. These assumptions are reasonable for a hazardous waste emission source, such as an industrial boiler burning a continuous stream of facility hazardous waste.

Although a combustion unit may remain in the same location for 100 years—and a person may have a lifetime of exposure to emissions from that combustion unit—U.S. Bureau of the Census data (1986) on population mobility 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 recommended by U.S. EPA OSW are presented in the following table.

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
Subsistence Fisher	30 years	U.S. EPA 1990f; 1994r
Subsistence Fisher Child	6 years	Assumed to be the same as the Child Resident
Subsistence Farmer	40 years	U.S. EPA 1994l; 1994r
Subsistence Farmer Child	6 years	Assumed to be the same as the Child Resident

6.5 AVERAGING TIME

For noncarcinogenic COPCs, U.S. EPA OSW recommends that a value of exposure duration (years-as specified for each receptor in Section 6.4) x 365 days/year be used 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 that COPC exposure pathway, because the exposure duration and, therefore, the quantity of exposure, will vary.

U.S. EPA OSW recommends that carcinogenic exposures for different receptor ages be evaluated separately, because the daily activities of these receptors (and, as described in Section 6.6, body weights) vary, including (1) the amounts of food and water consumed, (2) the types of food consumed, and (3) 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 quantification of carcinogenic COPC exposure depends on the duration of exposure, the age of the receptor is important. 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 1990e). 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) considered 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) recommends a 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, U.S. EPA OSW recommends that a longer averaging time and the adult body weight be used 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. However, consistent with other previous U.S. EPA guidance (U.S. EPA 1991b; 1994r), U.S. EPA OSW defines childhood as being an exposure duration of 6 years. It should be noted 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.

6.6 BODY WEIGHT

The choice of body weight for 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, consistent with other U.S. EPA guidance, U.S. EPA OSW recommends the child (exposure duration of 6 years) weight as 15 kilograms be used in the risk assessment (U.S. EPA 1991b; 1994r; 1994g).

The daily intake for an exposure pathway is expressed as the dose rate per body weight. Because children have lower body weights, typical ingestion exposures per body weight, such as for soil, milk, and fruits, are substantially higher for children—which is the primary reason for evaluating the child resident scenario (U.S. EPA 1996g). However, the use of these two body weights may not account for significant differences between weights of infants and toddlers or weights of teenagers and adults. It is important to remember, however, that the average body weight, not the actual chronological age, defines a child; obviously, the weight of a child changes significantly over the first several years. The average weight used is assumed to be a realistic average estimate for an exposure duration of 6 years that overestimates the weight of the child for the early years and then underestimates it for the later years (U.S. EPA 1996g).

Chapter 7
Risk and Hazard Characterization

What's Covered in Chapter 7:

- Individual Risk and Hazard Estimation
- Quantitative Estimation of Cancer Risk
- Quantitative Estimation of Noncancer Effects
- Target Levels
- Acute Exposure Resulting from Direct Inhalation

Risk characterization must exhibit the core values of transparency, clarity, consistency, and reasonableness. The final step of a risk assessment is the calculation of the upper-bound excess lifetime cancer risks (risk) and noncarcinogenic hazards (hazard) for each of the pathways and receptors identified in Chapter 4. Risks and hazards are then summed for specific receptors, across all applicable exposure pathways, to obtain an estimate of total individual risk and hazard for specific receptors.

Risk from exposure to combustion emissions is the probability that a receptor will develop cancer, based on a unique set of exposure, model, and toxicity assumptions. The slope factor is used in risk assessments to estimate and upper bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a potential carcinogen. For example, a risk of 1×10^{-5} is interpreted to mean that an individual has no more than, and likely less than, a one in 100,000 chance of developing cancer from the exposure being evaluated. In contrast, hazard is quantified as the potential for developing noncarcinogenic health effects as a result of exposure to COPCs, averaged over an exposure period. A hazard is not a probability but, rather, a measure (calculated as a ratio) of the magnitude 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 are typically characterized for a single receptor and are referred to as individual risks and hazards (U.S. EPA 1989e; 1994g; NC DEHNR 1997).

At least one U.S. EPA guidance document, concerning the characterization of risks and hazards associated with combustion facilities, suggests that population risks and hazards should be calculated in addition to individual risks (U.S. EPA 1993h). Population risk is defined as the aggregate risk of the exposed population; it takes into account the risk associated with various exposure scenarios and the number of individuals represented by each exposure scenario. Therefore, U.S. EPA OSW recommends that the risk assessment address only the individual risks and hazards; calculation of population risks and hazards is not required. However, if a permitting authority feels that site-specific conditions indicate calculation of population risks should be considered, U.S. EPA OSW recommends following the methodologies described in U.S. EPA NCEA document, *Methodology for Assessing Health Risks Associated with Multiple Exposure Pathways to Combustor Emissions* (In Press).

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 ESTIMATION OF INDIVIDUAL RISK AND HAZARD

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 by following the procedures described in Chapters 2 through 6 and Appendixes B and C will provide (1) quantitative and qualitative estimates of risk and hazard associated with exposure to COPCs, (2) estimates of health effects associated with exposure to lead, (3) evaluation of infant exposure to 2,3,7,8-TCDD TEQ present in breast milk, and (4) evaluation of acute exposure resulting from direct inhalation.

7.2 QUANTITATIVE ESTIMATION OF CANCER RISK

As described above, for carcinogenic chemicals, 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). These risks are calculated as follows:

$$Cancer\ Risk = LADD \cdot CSF \qquad \text{Equation 7-1}$$

where

$LADD$ = Lifetime average daily dose (mg/kg-day)
CSF = Cancer slope factor $(\text{mg/kg-day})^{-1}$

Within a specific exposure pathway, receptors may be exposed to more than one COPC. The total risk associated with exposure to all COPCs through a single exposure pathway is estimated as follows (U.S. EPA 1989e):

$$Cancer\ Risk_T = \sum_i Cancer\ Risk_i \qquad \text{Equation 7-2}$$

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

At particular exposure scenario locations, receptors may be exposed through a number of exposure pathways (see Table 4-1). Risks from multiple exposure pathways should be summed for a given receptor specific to each recommended exposure scenario. That is, risks should be summed across the receptor-exposure pathway combinations, which are identified in Table 4-1. In the context of risk assessments which evaluate the emissions from hazardous waste combustion units, the risks from all RCRA regulated combustion units that are permitted, have interim status, or expected to be constructed, should be summed for each receptor. For fugitive emissions from storage and handling of hazardous, the risk associated with fugitive emissions should be added to the risks from the combustion unit for each receptor at each exposure scenario location. For example, if a facility operates both an incinerator and a boiler that burn hazardous waste, then the risks from both types of units should be summed across all the units for each receptor. The total risk posed to a receptor is the sum of total risks from each individual exposure pathway expressed as follows:

$$Total\ Cancer\ Risk = \sum CancerRisk_T \qquad \text{Equation 7-3}$$

where

$Total\ Cancer\ Risk$ = Total cancer risk from multiple exposure pathways
$Cancer\ Risk_T$ = Total cancer risk for a specific exposure pathway

Equations used to calculate dose and risk levels are presented in Appendix C. Appendix A-3 presents oral and inhalation slope factors (*CSF*) for many potential COPCs. However, for each risk assessment, the IRIS and HEAST databases should be checked for updated values. If toxicity values for COPCs not identified in Appendix A-3 are included in the risk assessment, *CSF*s for these compounds can be obtained from the following sources, listed in the preferred order: (1) U.S. EPA's IRIS (U.S. EPA 1996a) and (2) U.S. EPA HEAST (U.S. EPA 1994b).

In the assessment of carcinogenic risk from COPCs, U.S. EPA-derived or reviewed health benchmarks (*CSFs*, *URFs*, and *Inhalation CSFs*) are recommended. However, for numerous compounds, a complete set of inhalation and oral EPA-derived health benchmarks are not available. In such cases, the health benchmarks presented in Appendix A-3 were calculated based on available U.S. EPA-derived benchmarks values.

If relevant information is not available from these sources, the applicant should contact the appropriate permitting authority, which may be able to assist in developing the necessary toxicity values. For example, Minimum Risk Values published by the Agency for Toxic Substances and Disease Registry (ASTDR) may be used at the discretion of the permitting authority.

7.3 QUANTITATIVE ESTIMATION OF POTENTIAL FOR NONCANCER EFFECTS

Standard risk assessment models assume that noncarcinogenic effects, exhibit a threshold; that is, there is a level of exposure below which no adverse effects will be observed (U.S. EPA 1989e). The potential for noncarcinogenic health effects resulting from exposure to a chemical is generally assessed by (1) comparing an exposure estimate (see Chapter 6) to an *RfD* for oral exposures, and (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 specific exposure duration. 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 1989e).

The exposure durations assumed for the exposure pathways identified in Table 4-1 range from subchronic to chronic in relative length. However, chronic *RfDs* and *RfCs* should be used to evaluate all exposure pathways. In the absence of a chronic *RfD*, a subchronic *RfD* with an Uncertainty Factor (3 to 10) can be considered. The comparisons of exposure estimates and COPC-specific air concentrations to *RfD* and *RfC* values, described above, are known as hazard quotients (*HQ*), which are calculated as follows:

$$HQ = \frac{ADD}{RfD} \quad \text{or} \quad HQ = \frac{C_a}{RfC} \qquad \text{Equation 7-4}$$

where

HQ = Hazard quotient (unitless)
ADD = Average daily dose (mg/kg-day)
C_a = Total COPC air concentration (mg/m^3)
RfD = Reference dose (mg/kg-day)
RfC = Reference concentration (mg/m^3)

It should be noted that each program office within U.S. EPA determines the 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). Generally, the more that the *HQ* value exceeds 1, the greater is the level of concern. 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). It should also be noted that background exposures may be an important consideration in setting safe levels. This is because non-cancer effects are generally modeled as thresholds. In specific cases, a permitting authority may elect to adjust the *HQ* downward to account for any exposure that individuals may have from other sources.

As with carcinogenic chemicals in a specific exposure pathway, a receptor may be exposed to multiple chemicals associated with noncarcinogenic health effects. The total noncarcinogenic hazard for each exposure pathway is calculated by following the procedures outlined in U.S. EPA (1986e) and U.S. EPA (1989e). Specifically, the total noncarcinogenic hazard attributable to exposure to all COPCs through a single exposure pathway is known as a hazard index (*HI*). Consistent with the procedure for addressing carcinogenic risks, the noncarcinogenic hazards from all RCRA regulated combustion units that are permitted, have interim status, or are expected to be constructed, should be summed for each receptor. Also, noncarcinogenic hazard from fugitive emissions sources, should also be included in the calculation of the *HI* for each receptor. The *HI* is calculated as follows:

$$HI = \sum_i HQ_i \qquad \text{Equation 7-5}$$

where

HI = Total hazard for a specific exposure pathway
HQ_i = Hazard quotient for COPC i

This summation methodology assumes that the health effects, of the various COPCs to which a receptor is exposed, are additive. Specifically, this methodology is a simplification of the *HI* concept because it does not directly consider the portal of entry associated with each exposure pathway or the often unique toxic endpoints and toxicity mechanisms of the various COPCs.

As discussed in Section 7.2 for carcinogenic risks, a receptor may be exposed to COPCs associated with noncarcinogenic health effects through more than one exposure pathway. For the purposes of the risk assessment, it is reasonable to estimate a receptor's total hazard as the sum of the *HI*s for each of the exposure pathways identified in Table 4-1. Specifically, a receptor's total hazard is the sum of hazards from each individual exposure pathway, expressed as follows:

$$Total\ HI = \sum HI \qquad \text{Equation 7-6}$$

where

$Total\ HI$ = Total hazard from multiple exposure pathways
HI = Total hazard for a specific exposure pathway

Consistent with U.S. EPA guidance (1989e), all total *HI*s exceeding the target hazard level are further evaluated. The total *HI* for an exposure pathway can exceed the target hazard level as a result of the presence of either (1) one or more COPCs with an *HQ* exceeding the target hazard level, or (2) the summation of several COPC-specific *HQ*s that are each less than the target hazard level. In the former case, the presence of at least one COPC-specific hazard greater than the target hazard level is interpreted as indicating the potential for noncarcinogenic health effects. In the latter case, a detailed analysis is required to determine whether the potential for noncarcinogenic health effects is accurately estimated by the total *HI*, because the toxicological effects associated with exposure to multiple chemicals, often through different exposure pathways, may not be additive; therefore, the total *HI* may overestimate the potential for noncarcinogenic health effects. To address this issue, COPC-specific hazards are summed according to major health effects and target organs or systems (U.S. EPA 1989e). It is especially important to consider any differences related to exposure route; this process is referred to as the segregation of the *HI*. Technically, segregation of the HI based only on target organs or systems is a simplification of HI. Ideally, the HI should be segregated considering also the often unique mechanisms of toxicity of the various compounds to which receptors may be exposed. However, segregating the HI based on mechanisms of toxicity is beyond a screening level or initial risk evaluation approach.

The highest segregated *HI* resulting from this process is considered. If the segregated *HI* exceeds the target hazard level, there is a potential for noncarcinogenic health effects. However, if the segregated *HI* is less than the target hazard level, the total *HI* of all COPC-specific results likely is too conservative, and noncarcinogenic health effects are not likely to result from exposure to COPCs.

Appendix A, Table A-2 identifies target organs and systems that are affected by each COPC in each exposure route. Appendix A-3 presents *RfDs* and *RfCs* for these same COPCs. If COPCs not identified in Appendix A-3 are included in the risk assessment, *RfDs* and *RfCs* for these compounds can be obtained from the following sources, listed in the preferred order: (1) U.S. EPA IRIS (U.S. EPA 1996a), and (2) U.S. EPA HEAST (U.S. EPA 1994b).

In the assessment of noncarcinogenic risk from COPCs, U.S EPA-derived or reviewed health benchmarks (*RfD*s, *RfC*s) are recommended. However, for numerous compounds, a complete set of inhalation and oral EPA-derived health benchmarks are not available. In such cases, the health benchmarks presented in Appendix A-3 were calculated 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; the *RfC* was calculated 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 based on a route-to-route extrapolation, which assumes that the toxicity of the given compound is equivalent over all routes of exposure.

This process does introduce uncertainty into the risk assessment. By using this method, the risk assessor must assume 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 risk assessor is 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, a qualitative assessment of the toxicity information available for the compound and exposure route should be performed. This will enable the risk assessor to make a well informed decision concerning the validity of values calculated based on route-to-route extrapolation. This qualitative assessment should also be included in the uncertainty section of the risk assessment.

If relevant information is not available from these sources, the applicant should work with the appropriate regulatory agency 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.

7.4 TARGET LEVELS

Target levels are risk management based and set by the regulatory authority. Target values are not a discrete indicator of observed adverse effect. If a calculated risk falls within target values, a regulatory authority may, without further investigation, conclude that a proposed action does not present an unacceptable risk. A calculated risk that exceeds these targets, however, would not, in and of itself, indicate that the proposed action is not safe or that it presents an unacceptable risk. Rather, a risk calculation that exceeds a target value triggers further careful consideration of the underlying scientific basis for the calculation.

7.5 ACUTE EXPOSURE RESULTING FROM DIRECT INHALATION

In addition to long-term chronic effects, short-term or acute effects should be considered from direct inhalation of vapor phase and particle phase COPCs. It is assumed that short-term emissions will not have a significant impact through the indirect exposure pathways (as compared to impacts from long-term emissions). Therefore, acute effects are only evaluated through the short-term (maximum 1-hour) inhalation of vapors and particulates exposure pathway of the acute risk scenario. U.S. EPA OSW recommendations for where and when to evaluate the acute risk scenario in completing a risk assessment is described in Sections 4.2 and 4.3. In order to establish acute inhalation exposure criteria (AIEC), it was necessary to identify and evaluate (1) existing guidelines for acute inhalation exposure, and (2) existing hierarchal approaches for developing acute inhalation exposure levels. Hierarchal approaches are composed of existing guidelines for acute inhalation exposure, ranked in order of applicability and technical basis, and all being protective of the general public. It should be noted that hierarchial approaches are needed because no single organization or methodology has developed acute criteria values or benchmarks for all of the potential compounds of concern.

7.5.1 Existing Hierarchal 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: (1) American Conference of Governmental Industrial Hygienists (ACGIH 1996); (2) Occupational Safety and Health Administration (NIOSH 1994); (3) National Institute of Occupational Safety and Health (NIOSH 1994); (4) American Industrial Hygiene Association (AIHA 1997); (5) National Research Council Committee on Toxicology (NRC COT 1986, U.S. EPA 1987b); (6) U.S. EPA (U.S. EPA 1987b); (7) Agency for Toxic Substances and Disease Registry (ATSDR 1997); (8) California Environmental Protection Agency (CEPA 1995); (9) National Advisory Committee (NAC 1997); and (10) Department of Energy (DoE 1997b); Subcommittee on Consequence Assessment and Protective Actions (SCAPA 1997b). 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.

Hierarchal approaches have been developed by a variety of organizations and teams of organizations for establishing acute inhalation exposure guidelines to protect the general public. These development organizations include:

- U.S. EPA Region 10 (U.S. EPA 1996a);
- Federal Emergency Management Agency, Department of Transportation (DoT), and U.S. EPA (U.S. EPA 1993k);
- U.S. EPA Region 3 (EPA 1996b);
- Department of Defense (DoD 1996); and
- Department of Energy (DoE) (SCAPA 1997a).

The acute inhalation exposure guidelines developed by these organizations are generally a very heterogenous group, developed to protect different subpopulations against different effects and apply to various exposure durations. Therefore, the hierarchal approaches developed by the first four organizations listed above have 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 hierarchal approaches developed using safety factors, the DoE's Emergency Management Advisory Committee's Subcommittee on Consequence Assessment and Protective Actions (SCAPA) has developed temporary emergency exposure limits (TEELs) based on statistical analyses between existing guidelines for acute inhalation exposure and AIHA emergency response planning guidelines (ERPG) (Craig et al. 1995). 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).

7.5.2 U.S. EPA OSW Recommended Hierarchal Approach

After reviewing the existing hierarchal approaches, U.S. EPA OSW recommends the following approach. This approach is based on existing acute inhalation values that do not require the use of arbitrary safety factors and are intended to protect the general public from discomfort or mild adverse health effects over 1-hour exposure periods. It includes level 1 acute inhalation exposure guidelines (AEGL-1), level 1 emergency response planning guidelines (ERPG-1), and level 1 acute toxicity exposure levels (ATEL-1); supplemented with DoE TEELs and the SCAPA toxicity-based approach. The hierarchal approach is summarized below:

1. AEGL-1 (NAC 1997)
2. ERPG-1 (AIHA 1996; SCAPA 1997b)
3. ATEL-1 (Cal/EPA 1995)
4. TEEL-1 (SCAPA 1997a)

5. SCAPA Toxicity-based approach (DoE 1997a)

The hierarchy is presented in order of preference; from 1 (most preferred) to 5 (least preferred). This preference is based on (1) applicability to a 1-hour exposure duration for protection of the general public (versus only occupational exsposure), and (2) level of documentation and associated review. It should also be noted that the hierarchy approach of preference for AEGL-1 and ERPG-1 values is also consistent with the CAAA 112r Final Rule, Risk Management Program.

To obtain a COPC-specific AIEC, one should begin with review of the AEGL-1 values specific to the COPC of interest. AEGL-1 values are currently available for 12 compounds. The Federal Register (October 30, 1997) provides a list of the proposed AEGL-1 values, which can be accessed through the web (www.EPA.FEDRGSTR). If there is not an available AEGL-1 value for a respective COPC, review the ERPG-1 values, and so forth until an AIEC value is obtained specific to the COPC of interest. Appendix A-4 provides an abbreviated listing of example AIECs developed based on values currently available for the hierarchal approach presented above.

It should also be noted that DoE's approach for developing TEELs contains existing acute inhalation exposure values that were not specifically developed to protect the general public from discomfort or mild adverse health effect over 1-hour exposure periods. AEGL-1, ERPG-1, and ATEL-1 values are all developed in accordance with the principals outlined in NRC's Committee on Toxicology (COT) guidelines for developing a detailed step-by-step process for developing defensible acute exposure levels. However, because AEGL, ERPG, and ATEL values are only available for a limited number of compounds, it becomes necessary to use TEEL values (currently available for 471 compounds).

For any COPCs for which acute inhalation exposure values cannot be developed using DoE's TEEL approach, AIECs can be developed following the toxicity-based approach used by SCAPA (Tier 5).

To characterize the potential for adverse health effects from acute exposure to COPC-specific emissions, the acute air concentration (C_{acute}) resulting from maximum emissions over a 1-hour period should be compared to the COPC-specific AIEC to calculate the acute hazard quotient (AHQ_{inh}) (see Appendix C, Table C-4-1). See Chapter 3 for discussion on air dispersion modeling related to obtaining 1-hour maximum values to calculate C_{acute} (see Appendix B, Table B-6-1). The AHQ_{inh} can be calculated as follows:

$$AHQ_{inh} = \frac{C_{acute} \cdot 0.001}{AIEC} \qquad \text{Equation 7-7}$$

where

AHQ_{inh} = Acute hazard quotient (unitless)
C_{acute} = Acute air concentration ($\mu g/m^3$)
$AIEC$ = Acute inhalation exposure criteria (mg/m^3)
0.001 = Conversion factor ($mg/\mu g$)

Acute hazard quotients should be calculated 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. Target levels for acute hazard quotient evaluation is a risk management decision and will be set by the permitting authority.

Chapter 8
Uncertainty Interpretation for Human Health Risk Assessment

What's Covered in Chapter 8:

- Uncertainty and Limitations of the Risk Assessment
- Types of Uncertainty
- Qualitative Estimates of Uncertainty
- Quantitative Estimates of Uncertainty
- Risk Assessment Uncertainty Discussion

This section describes how to interpret 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.*

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 if the most accurate data with the most sophisticated models are used. The methodology outlined in this document relies on a combination of point values—some conservative 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; instead, it is known that the values combine many conservative factors and are likely to overstate actual risk (Hattis and Burmaster 1994). Therefore, a formal uncertainty analysis is required to determine the degree of conservatism. This section discusses the types of uncertainty and the areas in which uncertainty can be introduced into an assessment. In addition, this section discusses 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, and cannot 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 will not 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) 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; decision rule is of greatest concern to the risk manager.

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 (1) animal models used as surrogates for testing human carcinogenicity, (2) the dose-response models used in extrapolations, and (3) the computer models used to predict the fate and transport of chemicals in the environment. The use of 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 risk assessor needs to consider the importance, in consultation with the modeler, of excluded variables on a case-by-case basis, because 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 specified for use in this document were selected on the basis of scientific policy. Therefore, the air dispersion and deposition model and the indirect exposure models were selected, because they provide the information needed to conduct indirect assessments and are considered by U.S. EPA to be state-of-the-science models. This choice of models could also be considered under decision rule uncertainty. ISCST3—the air dispersion model recommended for use—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 (1) select input variables related to deposition, and (2) validate modeled deposition rates. Because dry deposition of vapor phase materials is evaluated external to the air dispersion model, the plume is not depleted and, as a result, mass balance is not maintained. The effect of this would be to overestimate deposition, but the magnitude of the overestimation is unknown. Mass balance is maintained for other forms of deposition (such as wet deposition and particle phase dry deposition). Long-range transport of pollutants into and out of the areas considered was not modeled, resulting in an underestimation of risk attributable to each facility.

In addition to air dispersion modeling, the use of 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 conservative for waterbodies located in estuarine environments with tidal influence. Because tidal influence is not considered in the models presented in Chapter 5, the resultant dilution of COPC concentrations in water and sediments likely caused by tidal influence will not be considered in the risk assessment. Thus, the risk assessment results will likely be more conservative for tidally influenced waterbodies than for those waterbodies that are not tidally influenced. Permitting decisions based on risk estimates for estuarine environments should 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 included in the analysis. Constituents that will be identified on the basis of guidance 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 U.S. EPA default values in the analysis. These include inhalation and consumption rates, body weight, and lifespan, which are standard default values used in most U.S. EPA risk assessments. Inhalation and consumption rates are highly correlated to body weight for adults. Using a single point estimate for these variables instead of a joint probability distribution ignores 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 U.S. EPA-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 U.S. EPA risk assessments. This process is utilized to account for much of the uncertainty and variability associated with the health benchmarks values. With the exception of the dioxin toxicity equivalency methodology, health benchmarks which can be found on IRIS, have been verified through U.S. EPA work groups. This document will not estimate the uncertainty in using U.S. EPA verified health benchmarks or the dioxin toxicity equivalency methodology.

8.3 DESCRIPTION OF QUALITATIVE UNCERTAINTY

Often, sources of uncertainty in a risk assessment can be determined but cannot be quantified. For example, this can occur when a factor is known or expected to be variable, but no data are available (e.g., presence of COPCs without toxicity data, amount of time that people at a specific site spend outdoors). In this case, 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 the uncertainty cannot be quantified 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 uncertainty can be presented only qualitatively, the possible direction and orders of magnitude of the potential error should be considered.

8.4 DESCRIPTION OF QUANTITATIVE UNCERTAINTY

Knowledge of experimental or measurement errors can also be used 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 can be used to quantify uncertainty. In many cases, uncertainty associated with particular variable values or estimated risks can be expressed quantitatively and further evaluated with variations of sensitivity analyses. Finkel (1990) identified a six-step process for producing a quantitative uncertainty estimate:

- 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 should be quantified or reached individually.

- 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.

- 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.

- Combine the individual distributions into a composite uncertainty distribution.

- 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.

- 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 a detailed quantitative treatment of uncertainty is required, statistical methods are employed. Two approaches to a statistical treatment of uncertainty with regard to variable values are described here and were used in this analysis where appropriate. The first is to use an appropriate statistic to express all variables for which uncertainty is a major concern. For example, if a value used is from a sample (such as yearly emissions from a stack), the mean and standard deviation should both be presented. If the sample size is very small, it may be appropriate to (1) give the range of sample values and use a midpoint as a best estimate in the model, or (2) use the smallest and largest measured value to obtain two estimates that bound the expected true value. Selection of the appropriate statistic depends on the amount of data available and the degree of detail required. Uncertainties can be propagated 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 used in a risk analysis. A probability distribution of expected values is then developed 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. Numerical or statistical methods are often In Monte Carlo simulations, for example, 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 the risk assessor to choose the value corresponding to the appropriate percentile in the overall distribution. For example, the risk assessor can 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.

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 of facilities that burn hazardous waste to fully explain the areas of uncertainty in the assessments and to identify the key assumptions used in conducting the assessments. Toward that end, a table should be added to the end of each section (e.g., stack emissions, air modeling, exposure assessment, toxicity evaluation, risk characterization) which 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 can be used to evaluate the extent to which public health protective assumptions were used in the risk assessments. They can 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, should be the most significant assumptions from each of the previous sections.

Within this guidance, identification of uncertainties and limitations are also included with the discussion of specific technical issues (e.g., TOE, estimates of emission rates, COPC selection process, quantification of non-detects) as they are presented in their respective sections. Limitations associated with parameter values and inputs to equations are presented in Appendices A, and 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 (1) the ability of the model algorithms to depict atmospheric transport and dispersion of contaminants, and (2) 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).

Uncertainties specific to other technical components (e.g., TOE, quantification of non-detects) of the risk assessment process are further described in their respective chapters or sections of this guidance.

Chapter 9
Completion of Risk Assessment and Follow-On Activities

What's Covered in Chapter 9:

- Conclusions
- Activities Following Risk Assessment Completion

This chapter summarizes the conclusion of the risk assessment and the activities that will be conducted following risk assessment completion.

It is also important to note that final risk assessments should include both human health and ecological evaluations. In addition to available U.S. EPA guidance for conducting ecological risk assessments (U.S. EPA 1997e) and Volume 63, Number 93, of the Federal Register, U.S. EPA OSW is currently finalizing an ecological risk assessment guidance document titled *U.S. EPA OSW Screening Level Ecologcial Risk Assessment Protocol*, prepared as a companion to this guidance.

9.1 CONCLUSIONS

Each risk assessment should include a Conclusions section. This section should primarily interpret the results of the risk and hazard characterization in light of the uncertainty analysis. At a minimum, it should present and interpret all risk and hazard results exceeding target levels. It should 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 the risk assessor to present, and defend, his or her position on whether actual or potential releases from combustion units pose significant risks and hazards to human populations.

9.2 ACTIVITIES FOLLOWING RISK ASSESSMENT COMPLETION

The risk assessment process does not end following the completion, submittal, and approval of a successful risk assessment report. The HHRAP has been developed to promote a consistent approach, for completing risk assessments, that (1) encourages the use of appropriate site-specific information early in the process (2) minimizes inefficient expenditure of time and resources by regulated facilities, and (3) provides a comprehensive explanation of the procedures and uncertainties involved in the process. However, completion of the risk assessment process includes more than the completion of a report; the main purpose of developing the HHRAP was to provide risk assessors with a tool for completing quality, consistent, and defensible risk assessments in a short amount of time, rather than spending years to determine which COPCs, exposure pathways, and receptors the risk assessment report should include and evaluate.

Facilities operating hazardous waste combustion units are also responsible for communicating the results of the risk assessment process to affected members of the community. One purpose of U.S. EPA OSW comprehensive explanation of the procedures and uncertainties involved in the process was to provide the facilities, risk assessors, and regulators with the tools needed to clearly communicate the procedures, results, and limitations of the risk assessment process. This is an ongoing process.

Finally, the completion of the risk assessment process involves the use of (1) site-specific environmental data, (2) various assumptions, and (3) an evolving procedure for estimating risk. U.S. EPA OSW expects that facilities will periodically review each of these factors, in and up date the process with the latest facility-specific operating and emission information, to determine whether the best data and procedures have been used to estimate the risk resulting from the operation of the facility hazardous waste combustion unit. The permit writer may establish the period for this review in the operating permit; however, significant changes involving newly available data or risk assessment procedures that significantly affect the outcome of the risk assessment process should be reviewed as they become available.